Title: How pesticide exposure and effects match with the intention of European pesticide regulation – a mini review

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Abstract:

Pesticides are widely used in European agriculture, requiring robust prospective risk assessment of their environmental exposure and effects on non-target species to safeguard biodiversity. We conducted a high-level evaluation to test whether: 1) measured environmental exposure concentrations remain below those established by risk assessment and 2) these concentrations prevent population- or community-level effects in non-target organisms. We systematically analysed meta-analyses, quantitative reviews and syntheses that compared predicted and measured concentrations or assessed the effects of pesticides on non-target organisms. For exposure, studies show that in both aquatic and soil ecosystems the predicted concentrations of exposure models or regulatory thresholds are frequently exceeded. For effects, the data demonstrate frequent occurrence of negative, i.e. detrimental effects on non-target organisms. Impacts on aquatic species appear more pronounced than on terrestrial communities. Overall, the evidence from synthetic scientific studies suggests that current environmental risk assessment in the European Union recurrently underestimates both environmental exposure to pesticides and their ecological effects.

Keywords: Insecticide, Fungicide, Herbicide, Modelling, Meta-analysis, Impacts, Agriculture.

Highlights:

- Evaluate predictive quality of European pesticide risk assessment
- Meta-analysis of quantitative syntheses on pesticide exposure and effects
- Predicted exposure concentrations and regulatory thresholds frequently exceeded
- Frequent occurrence of detrimental effects on non-target organisms
- EU risk assessment underestimates both exposure and effects

1. Introduction

Agricultural land use has resulted in major changes in world’s ecosystems and biodiversity [1]. It is the dominant land use in Europe, making it one of the world’s most intensively farmed regions. Several studies documented that European agriculture has been associated with the loss of species richness of several organism groups including plants [2,3], terrestrial insects such as butterflies, moths and bees [4], farmland birds [5] and freshwater organisms such as macrophytes, fish and invertebrates [6]. Agricultural land use is the primary contributor to the extinction risk faced by approximately 20% of species on European Red Lists [7], with farmland species exhibiting the strongest declines in biodiversity in several meta-analyses [8–11]. A wide range of practices associated with agricultural intensification drive the biodiversity decline, including excessive fertilizer use, degradation and loss of edge habitats and landscape simplification in general, deep ploughing, abandoning of crop rotation and pesticide use [12]. Several authors have raised concerns that pesticide use, in particular, is an important driver of biodiversity decline, despite a comprehensive environmental risk assessment in Europe designed to mitigate effects on biodiversity [13–19]. Thus, current environmental risk assessment may underestimate pesticide exposure and effects on non-target organisms. Here, we provide an overview of results from studies that compared the predictions from risk assessment to measured pesticide concentrations and that assessed effects on non-target organisms.

2. Methods

For both exposure and effects, we conducted a literature analysis in the Web of Science with the aim to identify meta-analyses, quantitative syntheses and reviews, hereafter "synthetic studies", that compare predicted and measured pesticide concentrations in ecosystems or assess the effects of pesticides on non-target organisms in ecosystems (see Appendix A for details). To balance the number of studies with regulatory relevance, we limited the search to studies published in the last decade (1 January 2015–1 May 2025), recognising that the number of currently used pesticides represented in older studies diminishes over time. We considered comparisons between predicted and measured pesticide exposure irrespective of pesticide identity, assuming that the predictive capacity of exposure models is largely independent of pesticide identity and rather governed by general physicochemical properties. This assumption is in line with the generic approach of pesticide exposure models that use physicochemical properties in prediction and ignore pesticide classes [20], although this assumption may fail for very hydrophobic and ionizable compounds [21,22]. For pesticide effects, we aimed to focus on current-use pesticides in Europe, defined as those currently authorised for use in at least one country of the European Union (EU). Accordingly, we excluded studies dealing exclusively with pesticide classes such as organophosphate or organochlor insecticides, which have been largely withdrawn or banned within the EU. Here, we placed greater emphasis on the pesticide class than in the exposure analysis, as the current regulatory effect assessment has been a primary driver for the withdrawal or non-renewal of authorisations for older, high-risk pesticides. Consequently, comparisons involving pesticide classes that are already banned do not provide relevant evidence for evaluating the effectiveness of current regulation. While we found a larger number of synthetic studies on pesticide effects, such studies were rare for pesticide exposure so that we also included original studies. Details on the literature search and on the initial screening of abstracts with a machine learning-based screening tool are provided in Appendix A.

For exposure, we extracted 23 cases from 11 studies (Appendix B). The vast majority of these cases (21) evaluated measured pesticide concentrations with respect to a regulatory threshold or the prediction from an exposure model. The fraction of measured pesticide concentrations exceeding a threshold or prediction could therefore be used as a response variable across these cases. Given the differences in study design (e.g. focussing on water or soil samples) and methods, we abstained from computing a pooled effect estimate for this response variable. Instead, we used beta regression to test for potential differences in proportional exceedances between environmental media sampled, i.e. soil or water, reference type, i.e. regulatory threshold or model prediction, or between types of exceedance. We defined the latter by the level of aggregation used to calculate the frequency of exceedance. For instance, we measured exceedance as the fraction of sites, the fraction of pesticides, or the fraction of detected concentrations that exceeded the thresholds or predictions. The factors environmental media sampled and reference type were highly correlated (Cramer’s V correlation coefficient: 0.89), which prohibited disentangling their individual effects. We used a logit link for the mean and an identity link for the precision parameter phi. Model fitting was performed using maximum likelihood estimation in the betareg R package [23]. We used randomized quantile residuals in model diagnostics.

For effects, we extracted 51 cases from 10 studies (Appendix C). The fact that studies used different measures of effect sizes (e.g. log response ratios (LRR), log odds ratio, Hedge’s d, proportional) that lack a direct method of conversion between them, hampered an overall evaluation of effect sizes. Therefore, we evaluated the cases with respect to the effect direction and statistical significance. The latter was only assessed for cases that reported 95% confidence intervals, where intervals excluding 0 were interpreted as statistically significant. For the proportional effect size (i.e. % positive, negative or neutral effects), we used a multinomial logistic regression with the outcome category as response and the response type in the study (i.e. abundance, biomass, mortality, diversity (e.g. Shannon diversity) and taxon richness) as predictor. The model was implemented with the function multinom() in the nnet R package [24]. To evaluate statistical significance of the response type, the fitted model was compared to an intercept only null model using a likelihood ratio test. Note that all data on proportional effect sizes originated from a single study on soil invertebrates. Finally, we analysed whether the log response ratios (LRR) differed between organism groups, pesticide type in terms of fungicides, herbicides or insecticides and between the aquatic and terrestrial biome. This was done using the function rma() in the metafor R package [25]. Prior to the analysis, we pooled effect estimates for responses of soil invertebrates, where multiple cases were available for the same pesticide type. In the model, we included an interaction between organism group and pesticide type, as it is well documented that the most sensitive organism group differs across pesticide classes [26]. All data and computer code to reproduce the analysis is publicly available at: https://github.com/schaeferRCOHR/Meta_pesticides

3. Results

3.1 Comparison of measured pesticide concentrations with regulatory thresholds and model predictions

The proportions of exceedances differed significantly between types of exceedances, i.e. depending on whether data were aggregated at the level of sites, pesticides or detected concentrations (beta regression, $p < 0.001$). The estimated mean proportion of exceedances in beta-regression was 0.73

for sites and therefore considerably higher than for pesticides (0.12) and detected concentrations (0.13). The proportion of sites exceeding regulatory thresholds or exposure model predictions ranged from 25% to 100% of sites (Figure 1). In most studies, less than 12% of detected pesticides were responsible for exceedances, whereas the exceedances of detected pesticide concentrations varied from 5% to 45%. Regulatory thresholds and model predictions exhibited similar exceedances at the analytical levels of sites and detected concentrations (Figure 1). At the level of pesticides, regulatory thresholds showed exceedances in 25–38% of cases, whereas model predictions indicated at most 13%. However, neither the sampling compartments water vs. soil nor the reference type regulatory threshold vs. model prediction exhibited a significant difference ($p = 0.54$ and $p = 0.67$). The two studies that considered the degradation of pesticides found that 65% and 90% of pesticide residues degraded more slowly than based on the degradation half time (DT50) determined in regulatory exposure assessment (Appendix B).

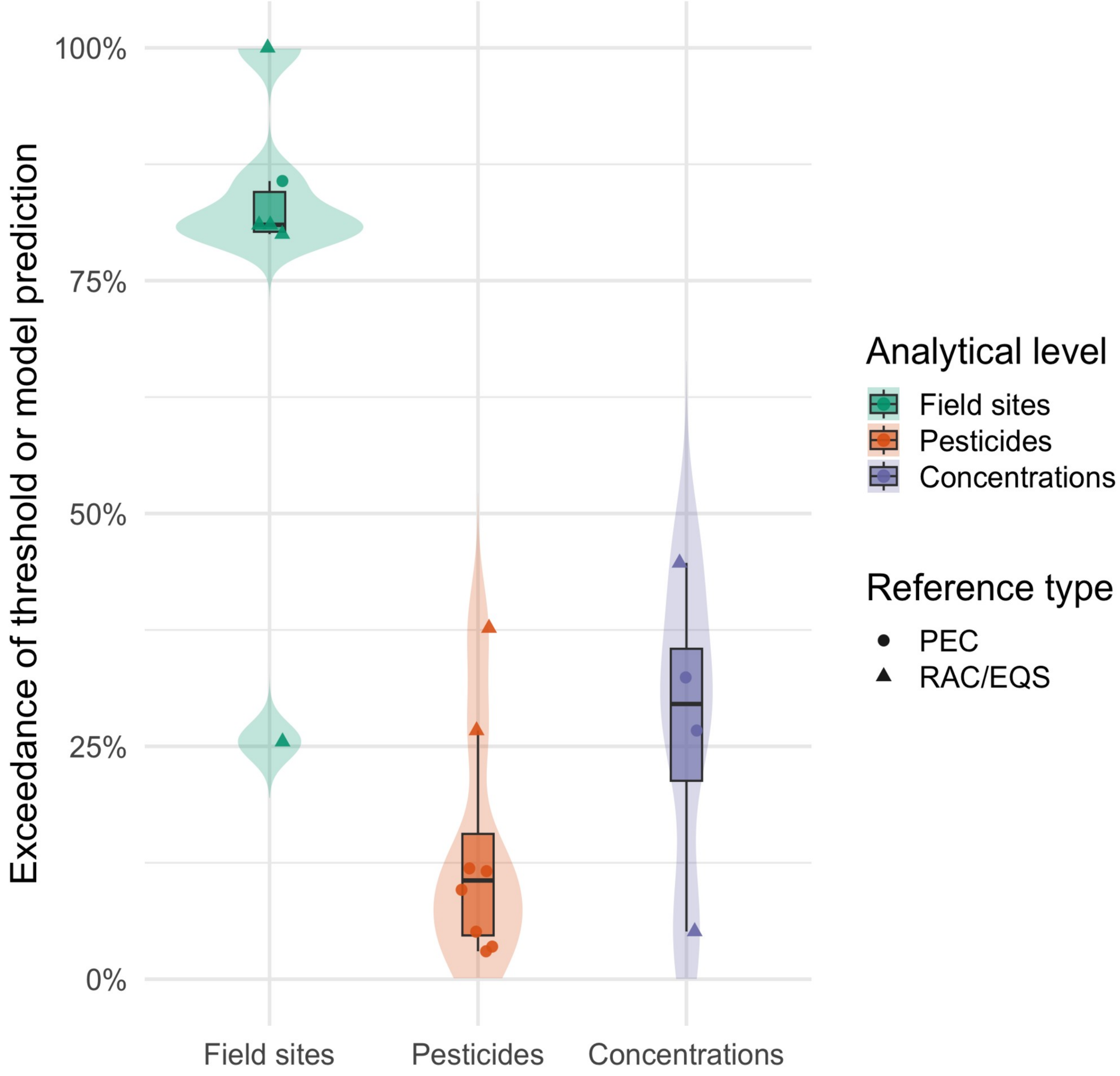

Figure 1: Exceedance of regulatory thresholds (i.e. environmental quality standard (EQS) and regulatory acceptable concentration (RAC)) or predicted exposure concentrations (PECs) at the level of field sites, pesticides or detected concentrations, termed type of exceedance. Each dot represents the result of a case for aggregation at the level of sites, pesticides or detected

concentrations (Appendix B, n = 18). Shapes differentiate regulatory threshold from PECs. Four cases related to days where pesticides exhibited an exceedance (2 cases) and to degradation time (2 cases) are not shown. In the one case where exceedance values were available for both detected and all analysed concentrations, we only present the detected concentrations to ensure metric consistency with all other included studies. Note that the factor reference type was strongly associated with the sampling compartment (water vs. soil), so any apparent differences in exceedances cannot be attributed to one factor unambiguously.

3.2 Widespread effects on non-target organisms at environmentally realistic concentrations

Of the 51 cases, 39 effect directions were negative, 7 indicated no effect and 5 were positive (Figure 2, Appendix C). When restricting this only to the 17 statistically significant cases, all effect directions were negative (Appendix C). Only 7 cases focussed exclusively on current-use pesticides, and all of these were negative with 4 statistically significant.

The proportional effects were approximately evenly distributed between % negative and % no effects, except for mortality where the proportion of negative effects was twice that of neutral effects (Figure 2). However, the likelihood ratio test comparing the model with response type as predictor to the null model was not statistically significant (multinomial logistic regression, $\chi^2_{30} = 30.21$, $p = 0.46$), suggesting that, overall, the distribution of positive, neutral, and negative effects did not differ significantly across response types. Positive effects of pesticides were negligible (Figure 2).

All estimated LRRs were negative except for the combinations of terrestrial plants or animals with fungicides, which showed slightly positive values (Figure 3). For most combinations, the confidence intervals reached from half a log unit to one log unit. Among terrestrial organisms, only 2 of the 10 confidence intervals excluded zero, whereas in aquatic organisms 3 out of 5 estimates were clearly below zero, and the remaining 2 marginally overlapped with zero. This result is reflected in the meta-analysis model, where only the moderator biome (terrestrial vs. aquatic) was statistically significant ($p = 0.02$). The other moderators (pesticide type, organism group and their interactions including with biome) were not statistically significant and removed during model simplification. The test for residual heterogeneity was not significant ($p = 0.60$), indicating that little unexplained variance remained after accounting for the biome effect.

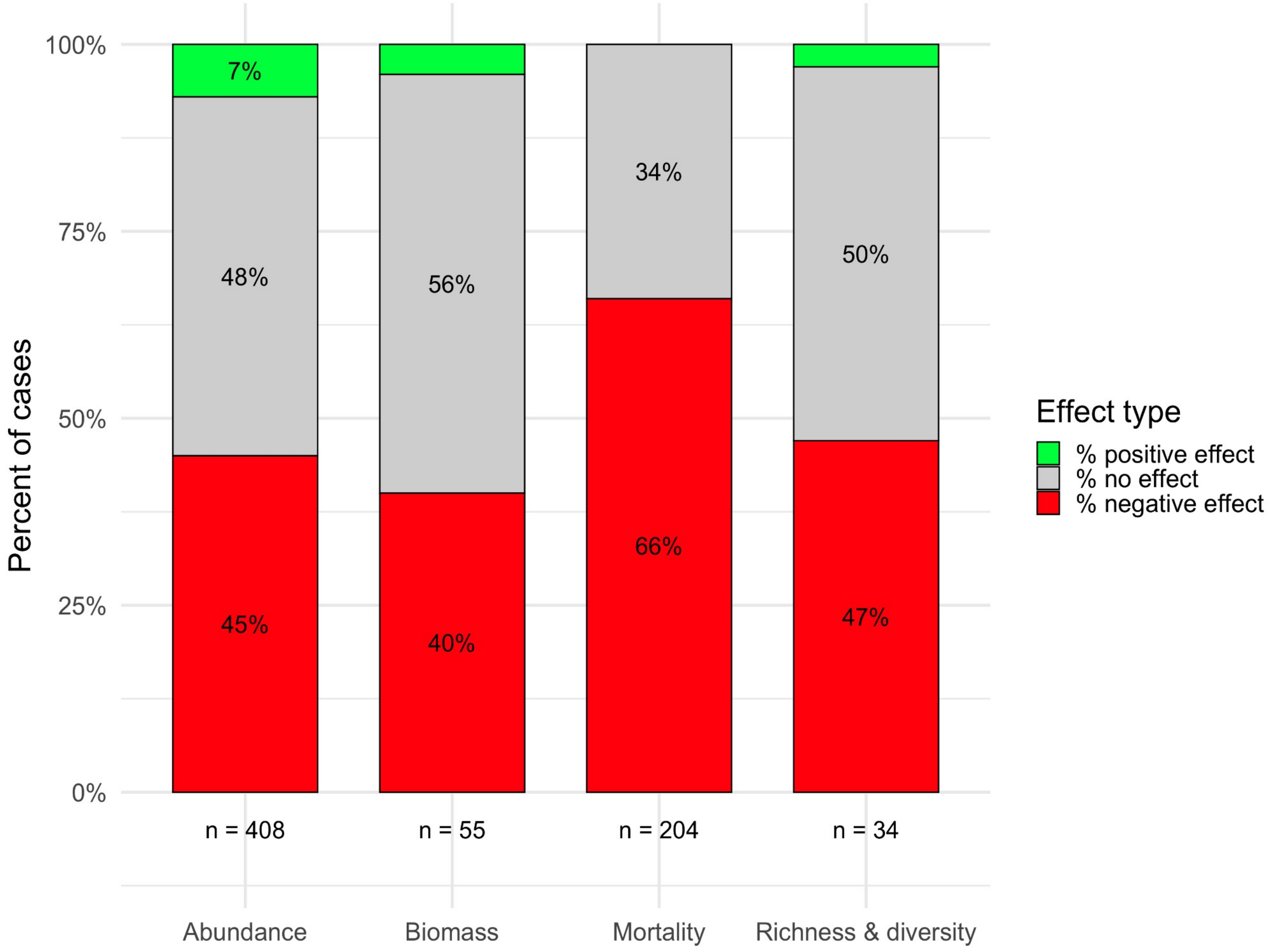

Figure 2: Distribution of effect types across response types in the studies. n = number of cases considered in the meta-analysis. A positive and negative effect refers to an increase or decrease in abundance, biomass and richness or diversity in comparison to a control treatment, respectively. An increase or decrease in mortality compared to a control treatment represented a negative and positive effect, respectively.

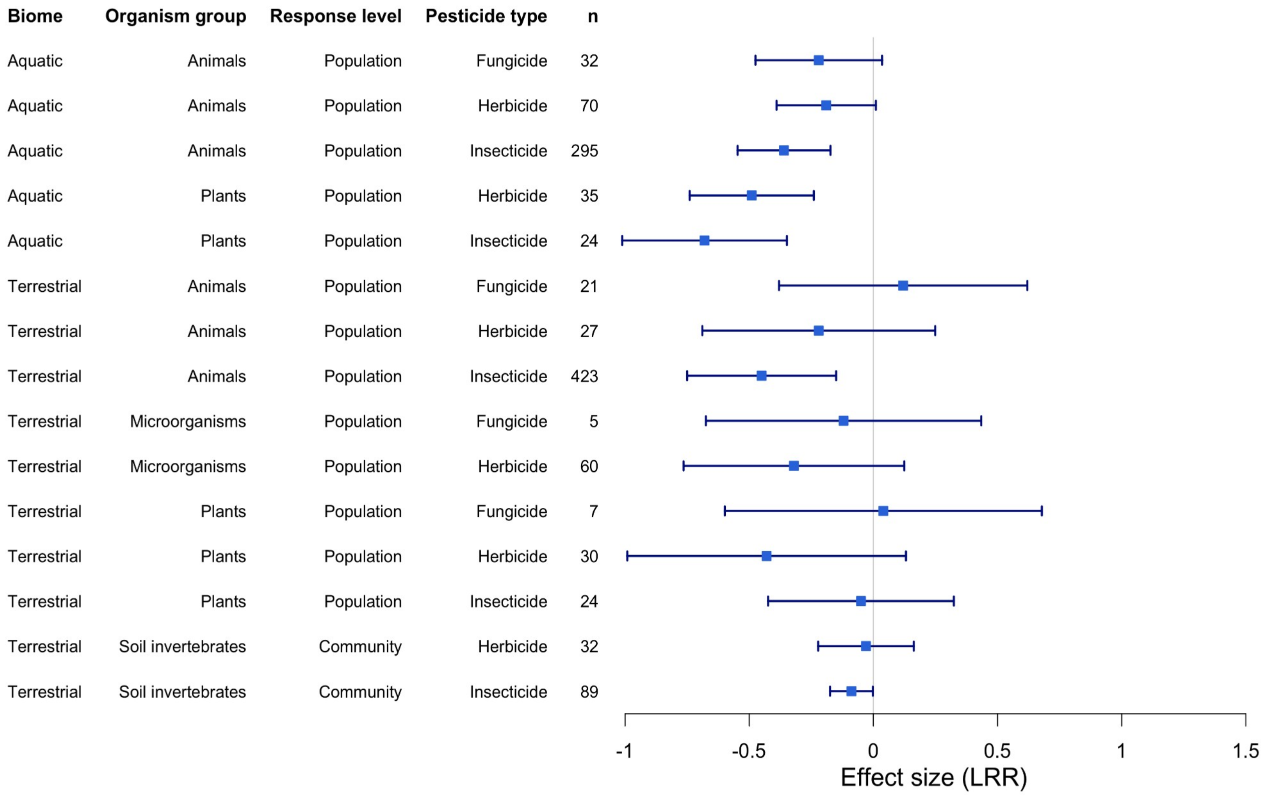

Figure 3: Log-response ratios (LRR) for different combinations of biome, organism group, response level and pesticide type with sample size (n) and 95% confidence intervals.

4. Discussion

4.1 Reasons for the frequent exceedance of thresholds and model predictions

Measured environmental concentrations frequently exceeded regulatory thresholds and model-predicted exposure concentrations (Figure 1), although the magnitude and frequency of exceedances varied with the analytical level of aggregation (sites, substances, or individual concentrations, Figure 1). When considering the fraction of exceedances at the analytical level of sites, 75% to 100% of sites exhibited at least one exceedance in most studies. This widespread spatial prevalence of pesticide pollution contradicts the intended outcome of complex environmental risk assessment frameworks, which are designed to provide realistic worst-case exposure predictions and prevent frequent threshold exceedances [13,27]. Multiple reasons may explain this discrepancy. First, predicted exposure concentrations are initially derived from standardised models, typically relying on laboratory-based degradation half-lives and default assumptions about environmental conditions and pesticide use. However, these conditions rarely capture the complexity of real-world agricultural landscapes. Actual fate, transport, partitioning, and degradation of pesticides are governed by a dynamic interplay of intrinsic chemical properties (e.g. polarity, hydrophobicity) and highly variable environmental conditions (e.g. temperature, moisture, pH). Consequently, some scenarios of environmental conditions, non-extractable residue (NER) formation, and repeated or mixed applications can lead to slower degradation, i.e. greater persistence and higher actual concentrations than predicted [28–31]. Moreover, pesticide transport processes such as aerial transport, erosion, and remobilization may be underestimated in current models, leading to unexplained high concentrations [32–34]. Evidence of this off-target transport comes from the detection of pesticide mixtures in organically managed agricultural systems, even after decades without pesticide application, and in the atmosphere, including supposedly non-volatile compounds such as glyphosate [32,33]. These ubiquitously detected mixtures in agricultural soils, where urban or point-source pollution is minimal, challenge claims that the widespread occurrence of exceedances can be explained by non-agricultural sources or malpractice [see e.g. ,35], though these may locally play a role.

4.2 Reasons for effects on non-target organisms and study limitations

Across the meta-analyses, pesticides predominantly had negative effects on non-target organisms (Figure 2). Two reasons explain this finding. First, the vast majority of active substances affect physiological pathways that are shared by target and non-target species, sometimes from multiple organism groups [see, e.g. for fungicides 36]. Second, pesticide concentrations that appear unproblematic within standardised regulatory test systems can elicit substantial effects in complex ecosystems. The latter can for instance be due to the interaction of multiple stressors and chemical mixtures in their effects on organisms [37,38] as well as ecological processes such as biotic interactions [39,40] and latent effects [41]. Neglecting such processes is of particular concern, given that lower trophic level effects may propagate to higher trophic levels, as documented by declines in populations of insectivorous farmland birds [5].

For studies reporting log response ratios, the effects on aquatic organisms seemed stronger and less variable than on terrestrial organisms (Figure 3). This result requires cautious interpretation due to potential methodological differences such as bias from the selection of test species, test setup (e.g., laboratory vs. field) and in the influence of environmental conditions. The variability in soil types, pH, vegetation diversity, and climatic conditions may create a higher variability compared to aquatic studies under frequently standardised conditions.

Our findings support the long-known negative effects of insecticides as documented by several hundreds of experimental cases[38,42,43] . While negative impacts on animals including invertebrates are expected given the mechanisms of action, the strong negative responses of aquatic plants to insecticides were unexpected and suggest potential indirect effects on aquatic animals.. However, pooling of heterogeneous studies, combining diverse pesticides, organism groups, and effect types (e.g. biomass, reproduction) may have incurred substantial variability, which is a common limitation of meta-analyses. For example, the broad group "animals" encompasses both invertebrates and vertebrates, which differ substantially in sensitivity to modern insecticides, so pooling may mask stronger effects on invertebrates. Without more detailed re-analysis of the underlying primary studies, which is beyond the scope of our synthesis, it remains unclear why aquatic plants appeared more sensitive than animals to insecticides. In addition, variability among source studies, potential publication bias, and limited taxonomic resolution in the syntheses constrain mechanistic interpretation and may affect the precision and ecological generality of the findings of meta-analyses and syntheses including ours.

5. Conclusions for current risk assessment

Despite substantial methodological advances in risk assessment frameworks in the last decade, contemporary regulatory approaches continue to fail to adequately protect ecosystems from the non-target effects of agricultural pesticide use, and regulatory thresholds and predicted exposure concentrations are frequently exceeded. A critical limitation is that most meta-analyses on effects included pesticides no longer authorised, making current regulatory frameworks difficult to evaluate. However, of the remaining 7 synthetic studies focusing exclusively on current-use pesticides (or re-analysed accordingly), all effect directions were negative and 4 demonstrated statistically significant effects on microorganisms and terrestrial invertebrates. These results suggest that current-use pesticides continue to affect non-target organisms and biodiversity.

Author contributions: CRediT

Conceptualization: All. Data curation: RBS, AS, MS. Formal analysis: RBS. Methodology: All. Writing – original draft: All. Writing review and editing: All.

Acknowledgements

Two thank two anonymous reviewers for helpful suggestions.

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.

Funding sources

This work was carried out within the framework of the European Partnership for the Assessment of Risks from Chemicals (PARC) and has received funding from the European Union’s Horizon Europe research and innovation programme under Grant Agreement No. 101057014. However, the views and opinions expressed are those of the authors only and do not necessarily reflect those of the European Union or the Health and Digital Executive Agency. Neither the European Union nor the granting authority can be held responsible for them.

Appendices

Appendix A: Literature search and AI-based abstract screening

While we found a larger number of synthetic studies on pesticide effects, such studies were rare for pesticide exposure so that we also included original studies. For pesticide exposure, we therefore used the following search terms: (pesticide* OR insecticide* OR herbicide* OR fungicide* OR agrochemical*) AND (fate* OR exposure* OR degradation* OR half life* OR half-life* OR DT50* OR decline* OR predicted* OR PEC* OR RAC*) AND (soil* OR water* OR sediment* OR air*). For pesticide effects, the following terms were used: TS=(("pesticide*" OR insecticide* OR herbicide* OR fungicide* OR agrochemical*) AND ("effect*" OR "impact*" OR "affect*" OR "impair" OR "ecotoxicology" OR "ecological risk" OR "decline" OR "loss") AND (plant* OR invertebrate* OR insect* OR pollinator* OR bee* OR butterfly* OR vertebrate* OR amphibian* OR bird* OR fish OR mammal* OR microbe* OR bacteria OR fung* OR "non-target organism*

OR soil organism* OR mycorrh* OR nematod* OR earthw* OR lumbric* OR enchytr* OR collembol* OR mite* OR acari* isopod* OR coleopt* OR dipter* OR hymenopter* OR ant* OR arachnid* OR opilion* ")) AND TS=("meta-analysis" OR "systematic review" OR synthesis OR review).

The searches resulted in 4,000 and 10,798 unique records for the exposure and effects query, respectively, which were then imported and screened in the machine learning-based screening tool ASReview, version 2.1.1 [44]. Among these records, 9 and 13 records were selected as priors for the learning algorithm in terms of papers that are considered relevant based on their abstract. The tool presents abstracts to the user, who rates their relevance. Based on this rating, the tool successively improves the discrimination of relevant from non-relevant abstracts. The overall aim is to reduce screening time by training the tool to identify non-relevant abstracts. A typical stopping rule requires that, first, a minimum fraction of the articles has been screened and, second, a specified number of abstracts are rated as non-relevant in a row. Our stopping criterion was that at least 7.5% of the abstracts were rated and that subsequently at least 100 abstracts were rated as non-relevant in a row. This is higher, and more conservative, than the stopping rule of a threshold of 50 consecutive irrelevant records that was considered by experts as safe and reasonable when using the tool [45]. Moreover, compared to a recent screening study on multiple stressor studies that set a threshold of at least 5% for abstract rating [46], we selected a proportionally larger fraction of abstracts. This higher inclusion rate was justified by the considerably smaller total number of abstracts in our search, which allowed for a comprehensive screening of a larger proportion within reasonable time. For the exposure data set, the stopping criterion of at least 7.5% of abstracts screened coincided with the 100 non-relevant reviews in a row criterion. For the effects data set, 100 non-relevant consecutive reviews were suggested by ASReview once 7.7% of abstracts were screened.

For exposure, we considered the following criteria when screening the abstracts: 1) Pesticide concentrations measured and compared to predictions or regulatory thresholds, 2) spatial focus on Europe, 3) synthetic organic chemicals (i.e. no oil-based biopesticide, or inorganic pesticide) and 4) larger-scale studies with at least 20 observations. The latter criterion was selected to exclude case studies on single fields or water bodies, and to aim for a larger spatial coverage. When abstracts lacked this information, we rated them as potentially relevant and included them in the deep screening. We found 41 relevant records of which the full text was subsequently deep screened. During deep screening, we re-applied the above criteria and also excluded four studies that lacked quantitative data, focussed on the marine environment or reanalysed the same data set (Appendix B).

For effects, the criteria to rate an abstract as relevant were: 1) evaluation of pesticide effects on non-target organisms that occur in Europe (e.g. exclusion of studies on tropical bee species), 2) effects reported at population-level or higher such as changes in the density or biomass of individual taxa or taxa within communities; we excluded studies focussing only on sublethal effects without information on population level effects, 3) synthetic organic and current use-pesticide in the European Union and 4) meta-analysis, review or synthesis; no individual studies considered. The latter criterion was selected because in comparison to the exposure studies, a much higher number of synthetic studies was available, which provide a more representative overview than individual studies. This led to 130 relevant records. Two non-peer reviewed journal articles were removed. The related full texts were screened manually to confirm that they fulfil the previous criteria. In detail, we specified the criteria during deep screening as follows: 1) Clear attribution of pesticide effects to agricultural pesticide use in the case of field studies, 2) environmentally-realistic exposure, in non-field studies, 3) pesticides currently authorised, where data allowed to evaluate this; we also considered studies that include non-authorised besides currently authorised pesticides and where data allowed us to remove the effect of non-authorised pesticides in our own calculations, 4) systematic review or synthesis aimed at exhaustive or representative literature sample, e.g. narrative reviews were omitted and 5) at least five observations from at least three different studies available.

Appendix B: Table with information on the exposure studies and extracted information

Appendix C: Table with information on the effect studies and extracted information

References and recommended reading

Papers of particular interest, published within the period of review, have been highlighted as:

* of special interest

** of outstanding interest

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