Environmental Pollution 348 (2024) 123836

Contents lists available at ScienceDirect

Environmental Pollution

journal homepage: www.elsevier.com/locate/envpol

Pesticide risk assessment in European agriculture: Distribution patterns, ban-substitution effects and regulatory implications☆

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ARTICLE INFO

Keywords:
Pesticide load
Plant protection products
Risk trade-off
Pesticide regulation
Ban substitution

ABSTRACT

This study estimates the risks of agricultural pesticides on non-target organisms and the environment by combining detailed pesticide application data for 2015 with the Danish risk indicator Pesticide Load. We quantify and map the pesticide load of 59 pesticides on 28 crops and pastures in the EU. Furthermore, we investigate how recent bans on 14 pesticides in the EU could reduce pesticide use and load. Key findings show that the highest pesticide loads per hectare occur in Cyprus and the Netherlands due to high application rates and a high proportion of vegetable production. Chlorpyrifos caused the highest pesticide load per hectare on more than half of the assessed crops before its ban. The ban of 14 pesticides between 2018 and 2023 potentially reduced pesticide loads by 94%, but unobserved substitution effects could offset pesticide load reductions. Although bans on active substances are justified to control certain endpoint risks, our results highlight the potential weaknesses of bans that merely shift risks. These findings contribute to the ongoing scientific and societal discourse on efficiently mitigating pesticides' impacts on non-target organisms and the environment. However, to improve the evaluation of pesticide use, it is vital to enhance the reporting on detailed pesticide use for individual crop-pesticide combinations.

1. Introduction

Pesticide use in agricultural systems worldwide has nearly doubled since the 1990s (FAO, 2022a). Apparent benefits of pesticides[1] include increased agricultural yields, enhanced product homogeneity, and reduced agricultural labor and energy expenses. However, pesticide use also has adverse consequences on non-target organisms and the environment, many of which are not immediately visible. These effects include damage to ecosystems, bee colony losses, contamination of soil, food, ground and drinking water and the potential adverse health effects on applicators, pickers and consumers (Leach and Mumford, 2008). Non-target organisms may absorb pesticide residues, causing chronic and lethal health conditions such as reproductive failure, cancers and tumors, and DNA damage (Wagner et al., 2014; Khan, 1980; Beaumelle et al., 2023). Human health hazards arise from the inhalation or ingestion of pesticide residues, which are suspected to increase the risk of acute and chronic diseases ranging from asthma to diabetes, Alzheimer's disease and cancer (European Environment Agency, 2023).

The negative effects of pesticide use on non-target organisms and the environment persist because they are unintended consequences that are borne by society and ecosystems beyond the immediate producer or user of the pesticide. These effects are known as externalities, also referred to as external effects or external costs. Externalities lead to inefficient market outcomes since producers and consumers do not account for these external effects in their decision-making processes. Therefore, policy regulations are necessary to monitor and control pesticide use and its associated risks. The European Commission uses two Harmonized Risk Indicators to estimate and monitor pesticide risk. Pesticides usually contain active substances and adjuvants. The active substances control the pest, while the adjuvants improve the usability or product

☆ This paper has been recommended for acceptance by Charles Wong.

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[1] In this paper, by pesticides we refer exclusively to plant protection products.

https://doi.org/10.1016/j.envpol.2024.123836
Received 10 November 2023; Received in revised form 19 March 2024; Accepted 20 March 2024
Available online 22 March 2024

performance (European Parliament and Council, 2009). The Harmonized Risk Indicator 1 is calculated as the weighted sum over the annual sold quantities of all active substances on the market (European Commission, 2019a), where the weighting depends on the classification of the active substance into one of four hazard categories. The Harmonized Risk Indicator 2 is calculated by multiplying the number of emergency authorizations granted for active substances by the same weighting as above and aggregating these calculations over all active substances on the market (European Commission, 2019a). In 2021, the Harmonized Risk Indicator 1 decreased by 6% compared to 2020 and 38% compared to 2011–2013. However, indicators based on the amount of pesticides sold, such as the Harmonized Risk Indicator 1, could create incentives for more toxic products with lower standard application rates (Möhring et al., 2019). Studies have also shown that quantity-based indicators do not correlate well with more detailed risk indicators (Reus et al., 2002; Möhring et al., 2019; Bub et al., 2022).

Adequate pesticide risk indicators should consider the toxicity of pesticides on non-target organisms and the environment (Reus et al., 2002). For example, Bub et al. (2022) used the Total Applied Toxicity indicator to measure trends in pesticide risk in Germany between 1995 and 2019. Like other indicators, such as the SYNOPS indicator or the Environmental Yardstick for Pesticides, the Total Applied Toxicity indicator only considers the environmental effects of pesticide use (Gutsche and Rossberg, 1997; Reus and Leendertse, 2000). The Environmental Impact Quotient depicts a scoring system that averages the risk potential of an active substance for farm workers, consumers and the environment by multiplying its toxicity with exposure potential to form a single value. However, this indicator poorly reflects the environmental impact of herbicides (Kniss and Coburn, 2015) and has conceptual problems with the weighting and scaling of the categories (Dushoff et al., 1994). The Danish pesticide risk indicator Pesticide Load was developed to quantify and monitor annual pesticide use and risk and is used as a basis for the pesticide taxation scheme in Denmark (Anon, 2012; Kudsk et al., 2018). The Pesticide Load method combines a product's application quantities with the product's toxicity in the three categories human health, environmental toxicity and environmental fate. Kudsk et al. (2018) estimated and mapped the pesticide load for Denmark over 2010 to 2014, aggregated on four pesticide classes. Möhring et al. (2019) and Möhring et al. (2020) worked with the Pesticide Load indicator on Swiss wheat and potato farming data sets. To our knowledge, however, the Pesticide Load method has not been used to quantify the risk of pesticide use for individual crops across the EU.

Under EU Regulation 1107/2009, which regulates the approval process for pesticides, various active substances have been banned from being used in the EU in the last decade. The bans were enforced because unacceptable risks to non-target organisms or the environment were present or could not be ruled out (i.e., European Commission (2020a,b)). Bans on active substances often trigger their replacement by approved substitutes with further consequences for non-target organisms and the environment (Liu et al., 1995; Gray and Hammitt, 2000). Tesfamichael and Kaluarachchi (2006) found that 80% of the atrazine-treated area in a study region in the US was likely to be treated with 2,4-D, bromoxynil, dicamba and nicosulfuron after an atrazine ban. However, bromoxynil was banned in the EU in 2020 due to unacceptable non-dietary risks to child residents (European Commission, 2020a). In US maize, pyrethroid and organo-phosphate insecticides were substituted by newly developed neonicotinoid insecticides, resulting in a reduced risk for mammals, birds and fish but a greater risk for honeybees (Perry and Moschini, 2020). When neonicotinoid active substances were banned in the EU due to their risk to beneficial insects, including bees, case studies on maize, oilseed rape and sunflower in seven countries concluded that pyrethroid insecticide usage increased again (Kathage et al., 2018; Scott and Bilsborrow, 2019). So far, however, no study has looked at the substitution effects and possible trade-offs of pesticide bans more generally.

Until recently, EU-wide detailed pesticide application data to quantify the risk of pesticide use were lacking. Udias et al. (2023) published an EU-wide pesticide application data set at NUTS level 3. However, they do not differentiate pesticide applications between crops. In comparison, PEST-CHEMGRIDS is a detailed pesticide application rates data set for the 20 most used active substances on ten crop classes worldwide (Maggi et al., 2019).

This paper has three main research objectives. Firstly, we quantify distribution patterns of pesticide-related risks for individual crops and active substances across the EU. We use the Pesticide Load method to estimate the risk of pesticide use in the EU in combination with the PEST-CHEMGRIDS pesticide application data set (Maggi et al., 2019). Secondly, we examine changes in pesticide load due to substitution effects for 14 active substances that have been banned in the EU between 2018 and 2023. Here, we address the uncertainty of possible substitution choices by scenario analysis, which includes scenarios of substitution without further pesticide use, e.g. through increased manual labor, and scenarios in which banned active substances are replaced by approved active substances. Thirdly, we examine regulatory implications for the EU's active substance approval and ban framework. Our findings contribute to the scientific and societal discussion on efficiently mitigating the impacts and risks of pesticides on non-target organisms and the environment.

2. Methods and data

2.1. Study region

The study region encompasses 26 member states of the European Union and the United Kingdom of Great Britain and Northern Ireland (EU26 + 1): Austria (AT), Belgium (BE), Bulgaria (BG), Croatia (HR), Czech Republic (CZ), Cyprus (CY), Denmark (DK), Estonia (EE), Finland (FI), France (FR), Germany (DE), United Kingdom of Great Britain and Northern Ireland (UK), Greece (EL), Hungary (HU), Ireland (IE), Italy (IT), Latvia (LV), Lithuania (LT), Luxembourg (LU), the Netherlands (NL), Poland (PO), Portugal (PR), Romania (RO), Slovakia (SK), Slovenia (SI), Spain (ES) and Sweden (SE). Malta is left out due to missing pesticide application data.

2.2. Pesticide load

We use the Pesticide Load method to estimate the adverse effects of pesticide use on non-target organisms and the environment (Kudsk et al., 2018). The Pesticide Load method does not consider the actual exposure to pesticides but instead estimates the relative risks of pesticide use based on quantities and properties of the sold pesticide product (Anon, 2012). The Pesticide Load can be expressed as pesticide load per unit product (L kg^{-1}), total pesticide load (L) or load per hectare (L ha^{-1}). The Pesticide Load method consists of three sub-indicators that measure the load of a pesticide product on human health, environmental toxicity and environmental fate. The load for human health focuses on operator exposure and is based on the hazard classification of the product in the form of risk phrases on the label. Each risk phrase is assigned a score from 10 to 100, where a score of 10 is given to minor hazards such as skin irritation and a score of 100 is given to possible irreversible damages such as genetic defects or cancer (see Appendix A in the supplementary data for the exhaustive list of risk phrases and their scores). The scores for each risk phrase are added up and the total score is converted into a load by dividing it by 300. The value of 300 was chosen as the norm in developing the indicator to ensure that the contribution of the human health load to the total pesticide load was close to one-third in the reference year 2007. The environmental toxicity and fate load consider several input parameters (Table 1) taken from the Pesticide Property Database (Lewis et al., 2016). For environmental toxicity, the median lethal dose LD_{50} describes the acute toxicity for birds, mammals and bees, measured as the dose required to kill half the members of the tested population (Lewis et al., 2016). The median lethal concentration LC_{50} describes the acute toxicity for fish and earthworms,

Table 1
Parameters for calculating the environmental toxicity load, environmental fate load and human health load.

	Input parameters	Unit	Reference value	Load factor (L kg^{-1})	Value for glyphosate	Calculation for glyphosate
Environmental toxicity	Birds acute	$^{LD}50$ mg per kg body weight	49	1	2000	$49/2000 \cdot 1 = 0.0245$
	Mammals acute	$^{LD}50$ mg per kg body weight	20	1	300	$20/300 \cdot 1 = 0.01$
	Fish acute	$^{LC}50$ mg per liter water	0.00021	30	100	$0.00021/100 \cdot 30 = 0.000063$
	Daphnia acute	$^{EC}50$ mg per liter water	0.0003	30	40	$0.0003/40 \cdot 30 = 0.000225$
	Algae acute	$^{EC}50$ mg per liter water	0.000025	3	19	$0.000025/19 \cdot 3 = 0.000003947$
	Aquatic plants acute	$^{EC}50$ mg per liter water	0.00036	3	12	$0.00036/12 \cdot 3 = 0.00009$
	Earthworms acute	$^{LC}50$ mg per kg soil	3.4	2	5600	$3.4/5600 \cdot 2 = 0.001214$
	Bees acute	$^{LD}50$ mg per bee	0.02	100	100	$0.02/100 \cdot 100 = 0.02$
	Fish chronic	NOEC mg per liter water	0.000115	3	1	$0.000115/1 \cdot 3 = 0.000345$
	Daphnia chronic	NOEC mg per liter water	0.000115	3	–	–
	Earthworms chronic	NOEC mg per kg soil	0.2	2	21.31	$0.2/21.31 \cdot 2 = 0.01877$
Sum						**0.075**
Environmental fate	Soil degradation	Half-life in soil $^{(DT}50^{)}$ in days	354	2.5	16.11	$16.11/354 \cdot 2.5 = 0.113771$
	Bioaccumulation	Bioconcentration factor (BCF)	5100	2.5	0.5	$0.5/5100 \cdot 2.5 = 0.000245$
	Leaching	Screening Concentration in Ground Water (SCI-GROW) index	10.91	20	0.00544	$0.00544/10.91 \cdot 20 = 0.00997$
Sum						**0.124**
Human health	Risk phrase				H318	0.233
Total Sum						**1.881**

measured as the concentration in water or soil required to kill half the members of the tested population. The median effect concentration EC_{50} describes the acute toxicity for daphnia, algae and aquatic plants and is measured as the concentration of a product at which 50% of its maximum response is observed. The no observed effect concentration NOEC describes the chronic effects for fish, daphnia, and earthworms, and it is the highest concentration for which no observable effect in long-term studies has been found. The environmental fate load is based on three input parameters: soil degradation (DT_{50} measured as the half-life of a substance in soil), bioaccumulation (accumulation of substances in the food chain measured with the bioconcentration factor (BCF)) and leaching (measured with the Screening Concentration in Ground Water (SCI-GROW) index).

The reference value for each input parameter was set by the most harmful active substance within each parameter (lowest LC_{50}/LD_{50}/EC_{50} and NOEC values, highest DT_{50}, BCF and SCI-GROW index) registered in Denmark in 2007. For each evaluated substance, the ratio between its input value and the reference value multiplied by the load factor gives the load per kg for each input parameter. The load factor represents a normative weighting of the input parameters, giving more importance to bees and pollinators, aquatic organisms and leaching to groundwater. In this study, we use a weighting system established by Anon (2012) for ordinary spray products. A different weighting should be considered for products applied to seed treatments because birds and mammals would have a higher risk of exposure. In our study, we assume that all pesticide applications are spray applications, as the pesticide application data from Maggi et al. (2019) were calibrated to the aggregated data from FAOSTAT, which does not include information on pesticide use for seed treatment in Europe. Initially, the Pesticide Load was developed to calculate the load of a pesticide product that might contain various active substances. The environmental toxicity and fate load are evaluated on the properties of a product's active substances. If a product contains multiple active substances, the load of each active substance will add up. The human health load considers a product's hazards (i.e., risk phrases), which could deviate from the sum of hazards of single active substances. Adding up the three sub-indicators human health load, environmental toxicity load and environmental fate load then gives the final pesticide load of a product. In this study, we do not have application data for pesticide products and are limited to calculating pesticide loads at the level of active substances. We, therefore, calculate the human health load of any active substance by considering its individual risk phrases.

The pesticide load calculation of glyphosate may be an example (Table 1). According to the Pesticide Property Database, glyphosate can cause severe eye damage in humans (H318), which leads to a human health load of 0.233 L kg^{-1}. Calculating the environmental toxicity load and environmental fate load gives a load of 0.075 and 0.124 L kg^{-1}, respectively. The overall pesticide load for glyphosate is the sum of the three category loads, i.e. 1.881 L kg^{-1}. Given an application rate of glyphosate at 0.2 kg ha^{-1} over 1000 ha, we can determine the load per hectare as follows: 1.881 L $kg^{-1} \cdot$ 0.2 kg ha^{-1} = 3.76 L ha^{-1}. Consequently, the total load is calculated as 3.76 L $ha^{-1} \cdot$ 1000 ha = 3760 L. We refer to Anon (2012) and Kudsk et al. (2018) for more details on the calculation and development of the Pesticide Load.

2.3. Pesticide application data

This study uses gridded data on small-scale pesticide applications from Maggi et al. (2019). The global data set contains annual active substance application rates in kg ha^{-1} for six dominating crops and four aggregated crop classes for 2015 for a grid of 5 arc-minutes (about 6 km in the study region). For each crop (class), the 20 active substances with the highest application mass worldwide were considered, resulting in 95 active substances in total in the data set. Dominating crops include maize, soybean, wheat, cotton, rice and alfalfa; aggregated crop classes include vegetables/fruits, orchards/grapes, pasture/hay and others. This study only includes 59 of the 95 active substances available because 36 active substances were banned in the EU before the publication of the pesticide application data set. Appendix B in the supplementary data lists all active substances included in this study for each crop class. Maggi et al. (2019) used spatial statistical methods to harmonize the FAO pesticide database (FAO, 2023a), which supplies data for aggregated pesticide classes on a national level, and the USGS Pesticide National Synthesis Project database, which supplies data on individual active substances only for the U.S. Globally gridded active substance application rates were estimated for 2015 based on a map of global crop-specific harvested area by Monfreda et al. (2008), soil properties, hydroclimatic and agricultural variables and socio-economic metrics. To reconcile discrepancies between the harvested area values reported by Monfreda et al. (2008) and the FAO (2023b) data we use averaged over 2010–2019, we scale harvested area values by Monfreda et al. (2008) with averaged FAO harvested area values. Subsequently, we recalculate original application rates to ensure total application amounts align with the FAO data on aggregated pesticide use (first minimization in Fig. 1).

2.4. Pesticide use scenarios

Since the publication of the PEST-CHEMGRIDS data set (Maggi et al., 2019), 14 active substances from this data set have been banned in the EU, including chlorpyrifos, mancozeb, imidacloprid and others. In practice, banned active substances could be substituted without further pesticide use, e.g. through physical control (Vincent et al., 2003) or with alternative, similar active substances (Gray and Hammitt, 2000; Perry and Moschini, 2020; Liu et al., 1995). To address substitution opportunities, we investigate four different pesticide use scenarios, a baseline scenario and three ban-substitution scenarios, to estimate the changes in pesticide load resulting from substitution (Fig. 2). The *Baseline2018* scenario includes all 59 active substances in PEST-CHEMGRIDS that were approved in the EU in 2018. The second scenario called *SubstitutePhyiscally* omits all banned active substances between 2018 and 2023, suggesting a substitution without further pesticide use, e.g. by increased physical control. The third and fourth scenarios, *SubstituteWorst* and *SubstituteBest,* simulate replacing all banned active substances with an active substance substitute. The chosen substitute is similar in mode of action, targets the same or similar pests and is authorized in the EU and approved in the member state (European Commission, 2023). We base our substitutions on various studies and recommendations (Table 2), but can only make assumptions about possible substitutions. If more than one possible substitute is available, we choose the active substance substitute with the highest median load per hectare for the scenario *SubstituteWorst* and the active substance with the lowest median load per hectare for the scenario *SubstituteBest*. For example, EFSA (2018) suggests that bromoxynil on durum wheat can be substituted with MCPA, clopyralid, 2,4-D or dicamba. In Austria, 2,4-D has the highest median load per hectare of these active substances (0.03 L ha^{-1}) and clopyralid has the lowest median load per hectare (0.000000002 L ha^{-1}). Therefore, we substitute the application of bromoxynil on durum wheat in Austria with the application of 2,4-D in *SubstituteWorst* and with clopyralid in *SubstituteBest*. Hence, we assume in the substitution scenarios that durum wheat in Austria is treated with the small-scale recalculated PEST-CHEMGRIDS application rates of 2, 4-D or clopyralid twice and not at all with bromoxynil.

Table 2 summarizes the 14 banned active substances and their substitution candidates. Due to a lack of further data, only active substances included in PEST-CHEMGRIDS can be used for substitution. If the active substance substitute was not estimated for the specific crop, we substitute the pesticide load per hectare of the banned active substance by the median pesticide load per hectare of the active substance substitute over the EU26 + 1. For example, chlorothalonil on winter rye can be substituted with azoxystrobin or pyraclostrobin according to European Crop Protection (2016). The PEST-CHEMGRIDS pesticide application data does not depict these substitutes on winter rye (or other crops in more general) in Germany. However, azoxystrobin is modeled on cereals in different countries of the EU26 + 1. Therefore, we replace the pesticide load per hectare of chlorothalonil in our calculations for Germany on winter rye with the median pesticide load per hectare of azoxystrobin on cereals of all other countries.

2.5. Constrained optimization model

We use a constrained optimization model to estimate a consistent reference data set for pesticide application rates (first minimization of Fig. 1, refer to Section 2.3) as well as cropland area and pesticide load (second minimization of Fig. 1). Our optimization models work on the scale of homogeneous response units (HRUs). HRUs are spatially delineated units at a 5° resolution grid intersected with altitude, soil and slope classes (Skalský et al., 2012). To depict geographical variation in Europe, we use NUTS (Nomenclature of territorial units for statistics) level 2 for Europe.

The objective function of the second optimization model minimizes the difference between simulated and reported cropland area for each crop on a NUTS level 2 reported by Eurostat (2023). If no cropland area data on NUTS level 2 is available, the objective function maximizes production (= area · yield) for each crop. EU-wide crop management simulations with a crop-pest model (Rasche, 2021) contribute detailed annual crop yield data for conventional and organic agriculture on the scale of HRUs for 27 crops. These crops include maize, maize silage, durum wheat, field pea, flax, oats, cotton, potato, rapeseed, rice, soybean, sugar beet, summer barley, sunflower, winter rye, winter wheat, triticale, mixed grain, olives, grapes, almonds, apples, tomatoes, dry beans, broad and horse beans, other pulses, and other vegetables. The total cropland area used for these 27 crops equals 94% of the average area for all crop production in the EU26 + 1 between 2010 and 2019 (FAO, 2023b). For alfalfa and pasture, data on cultivated area are provided by Eurostat (2023) and FAO (2023b), respectively, resulting in crop yield and/or area data for 28 crops and pasture.

One constraint of the optimization problem forces the simulated reference cropland area distribution and crop production values to agree

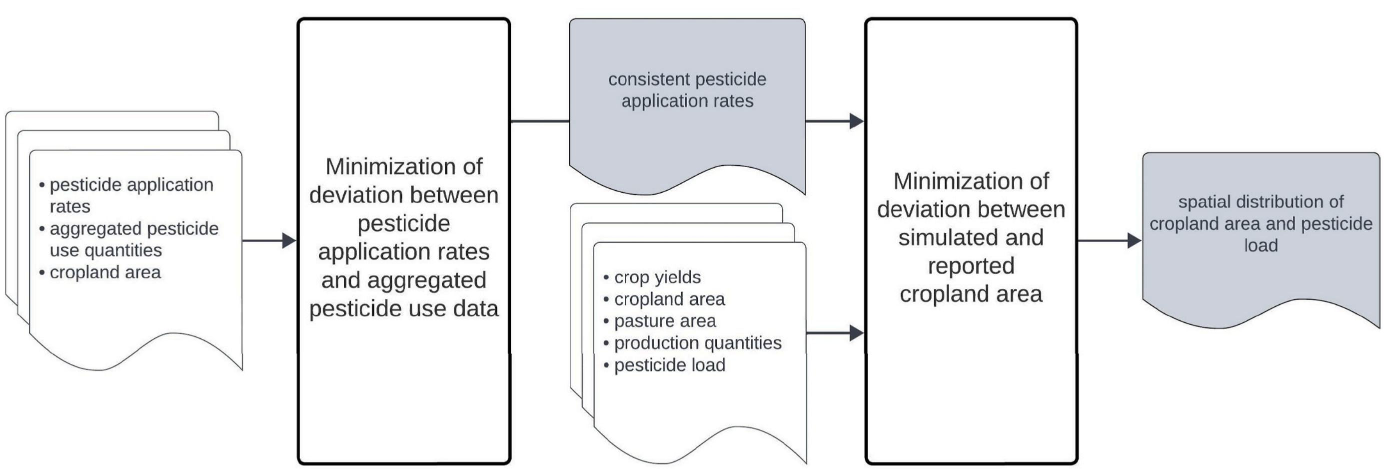

Fig. 1. Schematic representation of data and methods for this study. White curved boxes depict input data, grey curved boxes depict output data and large black rectangles show analysis tools.

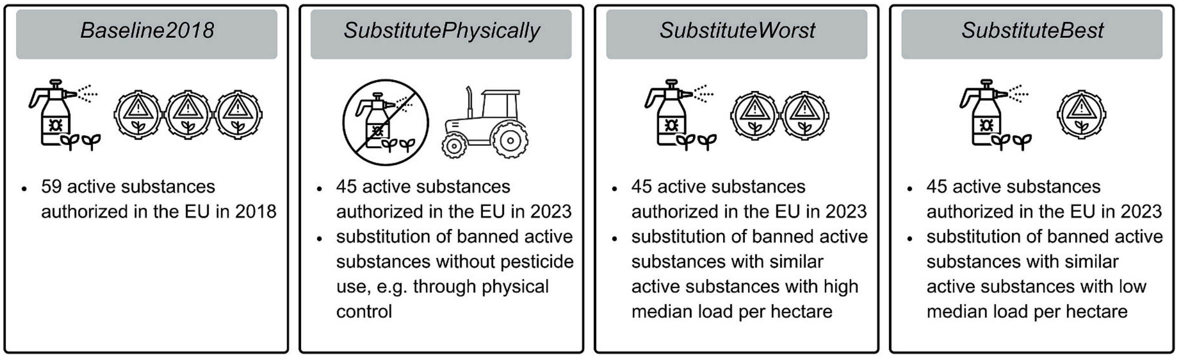

Fig. 2. Schematic of four pesticide use scenarios, where the first schematic depicts the baseline scenario *Baseline2018* and the other three scenarios comprise the ban-substitution scenarios.

with the average national FAO data for all 28 crops and pastures (FAO, 2023b). The CORINE land cover 2018 data (Copernicus Land Monitoring Service, EEA, 2021) constrains available cropland and pasture areas at the HRU scale. Another constraint ensures absolute active substance usage in each country matches reported values for aggregated pesticide classes from FAO. Based on our simulated cropland and pasture area, we calculate pesticide load with the recalculated pesticide application rates from Pest-CHEMGRIDS (Maggi et al., 2019) and the Pesticide Load method. Appendix C in the supplementary data describes the mathematical framework of the optimization problem in more detail.

3. Results

3.1. Distribution patterns of pesticide load

The total annual use of active substances in the *Baseline2018* scenario is 299,000 tons, resulting in a total pesticide load of 3.82 billion L. The insecticide chlorpyrifos is the active substance with the highest pesticide load of 3093 million L per year, followed by dimethoate and calcium polysulfide with 381 and 45 million L per year, respectively. Calcium polysulfide and chlorpyrifos have the highest pesticide load per hectare with 5.77 L ha^{-1} and 4.71 L ha^{-1} (Table 3). Several insecticides with a high pesticide load per hectare have been banned since 2018, including chlorpyrifos, dimethoate, ethoprophos and indoxacarb. Treated cropland areas with a high pesticide load are relatively small, amounting for all but three active substances to less than 10 million ha. Exceptions are chlorpyrifos (63 million ha), chlorothalonil (42 million ha) and dimethoate (19 million ha). Glyphosate is applied on 123 million ha but has a median pesticide load per hectare of 0.19 L ha^{-1} (Fig. 3).

In the *Baseline2018* scenario, chlorpyrifos is the active substance with the highest median load per hectare for 20 out of 28 crops, including winter wheat, summer barley, maize, rapeseed, olives, sunflower seed, durum wheat, grapes, triticale, oats, maize silage, alfalfa, winter rye, sugar beet, mixed grain, soybean, almond, apples, flax and cotton (Fig. 4, Appendix D in the supplementary data). The pesticide load per hectare for chlorpyrifos ranges from 0.65 L ha^{-1} on cotton to 56.26 L ha^{-1} on olives. The crops with the highest median load over all active substances are olives (0.43 L ha^{-1}), almonds (0.38 L ha^{-1}), grapes (0.30 L ha^{-1}), apples (0.30 L ha^{-1}) and dry beans (0.29 L ha^{-1}). Several active substances with high pesticide load per hectare on individual crops have been banned since 2018, including chlorpyrifos, dimethoate and chlorothalonil. Calcium polysulfide has a load of around 5.7 L ha^{-1} on olives, almonds, grapes and apples and is currently approved in the EU.

In the *Baseline2018* scenario, regions with the highest median pesticide load per hectare are Cyprus (0.81 L ha^{-1}), Flevoland (0.65 L ha^{-1}), Groningen (0.49 L ha^{-1}), Drenthe (0.43 L ha^{-1}) and Zeeland (0.42 L ha^{-1}) in the Netherlands. The median pesticide load in both countries is especially high for cultivating vegetables (1.18 and 1.79 L ha^{-1}, respectively). Other regions with high pesticide load per hectare are in Belgium, Italy and France. The particularly high pesticide load in Cyprus and the Netherlands is almost exclusively due to chlorpyrifos, which has a very high pesticide load and high application rates. The average application rate of chlorpyrifos is 0.74 kg ha^{-1} in Cyprus, 1.11 kg ha^{-1} in the Netherlands and 0.13 in the EU26 + 1. An active substance is applied on average at application rates of 3.4 kg ha^{-1} in Cyprus, ca. 1 kg ha^{-1} in the Netherlands and 0.48 kg ha^{-1} in the EU26 + 1.

3.2. Pesticide use scenarios

In 2018, the annual active substance usage equaled 299,000 tons and a pesticide load of 3825 million L (shown in the *Baseline2018* scenario in Fig. 5). Substitution of banned active substances without additional pesticide use in the *SubstitutePhysically* scenario results in 217,000 tons of pesticide use (27% decrease) and a pesticide load of 242 million L (94% decrease). If banned active substances are substituted with other active substances, the pesticide load amounts to 323 and 298 million L (*SubstituteWorst* and *SubstituteBest,* respectively). The ban on 14 active substances decreases the pesticide load more strongly in Middle and Eastern Europe than in Western and Southern Europe (Fig. 6). In the *SubstitutePhysically* scenario, the pesticide load takes the value 0 on the HRU scale more often than the SubstituteWorst or SubstituteBest scenarios, leading to a higher median load per hectare and darker colors in Fig. 6. Cyprus is the region with the highest pesticide load before Flevoland and Groningen with 0.57, 0.42 and 0.42 L ha^{-1} in the three ban-substitution scenarios, respectively. Ban-substitution effects could increase the pesticide load for the *SubstituteWorst* scenario only in Denmark from 2.57 to 2.88 million L due to the substitution of pendimethalin (median load of 0.32 L ha^{-1} in Denmark) for bromoxynil (0.15 L ha^{-1}). The most substantial relative decrease in pesticide load occurs in Greece with a reduction from 328.00 million L in the *Baseline2018* scenario to 4.72 million L in the *SubstituteBest* scenario, which stems primarily from the ban of chlorpyrifos (Fig. 5). Based on California Department of Pesticide Regulation (2023), we substitute bacillus amyloliquefaciens for chlorpyrifos, for example in Greece on winter wheat. However, because bacillus amyloliquefaciens was not modeled for wheat in PEST-CHEMGRIDS, we substitute the pesticide load per hectare of chlorpyrifos on winter wheat in Greece (11.72 L ha^{-1}) with the median pesticide load per hectare of bacillus amyloliquefaciens on other cereals in the EU26 + 1 (0.005 L ha^{-1}).

Our analysis shows that the chlorpyrifos and dimethoate account for a large proportion of the high pesticide load in 2018 (shown in the *Baseline2018* scenario). This conceals the effects of ban-substitutions of other substances. Therefore, we also investigate the change in total pesticide load excluding chlorpyrifos and dimethoate in the four pesticide use scenarios. Here, the total pesticide load for the SubstituteWorst scenario is higher than that of the *Baseline2018* scenario for 14 countries (Fig. 7). The substitution of glufosinate (0.01 L ha^{-1}) and diuron (0.06 L

Table 2
List of active substances that have been banned in the EU since 2018, their pesticide category and possible active substance substitutes for the crop groups given by PEST-CHEMGRIDS (Maggi et al., 2019).

Active substance	Category	Active substance substitutes	Reference
Bromoxynil	Herbicide	pendimethalin (alfalfa, other crops), dimethenamid(-p) (other crops), MCPA (wheat), clopyralid (wheat), 2,4-D (wheat), dicamba (wheat)	EFSA (2018)
Chlorothalonil	Fungicide/ Bactericide	captan (orchards/ grapes), ziram (orchards/ grapes), azoxystrobin (orchards/grapes, vegetables/fruits, other crops), pyraclostrobin (other crops), metoconazole (vegetables/fruits)	Jacometti et al. (2010), Michigan State University Extension (2023a, b), European Crop Protection (2016)
Chlorpyrifos	Insecticide	bacillus amyloliquefaciens (alfalfa, maize, cotton, orchards/grapes, soybean, wheat, other crops)	California Department of Pesticide Regulation (2023)
Clothianidin	Insecticide	Bacillus amyloliquefaciens (rice)	Jactel et al. (2019), European Crop Protection (2016)
Dimethoade	Insecticide	cyhalothrin-lambda (alfalfa, wheat)	Manitoba Pulse and Soybean Growers (2023), European Crop Protection (2016)
Diuron	Herbicide	metribuzin (alfalfa, orchards/grapes, pasture/hay), glyphosate (alfalfa, cotton, pasture/ hay), fluometuron (cotton)	Mississippi State University Extension (2023), Horticulture Australia Ltd (2023)
Ethoprophos	Insecticide	metam potassium (vegetables/fruits), axozystrobin (vegetables/fruits)	University of Georgia Extension (2018), (European Crop Protection, 2016)
Glufosinate	Herbicide	glyphosate (maize, cotton, orchards/grapes, pasture/hay, soybean), flumioxazine (orchards/ grapes), fluroxypyr (orchards/graps), pendimethalin (orchards/grapes) clopyralid (orchards/ graps), picloram (orchards/grapes), dimethamid(-p) (orchards/grapes), metribuzin (soybean), metolachlor(-s) (soybean)	North Carolina State University Extension (2018) European Crop Protection (2016)
Imidacloprid	Insecticide	cyhalothrin-lambda (cotton)	Furlan and Kreutzweiser (2015)
Indoxacarb	Insecticide	cyhalothrin-lambda (alfalfa)	Furlan and Kreutzweiser (2015)
Mancozeb	Fungicide/ Bactericide	coppyer hydroxide (orchards/grapes), azoxystrobin (orchards/ grapes, vegetables/ fruits), tebuconazole (orchards/grapes, vegetables/fruits), prothioconazole (orchards/grapes), captan (orchards/grapes, vegetables/fruits), metconazole (orchards/ grapes), metiram (vegetables/fruits)	PAN Europe (2020); Vinpro (2021), European Crop Protection (2016)
Phosmet	Insecticide	malathion (alfalfa), cyhalothrin-lambda (alfalfa), bacillus amyloliquefaciens (alfalfa)	Townsend et al. (1979) (European Crop Protection, 2016)
Propiconazole	Fungicide/ Bactericide	azoxystrobin (rice), prothioconazole (wheat), tebuconazole (rice, wheat), metconazole (wheat)	Uppala and Zhou (2018); Battel (2020), University of Tennessee Extension (2023), European Crop Protection (2016)
Thiophanate-methyl	Fungicide/ Bactericide	prothioconazole (vegetables/fruits, wheat), tebuconazole (vegetables/fruits, wheat), metconazole (wheat), azoxystrobin (wheat)	Petkar et al. (2017), European Crop Protection (2016)

Table 3
List of the 10 active substances with the highest total annual pesticide load per year and pesticide per hectare per year in the *Baseline2018* scenario and their approval status.

Active substance	Pesticide load (million L)	Banned	Active substance	Pesticide load (L ha^{-1})	Banned
Chlorpyrifos	3093	yes	Calcium polysulfide	5.77	
Dimethoate	381	yes	Chlorpyrifos	4.71	yes
Calcium polysulfide	45		Metam potassium	2.75	
Glyphosate	31		Malathion	1.03	
Chlorothalonil	27	yes	Dimethoate	0.70	yes
Flutolanil	23		Cyhalothrin-lambda	0.52	
Indoxacarb	20	yes	Imidacloprid	0.48	yes
Pyraclostrobin	19		Mancozeb	0.45	yes
Pendimethalin	17		Ziram	0.41	
Metolachlor (-s)	15		Ethoprophos	0.35	yes

ha^{-1}) with glyphosate (0.19 L ha^{-1}) increases the total pesticide load in all countries of the EU26 + 1.

4. Discussion

This study is the first to quantify the risk of pesticide use on non-target organisms and the environment using the risk indicator Pesticide Load for the EU26 + 1. We map the pesticide load per hectare spatially and on individual crops. Results indicate that the highest pesticide load per hectare occurs in the Netherlands due to high application rates of active substances and a high ratio of vegetable production. Chlorpyrifos caused the highest pesticide load per hectare on more than half of the assessed crops before its ban. Orchard fruit and vegetables have the highest pesticide load per hectare. Denmark's pesticide load per hectare is between 0.03 and 0.12 L ha^{-1} in the *Baseline2018* scenario, which is an order of magnitude lower than reported in Kudsk et al. (2018). This suggests we substantially underestimate the accurate pesticide load, which could stem from various limitations (see further

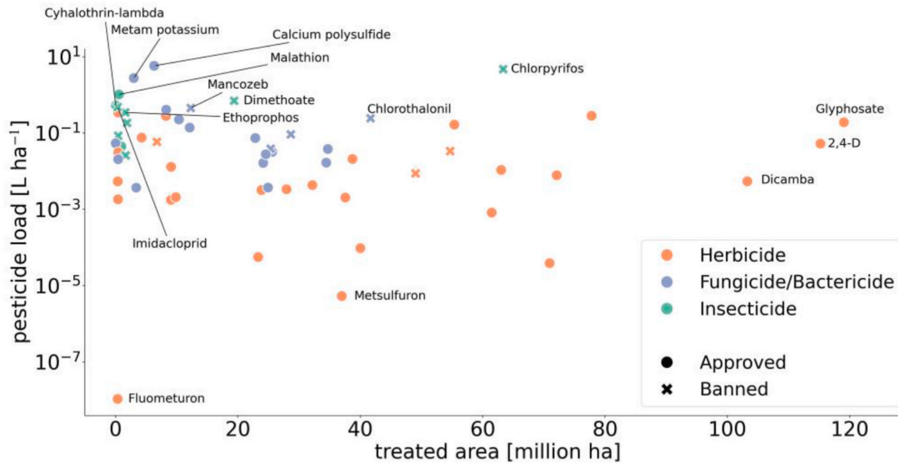

Fig. 3. Area of application vs. pesticide load per hectare for all 59 active substances in the *Baseline2018* scenario. Colors indicate the pesticide category. Approved active substances in 2023 are marked with a circle; banned active substances are marked with a cross. Names are annotated for the six banned active substances with the highest pesticide load and approved active substances with high/low pesticide loads or large treated areas. (For interpretation of the references to color in this figure legend, the reader is referred to the Web version of this article.)

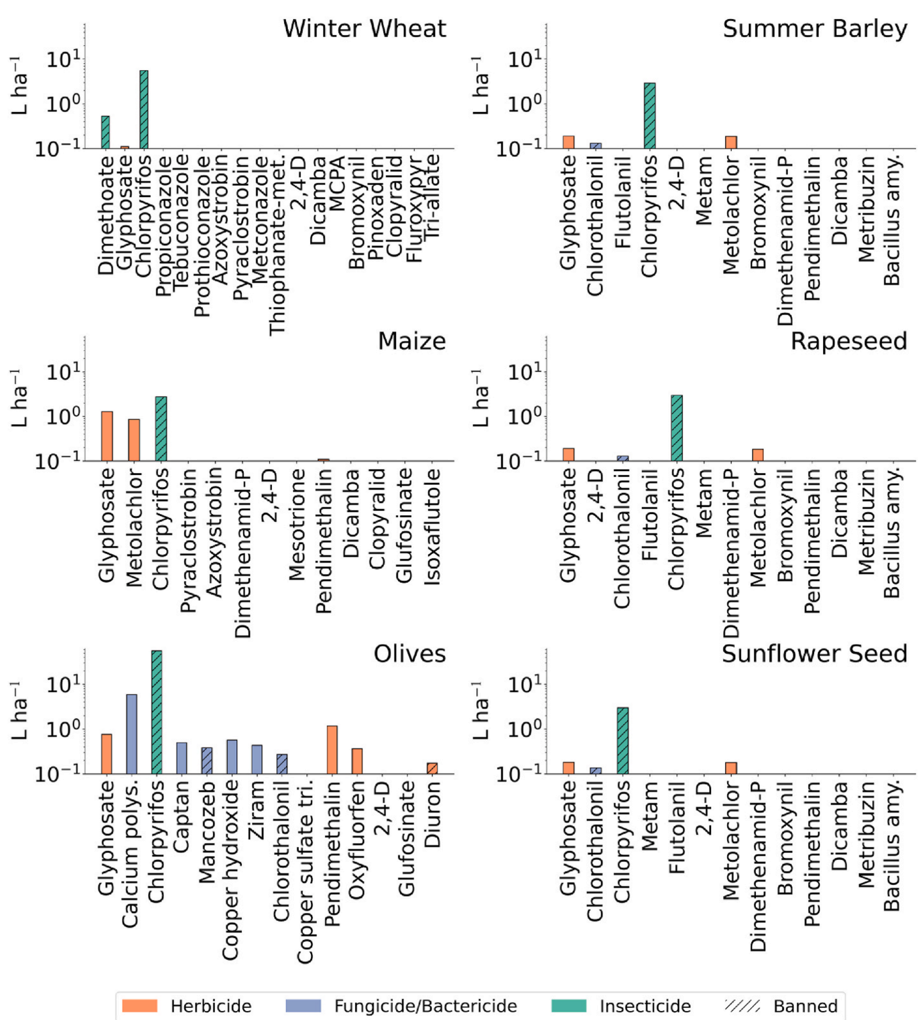

Fig. 4. Median pesticide load per hectare per active substance in the *Baseline2018* scenario over all regions in the EU26 + 1 for the six crops cultivated in the largest area. Colors indicate the classification of an active substance as herbicide, fungicide/bactericide or insecticide. Hatched bars represent active substances that have been banned in the EU between 2018 and 2023. The active substances are listed according to their total application amount in tons in descending order for each crop. (For interpretation of the references to color in this figure legend, the reader is referred to the Web version of this article.)

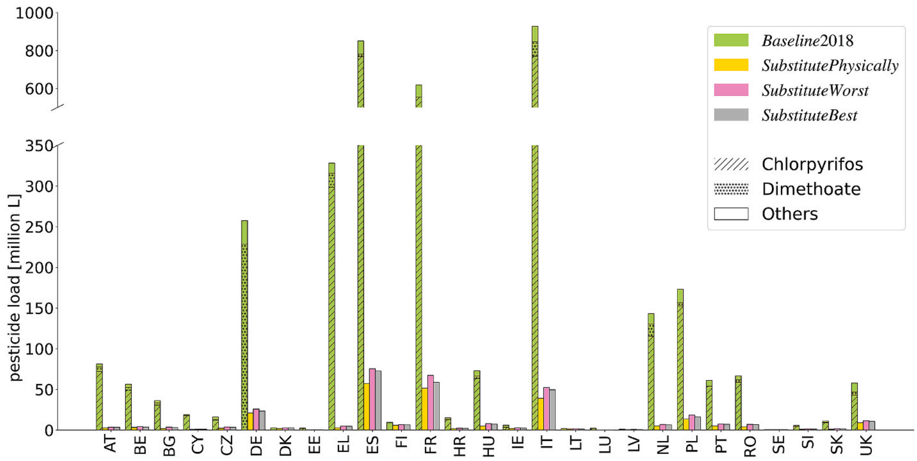

Fig. 5. Total pesticide load per country in million L for four pesticide use scenarios. The hatching indicates the pesticide load of the active substances chlorpyrifos (diagonal lines) and dimethoate (circles).

below). We also investigate ban-substitution effects of banning 14 active substances between 2018 and 2023. We substitute the banned active substances either without further pesticide use, e.g. with increased manual labor, or with active substance substitutes. Our results confirm a substantial decrease in pesticide load following the ban on 14 pesticides in the EU26 + 1. The substitution without further pesticide use (*SubstitutePhysically* scenario) reduces pesticide load by 94%. In the *SubstituteWorst* and *SubstituteBest* scenarios, pesticide load is reduced by 91.6% and 92%, respectively. The ban on chlorpyrifos and dimethoate significantly reduced the pesticide load. However, our results also show that ban-substitution effects can increase pesticide load in some cases, for example, by substituting glufosinate with glyphosate. While both glyphosate and glufosinate have similar pesticide load values per kg (0.44 L kg^{-1} and 0.71 L kg^{-1} and), glyphosate is used with much higher application rates (0.78 kg ha^{-1}) compared to glufosinate (0.04 kg ha^{-1}) leading to a higher median pesticide load per hectare (0.19 L ha^{-1} and 0.01 L ha^{-1}).

Although bans on active substances are justified to control certain endpoint risks, our research indicates that bans can also lead to risk shifts. Other studies have shown that active substance bans could lead to risk shifts. Gray and Hammitt (2000) investigated a potential ban on organophosphate and carbamate pesticides in U.S. agriculture and its effects on risk. They considered changes in diet, mortality due to loss of income and effects of pesticide substitutes for acute toxicity, cancer and non-cancer risks on farm workers. They concluded that countervailing risks could offset risk reductions through a ban but could not quantify effects due to data gaps and shortcomings in risk assessment methods. The introduction of neonicotinoids in U.S. maize significantly decreased the probability of organophosphate and pyrethroid insecticide use and the risks of pesticide use (Perry & Moschini, 2020). The authors adopted the risk quotient method, where the ratio of exposure (quantity applied) to toxicity (acute LD_{50}) is calculated for individual insecticides on various non-target organisms. The hazard quotient is then the sum of the insecticide risk quotients. They found that the ban on neonicotinoid seed treatments would increase the hazard quotient significantly on rats, birds and fish but decrease the hazard quotient on bees.

Other studies examined the effects of neonicotinoid restrictions in European agriculture in 2013 but did not quantify risk changes. Kathage et al. (2017) found that farmers switched to using other neonicotinoids or pyrethroids or adapted pest management practices such as a higher sowing density and more pest scouting. Farmers perceived these practices as more time- and cost-intensive. Scott & Bilsborrow (2019) concluded that crop losses due to neonicotinoid restrictions amounted to 3–5% and pesticide applications increased for seed dressings without neonicotinoid treatment. Further investigations concluded that bans can decrease available active substances and modes of action, making resistance management more complex or can lead to production shifts abroad (Zilberman et al., 1991; Moss et al., 2019; Kudsk and Mathiassen, 2020).

This has several regulatory implications. Policymakers should be aware that active substance bans would likely lead to substitutions with other available active substances. With this, the risks of pesticide use on non-target organisms and the environment could stay constant or increase, while options to counteract resistance would decrease. Bans could also lead to yield losses, especially when they are substituted solemnly by physical methods (Meemken & Qaim, 2018). Therefore, cost-benefit analyses of bans should carefully weigh the benefits and possible trade-offs through ban-substitution effects. Generally, Carvalho (2017) and Siviter and Muth (2020) have criticized the current way of development and approval of active substances as going in circles with identifying a pest, developing an active substance, observing secondary effects such as resistances or environmental damages, banning the active substance and developing a new active substance. Instead, new regulatory processes for approving active substances should be established, based more strongly on the precautionary principle. Topping et al. (2020) have illustrated that the environmental risk assessment under the EC Regulation 1107/2009 is outdated and does not account for stressors such as climate change, habitat destruction and landscape homogeneity. Once active substances are approved, they are placed on the market for ten years regardless of their use scale (Frische et al., 2018). In addition, effective policies should consider farmers' heterogeneous behavior, as the magnitude of ban-substitution effects depends mainly on strategies that farmers choose to replace the banned active substances (Böcker et al., 2018; Finger et al., 2023). We tried to account for this issue by examining three ban-substitution scenarios, but uncertainties are still high. This could be improved by using a modeling approach of landscape and farming practices that include the central role of farmers as well as mixing dynamics of pesticide groups (Topping et al., 2020). To reduce pesticide use and risk, a second pillar besides active substance bans and possible substitutions could be implementing a pesticide tax. Böcker and Finger (2016), Finger et al. (2017) and Nielsen et al. (2023) have shown that well-designed pesticide taxation systems, such as the Danish pesticide tax based on the Pesticide Load method, can result in a significant reduction of pesticide risk.

The Pesticide Load method, however, has some shortcomings. If data is missing, the load on the input parameter is omitted, which can lead to an underestimation of the load. For example, eight input parameters are missing for bacillus amyloliquefaciens, leading to an underestimation of the ban-substitution effect for chlorpyrifos. Imidacloprid, clothianidin, indoxacarb, and ethoprophos were previously banned in the EU because

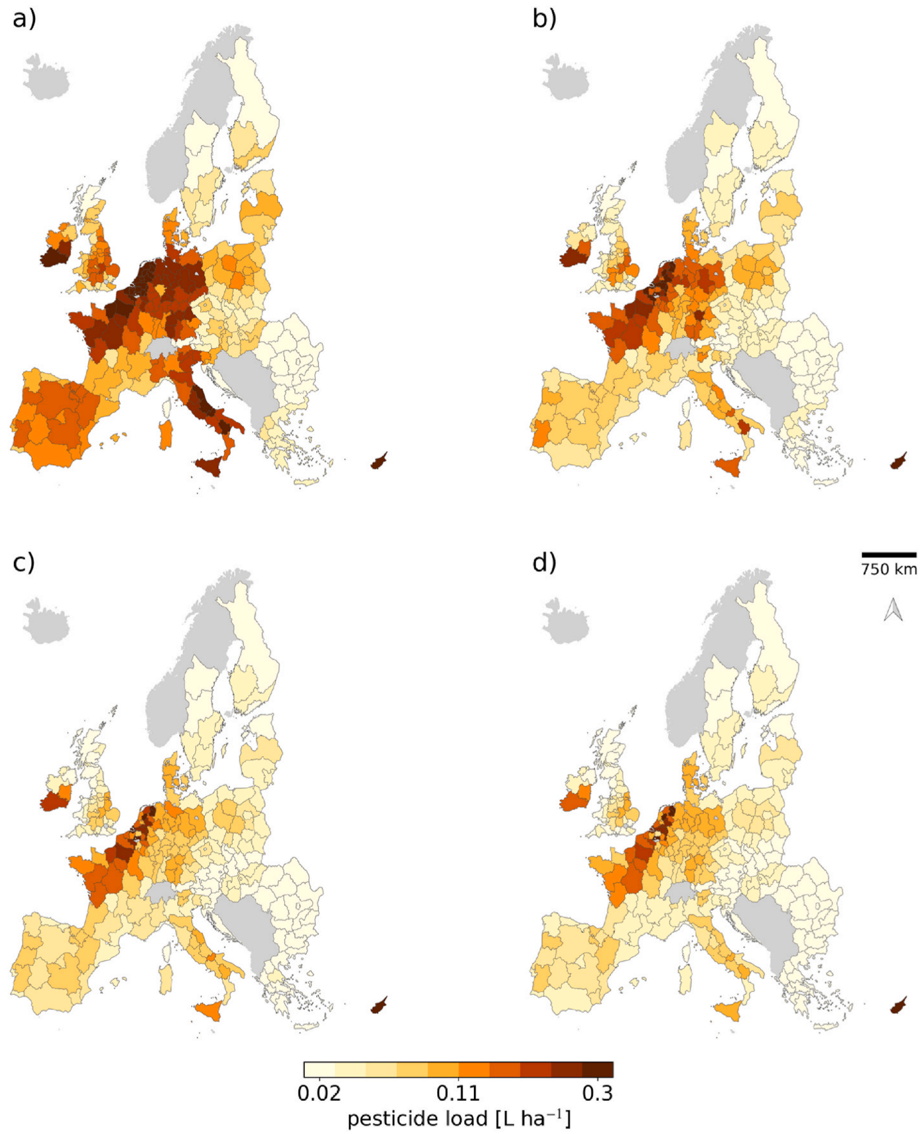

Fig. 6. Annual median pesticide load per hectare in L ha^{-1} for the scenarios *Baseline2018* (a), *SubstitutePhysically* (b), *SubstituteWorst* (c) and *SubstituteBest* (d) for the EU26 + 1. Regions highlighted in grey were not considered. The color code represents the distribution of calculated values of the *Baseline2018* scenario (a) in 10% percentiles, where light beige represents the 10% lowest values and dark red represents the 10% highest values. The eight color codes in between linearly represent the 10% percentiles in between the lowest and highest values. The labels of the color bar represent the median of the lowest percentile, the median of the whole value range and the median of the highest percentile. The color code in (b)–(d) follows the color code of (a) to make a comparison across scenarios visible. In the *SubstitutePhysically* scenario, the pesticide load takes the value 0 more often than the *SubstituteWorst* or *SubstituteBest* scenarios, leading to a higher median load per hectare and darker colors. (For interpretation of the references to color in this figure legend, the reader is referred to the Web version of this article.)

of unacceptably high risks to bees, beneficial arthropods, birds, and small mammals, or because such risks could not be excluded (e.g. European Commission (2019c, 2021)). Chlorpyrifos, chlorothalonil, dimethoate, phosmet, propiconazole and mancozeb were banned because of unacceptable risks to human health (e.g. European Commission (2019b,d, 2020b,c)). Not all of these circumstances are reflected in the pesticide load values. For example, chlorpyrifos only has a human health load of 0.166 for "H301: Toxic if swallowed", but the European Food Safety Authority reported that a genotoxic potential of chlorpyrifos could not be ruled out (EFSA, 2019). On the other hand, chlorpyrifos has high acute toxicity levels for daphnia and bees, which are weighted very strongly in the total pesticide load calculations. This weighting of the input parameters is somewhat arbitrary. We use the original weighting, which gives higher importance to bees and pollinators, aquatic organisms and leaching. Hence, active substances with high acute toxicity for fish, daphnia and bees, including bifenthrin, cyhalothrin-lambda, tebuconazole and clothianidin, have a very high total pesticide load. In addition, the Pesticide Load captures a generic risk exposure, but actual risk exposure is determined by environmental factors such as weather, soil and hydrology and how farmers handle active substances and protect themselves (Feola et al., 2011). A modeling approach may better determine pesticide leakage and risk, depending on the cultivation method, soil, and climate. However, Mankong et al. (2022) compared different modeling approaches for estimating ecosystem impacts of active substances in Thailand and the results differ substantially.

Pesticide load mapping helps contextualize and compare different pesticide usage scenarios, as we did in this study. However, our results are restricted by several assumptions and limitations. Primarily, detailed pesticide application data for individual crop-active substance combinations are lacking. In PEST-CHEMGRIDS, only ten aggregated crop classes are represented, but active substances are often applied on specific crops and not necessarily on whole crop classes. Another issue is the correctness of the modeled active substance application rates. For example, the average application rate of glufosinate in PEST-

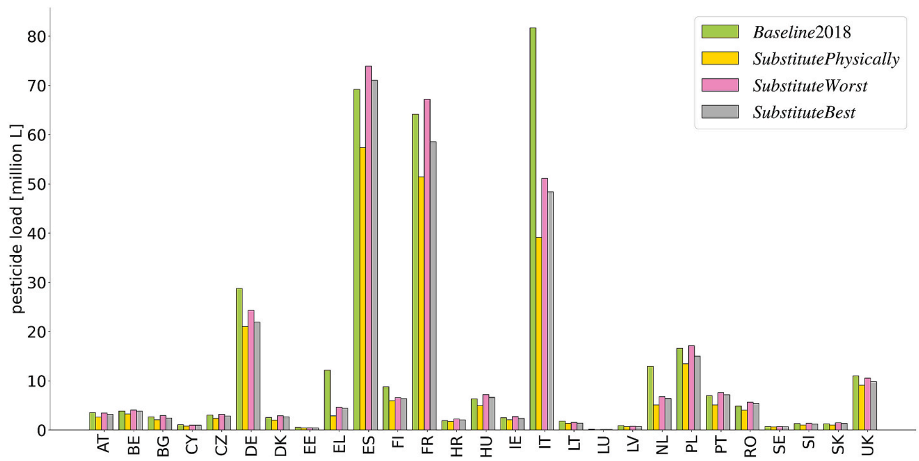

Fig. 7. Total pesticide load per country in million L excluding the active substances chlorpyrifos and dimethoate for four pesticide use scenarios.

CHEMGRIDS for the EU26 + 1 is 0.04 kg ha^{-1}, but application rates of up to 1.5 kg ha^{-1} are authorized in the EU (EFSA, 2015). Antier et al. (2020) reported average application rates of 0.24 kg ha^{-1} for glyphosate in Europe. PEST-CHEMGRIDS modeled an average of 0.78 kg ha^{-1}, which is within the same order of magnitude but overestimates application rates by a factor of three. The 299,000 tons of active substance use we modeled in the agricultural sector analysis correspond to 94% of the 319,000 tons reported active substance use by FAO for the three categories herbicides, fungicides/bactericides and insecticides, averaged over 2011 to 2020. However, aggregated FAO data on active substance use is a collection of national inventories that differ strongly in their accounting methods (FAO, 2022b). There is also evidence that formulations can exhibit increased toxicity compared to single active substances (Richard et al., 2005), suggesting the need for more detailed reporting of pesticide use to improve pesticide usage and load estimates. All these limitations could lead to a miscalculation of the accurate pesticide load.

Our substitution scenarios are only rudimentary because active substance substitutes are not always available from PEST-CHEMGRIDS. Due to missing data, we also did not consider newly developed active substances or emergency authorizations for specific active substances. Even though the ban-substitution scenarios we use are based on studies and recommendations for farmers, it is not clear if and how active substances are substituted. The ban-substitution scenarios, therefore, only give an idea of the magnitude of the impact.

5. Conclusion

In this study, we mapped and quantified the risk of pesticide use in the EU26 + 1 with the Pesticide Load method. Results give an idea of the distribution pattern and magnitude of the pesticide load of 59 active substances on 28 crops and pastures. For 2018, we report the highest pesticide load per hectare in Cyprus and the Netherlands due to high pesticide application rates and a high proportion of vegetable production. Chlorpyrifos caused the highest pesticide load per hectare on more than half of the crops modeled. Orchard fruits and vegetables show the largest pesticide load per hectare. However, more detailed pesticide application data for individual crop-active substance combinations are needed. It is, therefore, crucial to improve the reporting of detailed pesticide use in the EU and make it freely available to the scientific community. The EU maintains one of the most stringent frameworks for pesticide regulation, demonstrating a determined path toward reducing pesticide effects on non-target organisms and the environment. However, our results show that the ban of selected active substances between 2018 and 2023 could lead to substitutions that possibly offset pesticide load reductions. Therefore, regulatory procedures should be improved to enhance the approval mechanisms for active substances and consider the effects of ban-substitution following a ban on active substances. Our work contributes to the ongoing scientific and societal discourse on efficiently mitigating the impacts of pesticides on non-target organisms and the environment.

CRediT authorship contribution statement

Luisa Gensch: Conceptualization, Data curation, Formal analysis, Investigation, Methodology, Visualization, Writing – original draft. **Kerstin Jantke:** Supervision, Writing – review & editing. **Livia Rasche:** Resources, Writing – review & editing. **Uwe A. Schneider:** Conceptualization, Methodology, Supervision, Writing – review & editing.

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.

Data availability

Data will be made available on request.

6. Acknowledgments

We thank the anonymous reviewers for useful comments on the manuscript. This work is funded by the Deutsche Forschungsgemeinschaft (DFG, German Research Foundation) under Germany's Excellence Strategy—EXC 2037 'CLICCS—Climate, Climatic Change, and Society'—Project Number: 390683824, contribution to the Center for Earth System Research and Sustainability (CEN) of Universität Hamburg. This study further benefited from financial support for the Project AXIS-073 NorthWesternPaths, reference 01LS1910A, funded by the German Federal Ministry of Education and Research (BMBF).

Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.envpol.2024.123836.

References

Anon, 2012. The Agricultural Pesticide Load in denmark 2007–2010. Miljøstyrelsen: København, Denmark.

Antier, C., Kudsk, P., Reboud, X., Ulber, L., Baret, P.V., Messéan, A., 2020. Glyphosate use in the European agricultural sector and a framework for its further monitoring. Sustainability 12 (14), 5682.
Battel, R., 2020. Fungicide options to combat Fusarium head blight. URL: https://www.canr.msu.edu/news/fungicide-options-to-combat-fusarium-head-blight. (Accessed 22 September 2023).
Beaumelle, L., Tison, L., Eisenhauer, N., Hines, J., Malladi, S., Pelosi, C., Thouvenot, L., Phillips, H.R., 2023. Pesticide effects on soil fauna com- munities—a meta-analysis. J. Appl. Ecol. 60.
Böcker, T., Finger, R., 2016. European pesticide tax schemes in comparison: an analysis of experiences and developments. Sustainability 8, 378.
Böcker, T., Britz, W., Finger, R., 2018. Modelling the effects of a glyphosate ban on weed management in silage maize production. Ecol. Econ. 145, 182–193.
Bub, S., Wolfram, J., Petschick, L.L., Stehle, S., Schulz, R., 2022. Trends of total applied pesticide toxicity in German agriculture. Environ. Sci. Technol. 57, 852–861.
California Department of Pesticide Regulation, 2023. Chlorpyrifos action plan. URL: https://www.cdpr.ca.gov/docs/chlorpyrifos/pdf/chlorpyrifos_action_plan.pdf, 2023-09-22).
Carvalho, F.P., 2017. Pesticides, Environment, and Food Safety, vol. 6. Food and Energy Security, pp. 48–60.
Copernicus Land Monitoring Service, EEA, 2021. CORINE land cover - user manual. URL: https://land.copernicus.eu/en/technical-library/clc-product-user-manual/@@download/file. (Accessed 16 October 2023).
Dushoff, J., Caldwell, B., Mohler, C.L., 1994. Evaluating the environmental effect of pesticides: a critique of the environmental impact quotient. Am. Entomol. 40, 180–184.
EFSA, 2015. Reasoned opinion on the review of the existing maximum residue levels (MRLs) for for glufosinate according to Article 12 of Regulation (EC) No 396/2005. EFSA J. 13 (1), 3950.
EFSA, 2018. Evaluation of data concerning the necessity of bromoxynil as herbicide to control a serious danger to plant health which cannot be contained by other available means, including non-chemical methods. EFSA J. 16, e05391.
EFSA, 2019. Statement on the available outcomes of the human health assessment in the context of the pesticides peer review of the active substance chlorpyrifos. EFSA J. 17, e05809.
European Commission, 2019a. Commission Directive (EU) 2019/782 of 15 May 2019 amending Directive 2009/128/EC of the European Parliament and of the Council as regards the establishment of harmonised risk indicators (Text with EEA relevance.). URL: http://data.europa.eu/eli/dir/2019/782/oj, 2024-02-07).
European Commission, 2019b. Commission Implementing Regulation (EU) 2019/1090 of 26 June 2019 concerning the non-renewal of approval of the active substance dimethoate. In: Accordance with Regulation (EC) No 1107/2009 of the European Parliament and of the Council Concerning the Placing of Plant Protection Products on the Market, and Amending the Annex to Commission Implementing Regulation (EU) No 540/2011 (Text with EEA relevance). URL: http://data.europa.eu/eli/reg_impl/2019/1090/oj. (Accessed 9 October 2023).
European Commission, 2019c. Commission Implementing Regulation (EU) 2019/344 of 28 February 2019 concerning the non-renewal of approval of the active substance ethoprophos. In: Accordance with Regulation (EC) No 1107/2009 of the European Parliament and of the Council Concerning the Placing of Plant Protection Products on the Market, and Amending the Annex to Commission Implementing Regulation (EU) No 540/2011 (Text with EEA relevance). URL: http://data.europa.eu/eli/reg_impl/2019/344/oj. (Accessed 9 October 2023).
European Commission, 2019d. Commission Implementing Regulation (EU) 2019/677 of 29 April 2019 concerning the non-renewal of the approval of the active substance chlorothalonil. In: Accordance with Regulation (EC) No 1107/2009 of the European Parliament and of the Council Concerning the Placing of Plant Protection Products on the Market, and Amending Commission Implementing Regulation (EU). No 540/2011 (Text with EEA relevance). URL: http://data.europa.eu/eli/reg_impl/2019/677/oj. (Accessed 9 October 2023).
European Commission, 2020a. Commission Implementing Regulation (EU) 2020/1276 of 11 September 2020 concerning the non-renewal of the approval of the active substance bromoxynil. In: Accordance with Regula- Tion (EC) No 1107/2009 of the European Parliament and of the Council Concerning the Placing of Plant Protection Products on the Market, and Amending the Annex to Commission Implementing Regulation (EU) No 540/2011 (Text with EEA relevance). URL: http://data.europa.eu/eli/reg_impl/2020/1276/oj. (Accessed 2 August 2023).
European Commission, 2020b. Commission Implementing Regulation (EU) 2020/18 of 10 January 2020 concerning the non-renewal of the approval of the active substance chlorpyrifos. In: Accordance with Regulation (EC) No 1107/2009 of the European Parliament and of the Council Concerning the Placing of Plant Protection Products on the Market, and Amending the Annex to Commission Implementing Regulation (EU) No 540/2011 (Text with EEA relevance). URL: http://data.europa.eu/eli/reg_impl/2020/18/oj. (Accessed 2 August 2023).
European Commission, 2020c. Commission Implementing Regulation (EU) 2020/2087 of 14 December 2020 concerning the non-renewal of the approval of the active substance mancozeb. In: Accordance with Regulation (EC) No 1107/2009 of the European Parliament and of the Council Concerning the Placing of Plant Protection Products on the Market, and Amending the Annex to Commission Implementing Regulation (EU) No 540/2011 (Text with EEA relevance). URL: http://data.europa.eu/eli/reg_impl/2020/2087/oj. (Accessed 9 October 2023).
European Commission, 2021. Commission Implementing Regulation (EU) 2021/2081 of 26 November 2021 concerning the non-renewal of approval of the active substance indoxacarb. In: Accordance with Regulation (EC) No 1107/2009 of the European Parliament and of the Council Concerning the Placing of Plant Protection Products on the Market, and Amending Commission Implementing Regulation (EU). No 540/2011 (Text with EEA relevance). URL: http://data.europa.eu/eli/reg_impl/2021/2081/oj. (Accessed 9 October 2023).
European Commission, 2023. Pesticides Database - active substances, safeners and synergists. URL: https://ec.europa.eu/food/plant/pesticides/eu-pesticides-database/start/screen/active-substances. (Accessed 26 June 2023).
European Crop Protection, 2016. Low Yield. Cumulative impact of hazard- based legislation on crop protection products in Europe. Final Report. pp. 91–115.
European Environment Agency, 2023. How pesticides impact human health and ecosystems in Europe. EEA Briefings 06 ee a.europa.eu/ds_resolveuid/01773180470148d39810a205f20d04c5. (Accessed 7 February 2024).
European Parlimanet and Council, 2009. REGULATION (EC) No 1107/2009 OF THE EUROPEAN PARLIAMENT AND OF TH COUNCIL of 21 October 2009 concerning the placing of plant protection products on the market and repealing Council Directives 79/117/EEC and 91/414/EEC URL. http://data.europa.eu/eli/reg/2009/1107/oj. (Accessed 13 February 2024).
Eurostat, 2023. Annual Crop Statistics. Handbook, 2023 Edition. European Commission: Statistical Books, p. 200.
FAO, 2022a. Pesticides trade and pesticides indicators – global, regional and country trends, 1990–2020. FAOSTAT Analytical Briefs 46.
FAO, 2022b. World food and agriculture – statistical yearbook 2022. FAOSTAT, 2022a. FAOSTAT pesticides use - country notes, june 2022. URL: https://fenixservices.fao.org/faostat/static/documents/RP/RP_e_Country_Notes.pdf. (Accessed 7 August 2023).
FAO, 2023a. Pesticide Use. FAOSTAT, Rome. http://www.fao.org/faostat/en/#data/RP. (Accessed 7 August 2023).
FAO, 2023b. Production: Crops and Livestock Products. FAOSTAT, Rome. URL: https://www.fao.org/faostat/en/#data/QCL. (Accessed 7 August 2023).
Feola, G., Rahn, E., Binder, C.R., 2011. Suitability of pesticide risk indicators for less developed countries: a comparison. Agric. Ecosyst. Environ. 142, 238–245.
Finger, R., Möhring, N., Dalhaus, T., Böcker, T., 2017. Revisiting pesticide taxation schemes. Ecol. Econ. 134, 263–266.
Finger, R., Möhring, N., Kudsk, P., 2023. Glyphosate ban will have economic impacts on European agriculture but effects are heterogeneous and uncertain. Communications Earth & Environment 4, 286.
Frische, T., Egerer, S., Matezki, S., Pickl, C., Wogram, J., 2018. 5-point programme for sustainable plant protection. Environ. Sci. Eur. 30, 1–17.
Furlan, L., Kreutzweiser, D., 2015. Alternatives to neonicotinoid insecticides for pest control: case studies in agriculture and forestry. Environ. Sci. Pollut. Control Ser. 22, 135–147.
Gray, G.M., Hammitt, J.K., 2000. Risk/risk trade-offs in pesticide regulation: an exploratory analysis of the public health effects of a ban on organophosphate and carbamate pesticides. Risk Anal. 20, 665–680.
Gutsche, V., Rossberg, D., 1997. Synops 1.1: a model to assess and to compare the environmental risk potential of active ingredients in plant protection products. Agric. Ecosyst. Environ. 64, 181–188.
Horticulture Australia Ltd, 2023. Final Report - crop protection replacement for diuron in pineapple industry. URL: https://www.horticulture.com.au/globalassets/laserfiche/assets/project-reports/st15029/st15029-final-report-complete.pdf. (Accessed 22 September 2023).
Jacometti, M., Wratten, S., Walter, M., 2010. Alternatives to synthetic fungicides for botrytiscinerea management in vineyards. Aust. J. Grape Wine Res. 16, 154–172.
Jactel, H., Verheggen, F., Thíery, D., Escobar-Gutiérrez, A.J., Gachet, E., Desneux, N., Group, N.W., et al., 2019. Alternatives to neonicotinoids. Environ. Int. 129, 423–429.
Kathage, J., Castañera, P., Alonso-Prados, J.L., Gómez-Barbero, M., Rodríguez-Cerezo, E., 2018. The impact of restrictions on neonicotinoid and fipronil insecticides on pest management in maize, oilseed rape and sunflower in eight European Union regions. Pest Manag. Sci. 74, 88–99.
Khan, S.U., 1980. Pesticides in the Soil Environment. Elsevier.
Kniss, A.R., Coburn, C.W., 2015. Quantitative evaluation of the environmental impact quotient (EIQ) for comparing herbicides. PLoS One 10, e0131200.
Kudsk, P., Mathiassen, S.K., 2020. Pesticide regulation in the European Union and the glyphosate controversy. Weed Sci. 68, 214–222.
Kudsk, P., Jørgensen, L.N., Ørum, J.E., 2018. Pesticide Load—a new Danish pesticide risk indicator with multiple applications. Land Use Pol. 70, 384–393.
Leach, A., Mumford, J., 2008. Pesticide environmental accounting: a method for assessing the external costs of individual pesticide applications. Environ. Pollut. 151, 139–147.
Lewis, K.A., Tzilivakis, J., Warner, D.J., Green, A., 2016. An international database for pesticide risk assessments and management. Hum. Ecol. Risk Assess. 22, 1050–1064.
Liu, S., Carlson, G.A., Hoag, D.L., 1995. Trade-off analysis of herbicide withdrawals on agricultural production and groundwater quality. J. Agric. Appl. Econ. 27, 283–300.
Maggi, F., Tang, F.H., la Cecilia, D., McBratney, A., 2019. PEST- CHEMGRIDS, global gridded maps of the top 20 crop-specific pesticide application rates from 2015 to 2025. Sci. Data 6, 1–20.
Manitoba Pulse and Soybean Growers, 2023. Lambda-Cy and Alternative Chemistries: A Factsheet. URL: https://www.manitobapulse.ca/wp-content/uploads/2023/04/MCA_Lambda-Cy-and-Alternative-Chemistries-factsheet-v4.pdf. (Accessed 22 September 2023).
Mankong, P., Fantke, P., Phenrat, T., Mungkalasiri, J., Gheewala, S.H., Prapaspongsa, T., 2022. Characterizing country-specific human and ecosys-tem health impact and damage cost of agricultural pesticides: the case for Thailand. Int. J. Life Cycle Assess. 27, 1334–1351.
Michigan State University Extension, 2023a. Alternatives to EBDC fungicides in grapes. URL: https://www.canr.msu.edu/news/alternatives_to_ebdc_fungicides_in_grapes. (Accessed 22 September 2023).

Michigan State University Extension, 2023b. Broad spectrum fungicides for vegetables. URL: https://www.canr.msu.edu/news/broad_spectrum_fungicides_for_vegetables. (Accessed 22 September 2023).

Mississippi State University Extension, 2023. Alternate postemergence strategies for weed control in cotton. URL: http://extension.msstate.edu/publications/alternate-postemergence-strategies-for-weed-control-cotton. (Accessed 22 September 2023).

Möhring, N., Gaba, S., Finger, R., 2019. Quantity based indicators fail to identify extreme pesticide risks. Sci. Total Environ. 646, 503–523.

Möhring, N., Bozzola, M., Hirsch, S., Finger, R., 2020. Are pesticides risk decreasing? The relevance of pesticide indicator choice in empirical analysis. Agric. Econ. 51 (3), 429–444.

Monfreda, C., Ramankutty, N., Foley, J.A., 2008. Farming the planet: 2. Geographic distribution of crop areas, yields, physiological types, and net primary production in the year 2000. Global Biogeochem. Cycles 22.

Moss, S., Ulber, L., den Hoed, I., 2019. A herbicide resistance risk matrix. Crop Protect. 115, 13–19.

Nielsen, H.Ø., Konrad, M.T.H., Pedersen, A.B., Gyldenkærne, S., 2023. Ex- post evaluation of the Danish pesticide tax: a novel and effective tax design. Land Use Pol. 126, 106549.

North Carolina State University Extension, 2018. Are there alternatives to glyphosate for weed control in landscapes? URL: https://content.ces.ncsu.edu/are-there-alternatives-to-glyphosate-for-weed-control-in-landscapes. (Accessed 22 September 2023).

PAN Europe, 2020. Mancozeb factsheet 2020. URL: https://www.pan-europe.info/facsheets/factsheet-mancozeb. (Accessed 22 September 2023).

Perry, E.D., Moschini, G., 2020. Neonicotinoids in US maize: insecticide substitution effects and environmental risk. J. Environ. Econ. Manag. 102, 102320.

Petkar, A., Langston, D.B., Buck, J.W., Stevenson, K.L., Ji, P., 2017. Sensitivity of fusarium oxysporum f. sp. niveum to prothioconazole and thiophanate-methyl and gene mutation conferring resistance to thiophanate-methyl. Plant Dis. 101, 366–371.

Rasche, L., 2021. Estimating pesticide inputs and yield outputs of conventional and organic agricultural systems in Europe under climate change. Agronomy 11, 1300.

Reus, J.A., Leendertse, P.C., 2000. The environmental yardstick for pesticides: a practical indicator used in The Netherlands. Crop Protect. 19, 637–641.

Reus, J., Leendertse, P., Bockstaller, C., Fomsgaard, I., Gutsche, V., Lewis, K., Nilsson, C., Pussemier, L., Trevisan, M., van Der Werf, H., et al., 2002. Comparison and evaluation of eight pesticide environmental risk indicators developed in Europe and recommendations for future use. Agric. Ecosyst. Environ. 90, 177–187.

Richard, S., Moslemi, S., Sipahutar, H., Benachour, N., Seralini, G.E., 2005. Differential effects of glyphosate and roundup on human placental cells and aromatase. Environ. Health Perspect. 113, 716–720.

Scott, C., Bilsborrow, P.E., 2019. The impact of the EU neonicotinoid seed- dressing ban on oilseed rape production in England. Pest Manag. Sci. 75, 125–133.

Siviter, H., Muth, F., 2020. Do novel insecticides pose a threat to beneficial insects? Proceedings of the Royal Society B 287, 20201265.

Skalský, R., Tarasovičová, Z., Balkovič, J., Schmid, E., Fuchs, M., Moltchanova, E., Kindermann, G., Scholtz, P., McCallum, I., 2012. Global Homogeneous Response Units. PANGAEA.

Tesfamichael, A.A., Kaluarachchi, J.J., 2006. A methodology to assess the risk of an existing pesticide and potential future pesticides for regulatory decision-making. Environ. Sci. Pol. 9, 275–290.

Topping, C.J., Aldrich, A., Berny, P., 2020. Overhaul environmental risk assessment for pesticides. Science 367, 360–363.

Townsend, R., Pottinger, R., Rowe, S., et al., 1979. Evaluation of alternative insecticides for lucerne flea on pasture. In: Proceedings of the Thirty-Second New Zealand Weed and Pest Control Conference. Southern Cross Hotel, Dunedin August 7th to 9th, 1979. New Zealand Weed and Pest Control Society Inc, pp. 68–71.

Udias, A., Galimberti, F., Dorati, C., Pistocchi, A., 2023. Emissions of pesticides in the European Union: a new regional-level dataset. Sci. Data 10 (1), 869.

University of Georgia Extension, 2018. Chemical Nematicides for Control of Plant-Parasitic Nematodes in Georgia Vegetable Crops. //extension.uga.edu/publications/detail.html?number=B1502&title=chemical-nematicides-for-control-of-plant-parasitic-nematodes-in-georgia-vegetable-crops. (Accessed 22 September 2023).

University of Tennessee Extension, 2023. Management of small grain dis- eases: fungicide efficacy for control of wheat diseases 2023. URL: https://extension.tennessee.edu/publications/Documents/W341.pdf. (Accessed 22 September 2023).

Uppala, S., Zhou, X., 2018. Field efficacy of fungicides for management of sheath blight and narrow brown leaf spot of rice. Crop Protect. 104, 72–77.

Vincent, C., Hallman, G., Panneton, B., Fleurat-Lessard, F., 2003. Management of agricultural insects with physical control methods. Annu. Rev. Entomol. 48 (1), 261–281.

Vinpro, 2021. Alternatives to Manage Weeds and Fungal Diseases. https://vinpro.co.za/alternatives-to-manage-weeds-and-fungal-diseases/. (Accessed 22 September 2023).

Wagner, N., Rödder, D., Brühl, C.A., Veith, M., Lenhardt, P.P., Lötters, S., 2014. Evaluating the risk of pesticide exposure for amphibian species listed in Annex II of the European Union Habitats Directive. Biol. Conserv. 176, 64–70.

Zilberman, D., Schmitz, A., Casterline, G., Lichtenberg, E., Siebert, J.B., 1991. The economics of pesticide use and regulation. Science 253, 518–522.