Title: Insufficient environmental protection by the European regulatory framework for pesticides

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Abstract

Schriever et al. (2025) argue that environmental risk assessment of pesticides in the European Union is sufficiently protective and that regulatory thresholds are rarely exceeded. Here, we re-examine these claims based on new and previous evidence from monitoring, systematic reviews, and different types of field studies. The clear outcome is that measured pesticide concentrations frequently exceed predicted concentrations and regulatory thresholds and that they relate to adverse impacts on ecological communities. The mechanisms driving this result include the joint action of pesticides applied in temporal and spatial proximity, varied environmental conditions, interactive effects of pesticides with additional stressors, and indirect ecological effects propagating within biological communities. These neglected intricacies explain why current single-substance prospective assessments tend to underpredict real pesticide exposure and impacts in field settings. The corollary is that the current regulatory framework in Europe proves insufficient to protect biodiversity and ecosystems. Necessary improvements embedded in a policy reform include strengthening post-registration monitoring, refining predictive exposure models, and explicitly considering landscape contexts, indirect and mixture effects, and interactions with non-chemical stressors.

Keywords

Agricultural policy, agrochemicals, biodiversity, environmental fate, food security, plant protection products, risk assessment, ecotoxicity

Background

The global economy has seen a steep rise in activity since the mid-20th century [1], accompanied by substantial improvements in human well-being, such as increased food production to support a growing world population. However, the profound changes in post-war economic and agricultural systems have also resulted in greatly elevated resource consumption and widespread environmental change. This has caused pervasive impacts on biodiversity [2–4], which, according to the Convention on Biological Diversity, encompasses taxonomic and functional variation in ecosystems at the level of populations (genotypes) and communities (species) [5].

Meta-analyses of biodiversity trends over the past few decades have produced mixed trends of biodiversity change, depending on land use, habitat, taxonomic group, time period, world region considered, and other factors [6]. For example, populations of terrestrial insects and birds analysed across Europe at the country scale have continuously declined over the last decades [7–11], with farmland species being most heavily affected [8,12–14]. Likewise, following previously sharp declines, ecological conditions of fresh waters regularly remain well below targets throughout Europe [15,16], notwithstanding that aquatic macroinvertebrate communities have partly recovered in recent years [17–19]. Notably, these recovery trends have been least pronounced in streams draining agricultural landscapes [12,14,19].

Modern agriculture heavily relies on pesticides for crop protection [20,21]. Their quantity and diversity continue to increase globally [22,23], and despite major efforts to improve specificity, most pesticides exhibit toxicity to non-target organisms, particularly those species closely related to the target pests [24]. In response, the European Union (EU) has developed an environmental risk assessment (ERA) scheme that aims at protecting biodiversity from pesticides (Fig. 1). Several studies have concluded that the ERA currently in place frequently underestimates pesticide exposure and effects, thereby contributing to biodiversity declines [25–32].

This conclusion has been challenged by Schriever et al. [33], who provide what they describe as a “more balanced evaluation” of the European pesticide ERA, referencing and challenging in particular the broad synthesis by Schäffer et al. [27]. The argument advanced by Schriever et al. [33] is that regulatory exposure thresholds are rarely exceeded, implying that the current ERA effectively safeguards biodiversity as long as farmers comply with good agricultural practice and no other emissions sources, for instance non-agricultural biocide uses, are present.

Here, we re-evaluate this contention in view of the current state of knowledge, primarily based on systematic reviews of field studies on pesticide exposure and effects, analyses of large-scale monitoring datasets and replicated field experiments. The underlying rationale is that considering the full scope of available evidence, including insights from multiple types of field investigations, is crucial for a balanced account of pesticide impacts in realistic field settings. We begin by outlining the current European regulatory framework for plant protection products and their active ingredients (i.e. pesticides), followed by a detailed reappraisal of the arguments put forward by Schriever et al. [33] in light of the available evidence base, particularly with regard to pesticide exposure, effects, and management strategies. Furthermore, we highlight inconsistencies in Schriever et al.’s [33] assessment with the available evidence, and identify prospective avenues in research and policy to resolve controversies and support the development of advanced, protective and proportionate ERA policies.

The European regulatory framework for plant protection products

Under Regulation (EC) No 1107/2009, any active substance (i.e. pesticide) contained in a plant protection product must first be approved for use in the EU based on the chemical properties and environmental behaviour of the pure compound [34]. The data required for this evaluation, including degradation rate constants and effects on standard non-target test organisms, are submitted by the pesticide manufacturer to EU authorities. Plant protection products containing an EU-approved active substance may then be authorised by individual EU Member States. This second step is essential because the marketed products are formulations that can contain one or more active ingredients and additional formulation chemicals that may affect the behaviour, bioavailability and efficacy of the pesticide against target and non-target organisms.

Risk of plant protection products is assessed based on the relationship between the Predicted Environmental Concentration (PEC) resulting from the application of a representative formulation according to good agricultural practice, and the Regulatory Acceptable Concentration (RAC). The latter refers to the legally permissible concentration limit established to protect non-target organisms from the risk of unacceptable pesticide effects. For risks to be acceptable under this framework, the PEC must be smaller than the RAC (Fig. 1). Importantly, temporary, local and fully reversible effects are not necessarily considered unacceptable, as long as recovery within a defined time frame is demonstrated. Consequently, RACs provide an inherently lower level of protection of non-target organisms than other regulatory thresholds such as the Environmental Quality Standards for aquatic organisms that are used in the context of the Water Framework Directive for freshwaters [35].

Risk assessment for active substances under the EU regulatory framework begins with a conservative first-tier evaluation. Here, presumed realistic worst-case exposure scenarios are compared with the RAC [36]. When the expected PEC exceeds the RAC, leading to a risk level considered unacceptable, the active substance can be re-evaluated in a higher-tier evaluation process involving more sophisticated exposure modelling than the original assessment and data from field-based fate and effect studies. Since these approaches draw on more detailed evidence from more realistic settings, narrower safety margins are acceptable than those used in the first-tier. In addition, the higher-tier re-evaluation process may also incorporate risk-mitigation measures such as the definition of margin buffer zones around agricultural fields, orchards and vineyards. The aim of adopting such a scenario-based logic is to ensure effective risk mitigation, which is achieved when the re-evaluated PEC based on the extended evidence is lower than the RAC derived from the extended data set. However, a remaining key limitation of the approach is its reliance on exposure models that often lack a solid empirical basis from post-registration monitoring and field experiments capable of directly testing whether protection goals are met in practice.

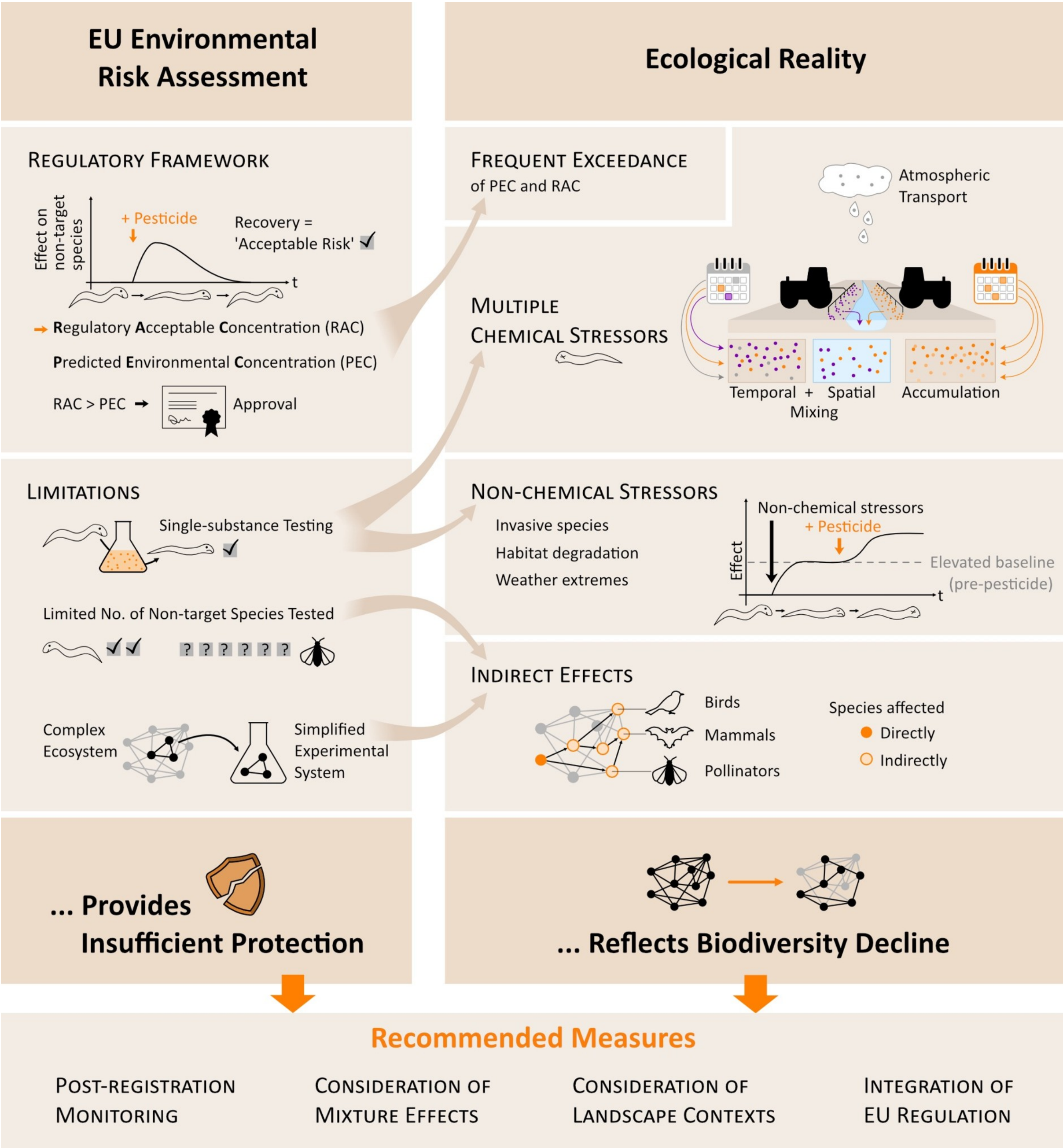

Figure 1. Conceptual framework contrasting EU environmental risk assessment for pesticides with ecological reality. The left panel illustrates the regulatory framework, including the key elements and limitations. The right panel depicts real-world exposure scenarios and ecological complexities that lead to frequent threshold exceedances and adverse effects on non-target organisms. The bottom panel summarises consequences for biodiversity protection and identifies recommended measures to address regulatory gaps.

Regulatory thresholds frequently exceeded in the field

Regulatory exposure modelling for estimating PECs in the EU draws on FOCUS models developed by the FOrum for the Coordination of pesticide fate models and their Use [36]. The models are calibrated to reflect 'realistic worst-case' scenarios intended to cover the 90th percentile of concentrations across space and time. In other words, Measured Environmental Concentrations (MECs) should not exceed the corresponding PECs, and therefore also the corresponding RACs, in more than 10% of the cases. However, a recent synthesis of large-scale exposure studies demonstrates that MECs exceed PECs, RACs and other thresholds in 5% to 45% of the cases [37]. Furthermore, this synthesis shows that in most published studies, these thresholds were exceeded at more than 75% of the investigated sites, with little difference between aquatic and terrestrial environments (Figure 1). This marked mismatch between predictions and field observations questions the reliability of current procedures to accurately predict pesticide risks in real-world situations, contrasting with the assertion by Schriever et al. [33] that the current system is protective for exposure.

In defence of the approach adopted by EU and national regulatory bodies, Schriever et al. [33] point out that the exceedance of thresholds could have at least three different causes: (i) a mismatch between local environmental conditions and FOCUS scenarios, (ii) non-compliant pesticide use by farmers, and (iii), particularly in relation to surface-water contamination, the potential contribution of non-agricultural pesticide use (e.g. urban applications and biocidal uses) and point sources to exposure not captured by the FOCUS models. Evaluating deviations of local environmental conditions from FOCUS scenarios can indeed be challenging. Likewise, reliable data on point source inputs of pesticides to the environment are often difficult to acquire, as is information on non-agricultural pesticide use and non-compliant applications on farms, which are assumed to be rare, similar to non-compliance rates for fertiliser use [38,39]. Moreover, only a minority of active substances are approved for substantial non-agricultural uses, suggesting that these sources are unlikely to account for the widespread occurrence of pesticides observed in the environment [40,41]. Importantly, all three causes are beyond the scope of current Environmental Risk Assessments for pesticides, as Schriever et al. [33] correctly indicate. However, this fact and the persisting difficulties notwithstanding, environmental impact derives from actual pesticide exposure and ecotoxicological effects, irrespective of emission sources. Therefore, even if FOCUS models were reliable and exceedances of thresholds were mainly driven by emissions from point sources, non-agricultural uses and/or non-compliant pesticide applications, risk management must still account for the impact of these emissions to ensure effective human and environmental protection. Ultimately, what matters for deciding on risk mitigation to protect biodiversity from chemical exposure is the total environmental load of hazardous substances, irrespective of the emission source. This reflects less a shortcoming of pesticide ERA itself than a lack of sufficient integration between the major regulatory frameworks (i.e. Water Framework Directive (WFD) [35], Regulation on Plant Protection Products [34], Biocidal Products Regulation [42] and REACH [43]).

Schriever et al. [33] refer to Bach and Hollis [44] to critique two empirical assessments indicating that FOCUS models systematically underpredict insecticide concentrations in the field [45,46]. This analysis ignores that most of the points raised by Bach and Hollis [44] have been already refuted [47,48]. In particular, that the observed underprediction of concentrations persists, or even increases, if the field studies contested by Bach and Hollis [44] are excluded from the analysis [47].

Conversely, evidence cited by Schriever et al. [33] suggesting that thresholds are rarely exceeded (<3% of days) derives from only a single study restricted to three locations [49]. The outcome of that study is at odds with a broad synthesis covering multiple independent studies, each covering tens to hundreds of sites distributed over large spatial scales [37]. This discrepancy is explained by the sampling design used by Hörold-Willkomm et al. [49], which involved daily sampling over multiple years. As a consequence, the great majority of samples were collected outside the seasonal application windows of plant protection products when environmental pesticide concentrations were often below detection limits. This case illustrates that monitoring pesticides decoupled from application windows is of limited value for validating the predictive performance of models of pesticide fate in the environment [50].

Additional evidence for pesticide risks comes from foundational studies comparing MECs and PECs, and data on pesticide fate. Both types of information are crucial for reliably assessing real-world pesticide impacts [51]. Two large-scale studies conducted where 30% of MECs exceeded PECs in agricultural field sites provide evidence of MEC exceedance over model predictions [52,53]. Similarly, the majority of pesticides detected in contaminated soils, for which pesticide application data were available, dissipated slower than was expected based on the degradation rate constants used in regulatory risk assessment [52,54]. Such discrepancies between predicted and measured exposure are likely to reflect the difficulty in regulatory risk assessments to mimic a wide variety of real-world environmental conditions where numerous factors can influence the half-life of active substances [55,56].

Two interrelated further problems are that 1) repeated application of pesticides is insufficiently addressed in EU risk assessment, leading to a neglect of potential pesticide residue accumulation [57,58] and 2) that additive and potential synergistic interactions among multiple compounds are not considered [59]. The application of multiple pesticides at any given site (Fig. 1) means that agricultural soils can commonly contain mixtures that comprise multiple compounds [41,52,60,61]. The capacity of soil microbial communities to degrade the individual compounds may thus be overwhelmed [41,62,63], an effect that can be exacerbated if synergistic toxic effects arise. This would lead to slower pesticide degradation than indicated by regulatory assays with single compounds – and potentially cause MECs to exceed PECs.

Appreciating a nuanced reality of pesticide risks

A further assert is that environmental risks from pesticide use have considerably declined over time [33]. However, analyses have shown that while risks may decrease in some setting for some organisms, they may increase for others, as shown by varied temporal patterns of vertebrates, bees and earthworms in an analysis addressing herbicide trends [64]. Schriever et al. [33] base their analysis primarily on a limited subset of the SYNOPS pesticide risk indicators, which combine chemical properties with application details to derive risk metrics [65]. Of the 10 available SYNOPS indicators for combinations of pesticide type and organism group, only 6 show reductions, with two remaining unchanged and two increasing [66]. An alternative metric, the Total Applied Toxicity (TAT) indicator [67] used in parallel, declined for four of the 10 combinations, being unchanged for two and increasing for four [66]. Reconciling the remaining divergence of outcomes of assessments using SYNOPS versus TAT is currently hindered by the lack of transparency, especially for the SYNOPS models, and a lack of indicator validation [see 66]. Without such a validation, claims of risk declines cannot be fully verified.

Are chemical risks to surface waters driven by legacy chemicals?

Schriever et al. [33] cite Rodea-Palomares et al. [68] to argue that chemical risks in surface waters are mainly driven by legacy chemicals and are confined to a limited number of sites, implying that currently used pesticides pose negligible risks. However, this interpretation does not account for key methodological choices in the underlying analysis. Rodea-Palomares et al. [68] analysed monitoring data from 2001–2015 but classified substances according to their regulatory status in 2022. This temporal mismatch leads to a misclassification of compounds as "legacy" chemicals, even though they were authorised and in use during the monitoring period. Of the 150 pesticides included, 102 were labelled as legacy substances, although 92 were approved for at least part of the monitoring period (26 of them throughout). Consequently, a substantial share of the risk attributed to "legacy" chemicals likely reflects exposure to then current pesticide use. This classification shapes the reported risk pattern. For example, excluding "legacy" chemicals yields exceedances in only 7% of the site–year combinations, whereas including them increases the proportion to 25%, a value more consistent with independent large-scale assessments reporting similar percentages (e.g. 37% [69] and 39% [70]). In these studies, pesticides were the dominant driver of chemical risk, accounting for over 50% [68] and even 85% [69] of the reported exceedances.

A limitation of analyses relying on monitoring data that must be acknowledged is that long-term data series suffer from methodological inconsistencies. This includes the numbers of monitored chemicals in the dataset used by Rodea-Palomares et al. [68] and Wolfram et al. [69]), which varied from <5 to ~40 across countries. Nevertheless, a recent harmonised screening of 610 chemicals across 22 European countries resulted in elevated levels of acute and chronic risks, with algae, invertebrates and fish being affected in 27%, 74% and 7% of the investigated sites, respectively [71]. Results that, together with information on exposure prevalence, point to current and recently used pesticides remaining as an important driver of ecological risk.

Chemical status of fresh waters as an indicator of pesticide risk?

Schriever et al. [33] emphasise that only 3% of EU surface waters fail to achieve good chemical status according to the Water Framework Directive (WFD) when 11 ubiquitous, persistent, bioaccumulative and toxic (uPBT) substances are excluded, a group of chemicals that does not comprise any current-use pesticides [72]. The low percentage is used to indicate that pesticide risks in European surface waters are negligible. However, under the WFD, chemical status is assessed for only 45 priority substances and eight additional pollutants, including the 11 uPBTs mentioned [73]. Only four pesticides authorised between 2010 and 2015, the time period of chemical status evaluation, are included in this list (n.b. it can take many years for substances to be nominated and ratified to this list), less than 1% of all authorised pesticides [72]. Most pesticides, along with pharmaceuticals and industrial chemicals, are instead treated as river-basin-specific pollutants within ecological, not chemical, status assessments. Furthermore, WFD monitoring covers only a limited subset of pesticides. Consistent with this, a comprehensive monitoring identified 39 pesticides exceeding thresholds (RACs) at 81% of sites, compared to 11 pesticides at 35% of sites under WFD-compliant approaches to compound selection and risk evaluation [31]. Additionally, WFD assessments may miss pollution hotspots by excluding small water bodies on the edge of fields [74]. Beyond these limitations, Weisner et al. [31] identified two additional reasons why WFD status assessments can underestimate pesticide risks in surface water: WFD monitoring schemes tend to miss episodic exposure events, and the applied regulatory thresholds may not fully capture ecological risks.

Schriever et al. [33] challenge the RACs used for some pesticides in the analysis by Weisner, et al. [31], noting that the RACs were lower than the thresholds initially established by the authorities. This critique, however, did not recognise that RACs, which initially are based largely on manufacturer data, evolve over time as additional information becomes available. Consequently, Weisner, et al. [31] used up-to-date RACs provided by the regulatory body responsible for authorising plant protection products in Germany, where the study was carried out [75]. The analysis highlights that concentrations of chemicals frequently exceed risk thresholds designed to protect the ecological status of European surface waters [see also 76,77]. Pesticides are prominent among those chemicals [69,78].

Relative importance of pesticides as drivers of biodiversity decline

Schriever et al. [33] recognise that biodiversity decline is real, but contest the extent to which pesticides are an important driver. They correctly note that Dirzo et al. [79], as cited in Schäffer et al. [27], does not allow clear conclusions on the relative importance of different drivers of biodiversity decline. However, numerous other studies do highlight the role of pesticides in biodiversity loss [10,e.g. 23,30,32,41,80–83]. Paradoxically, the references cited by Schriever et al. [33] to question the role of pesticides in biodiversity declines do not support their interpretation. Instead, the review by Leenhardt et al. [84] states that pesticide use contributes to biodiversity decline. The synthesis of Rumohr et al. [85], which covers 741 cases from 82 studies, also support the case for a role of pesticides on biodiversity. Although climate and land-use change were identified as dominant drivers, Rumohr et al. [85] also emphasise that most of the underlying studies assessed only one or a few drivers, limiting holistic conclusions on the relative importance across all potential factors. Notably, pesticides were considered in only 27 cases from 15 studies, with quantitative evidence largely lacking. In the small subset of cases where pesticide concentrations were measured (9 cases from 5 studies), pesticides were consistently identified as important drivers (Table S5 in [85]). Such evidence supports, rather than refutes, a role of pesticides in biodiversity decline.

As Schriever et al. [33] correctly point out, biodiversity decline has multifactorial causes, with pesticide exposure being only one of them. Given the ubiquity of multiple stressors in contemporary ecosystems, it remains challenging to quantify the contributions of individual drivers, not least because impacts vary among organism groups, regions and ecosystem types. The most comprehensive evaluation to date is based on a screening of 45,162 studies, 163 of which were suitable to compare the relevance of five highly aggregated drivers: land-use change, resource extraction, pollution, alien species and climate change [12]. Land-use change and pollution emerged as the most and third most important drivers, while the importance of other drivers varied among regions, ecosystems and type of biodiversity response [12]. A further meta-analysis assessing the same five drivers found that pollution had the strongest effects on local biodiversity [86]. Although a targeted analysis on pesticides is lacking, the aggregate driver termed pollution included pesticide contamination. Land-use change typically also involves agricultural expansion and intensification associated with pesticide release. Thus, while the jury is out as to the exact proportion of biodiversity loss attributable to pesticide spread in different settings, given the spatial and temporal complexities of driver pressures and trends, current evidence provides support for a role of pesticide use in biodiversity change at least at local and landscape scales [24].

Widespread pesticide effects on non-target organisms

A recent systematic synthesis of meta-analyses has revealed broad impacts of current-use pesticides at concentrations relevant to field situations in Europe [37]. Both aquatic and terrestrial organisms were affected, including bees, soil invertebrates, aquatic invertebrates, primary producers and microorganisms. In the meta-analyses assessing mortality, growth, reproduction and community-level effects, 39 of 51 cases (i.e. pairs of pesticides and organisms) produced adverse outcomes. Seventeen of them were significantly so, whereas none of the five positive effects reached significance. All seven cases where effect sizes related exclusively to current-use pesticides were negative, four of the relationships being significant. The results indicate that pesticides currently authorised in the EU exhibit adverse effects on non-target populations and communities under real-world conditions, as supported by further evidence detailed below.

To support their case that the current regulatory system is protective, Schriever et al. [33] cite two field studies that found no effects of pesticides on non-target species. However, other studies are available that come to different conclusions. As an example, two meta-analyses are available that provide evidence for the adverse effects of pesticide on pollinators [87,88]. On this topic, Schriever et al. [33] cite an individual study that did not identify pesticide effects on pollinators under field conditions [89]. However, this outcome may have been influenced by the adopted statistical design [90,91], as acknowledged by Schriever et al. [33]. The issues when combined with evidence of adverse effects published by other authors highlight the potential for pesticide exposure to impact pollinator communities.

Evidence for declining bird populations attributed to pesticides [8] was also challenged by Schriever et al. [33]. However, their claim that such a negative trend would be practically identical before and after the introduction of imidacloprid is poorly substantiated in view of the statistically significant trend change found after introduction [8]. That the abundances of insectivorous birds slightly increased after neonicotinoid insecticides were banned is further evidence for a potential impact [92]. An extensive study on 170 bird species monitored across 28 European countries [10], not cited by Schriever et al. [33] identified pesticide and fertiliser use as major drivers of bird declines. However, because these drivers co-occur with structural landscape change under agricultural intensification, their individual effects are difficult to disentangle, and they likely act both independently or interactively.

A comprehensive field study addressing pesticide effects in small agricultural streams found that pesticides were the dominant factor driving changes in ecological indicators for vulnerable insect populations [83]. Schriever et al. [33] suggest that this study insufficiently addressed the potential influence of abiotic factors including stream morphology, catchment characteristics and stream type. However, Figure 1 in Liess et al. [83] shows that stream morphology was explicitly considered in the analysis and associated with several ecological indicators, in line with results of previous studies [93,94]. Schriever et al. [33] refer to a companion study [95] which would suggest that chemicals other than pesticides could explain the observed changes in invertebrate populations. However, Neale et al. [95] assessed responses of mammalian reporter genes, so the identified chemicals differ from those relevant to freshwater insects. Notably, Liess et al. [83] considered the toxicity of metals and urban pollutants, but these were not retained in best-fit models for several ecological indicators, indicating a minor role in insect population change.

Drawing on a paired comparison of laboratory and field studies in soils [96], Schriever et al. [33] assert that current EU risk assessments ensure protection of earthworm populations. Generalising these results, however, could be compromised by the fact that nearly all of the examined studies covered grasslands rather than high-intensity farming systems receiving greater pesticide loads and having higher organic carbon contents than typical agricultural soils, which likely reduced pesticide bioavailability and accelerated dissipation [62,97–99].

Other evidence counter to the contention of Schriever et al. [33] were not cited. One case is a meta-analysis that revealed that fields receiving no or low amounts of pesticides support significantly higher earthworm abundances and biomass than conventionally farmed land [100]. A further example, levels of pesticide residues in 43% of 47 soils sampled across France under various land-uses were classified as medium or high risk for earthworms [52]. Another study found that soil concentrations of 614 pesticides in nearly half (46%) of 188 wheat fields distributed across Europe exhibited high risk levels for earthworms, springtails and other soil species [101]. In a further study, almost 70% of 201 soils screened for 209 pesticides in 10 European countries posed high risks to non-target arthropods such as mites [102], and a simulation of pesticide accumulation resulting from spray series indicated that acceptable regulatory levels for soil fauna were exceeded [57]. A continent-wide study across 373 sites spanning multiple land uses in 26 European countries detected pesticide residues at 70% of sites and identified them as a major driver of taxonomic and functional soil biodiversity patterns [41]. Finally, a re-analysis of a meta-analysis [30] restricted to current-use pesticides in Europe found significant adverse effects of insecticides on soil invertebrate communities [37]. Taken together, this range of large-scale studies clearly suggest that the assumption that EU legislation protects earthworms and other soil fauna in agricultural fields from adverse effects of pesticides in Europe is inconsistent with the wealth of current contrary data.

Multiple causes behind the limitations of current EU regulation

Several factors neglected in regulatory risk assessment at present have been advanced to explain the gap between regulatory intention and the evident realities of widespread pesticide effects [27,28]. The most prominent ones are multiple stressors, indirect effects, and mixtures of chemicals (Fig. 1):

1) Multiple stressors: Evidence is growing that the causes of biodiversity declines are multifactorial [76,83,103–105]. For example, climate warming and extreme events can exacerbate adverse effects of toxicants [106,107]. Schriever et al. [33] raise this issue in terms of the potential of multiple stressors (and variation in environmental conditions) to confound pesticide effects, but fall short of addressing how to account for such interference in pesticide risk assessment. This is a critical oversight since multiple stressors can produce joint effects beyond those of individual stressors acting alone [108]. The fact that ecological responses arise from the combined influence of multiple factors does not negate the contribution of any single factor to the overall effect. Solutions are needed because the complexity of unravelling and managing impacts of stressors such as climate extremes is difficult, but must not prevent their inclusion in risk assessments that are meant to reflect reality.

2) Indirect effects: A prominent feature of biological communities in ecosystems are multiple interactions between species in space and time. Pesticide exposure can modify these interactions and lead to unexpected outcomes caused by the propagation of direct

ecotoxicological effects even to insensitive species that are not directly affected [109–111]. Such cascading effects can also involve abiotic processes. An example case is the slope erosion observed after removal of riparian vegetation by herbicide application, which led to the remobilisation of legacy pesticides deposited in riparian soils into an adjacent lake [112]. Scenarios accounting for any such indirect impacts are poorly reflected even in higher-tier multi-species tests.

3) Chemical mixtures: Pesticide mixtures are ubiquitous in the environment. Reasons are variation in pesticide use at the landscape level, accumulation over time, spray series, tank mixtures and plant protection products containing multiple active substances [52,58,61,113]. Regulatory risk assessment yet only considers plant protection products containing multiple active substances, although mixtures of active substances originating from different products being ecotoxicologically relevant. For example, the cumulative toxicity of multiple pesticide mixtures correlated with the growth and production of bumble bee colonies [113]. In another case, pesticide mixtures reflecting applications of multiple plant protection products at recommended rates had stronger detrimental effects on soil fauna than individual pesticides [30].

The need for post-registration field monitoring and experiments

Detailed field studies are needed to evaluate the extent to which indirect effects, multiple stressors and chemical mixtures contribute to impacts of pesticides beyond those anticipated in standard regulatory risk assessment [114]. The studies could take the form of systematic post-registration field monitoring involving both the quantification of pesticides and the assessment of biological endpoints, including advanced biomolecular (omics) approaches [27,28,115,116] and acoustic and image-based surveys [117–119]. Such monitoring programs best integrate data across levels of ecological organisation and multiple ecosystem compartments [114]. Non-target chemical analysis should be used to detect toxicants beyond the target pesticides and their known metabolites such as trifluoroacetic acid, including additional and potentially confounding toxic effects and unknown metabolites.

The collected data, together with a FAIR repository on pesticide application and related ecotoxicological and fate data, would enhance risk assessment by elucidating mechanisms behind exceeded thresholds and facilitating efforts to refine exposure models. Leveraging advanced biomolecular approaches such as transcriptomics would also help to understand mechanisms behind observed pesticide effects and distinguish these effects from confounding factors [e.g. 120–123]. In addition, the issue of confounding factors could be addressed through smart gradient designs, replicated field experiments and the integration of standardised populations or communities (e.g. standardised bumble bee colonies) in effect monitoring [113,e.g. 124]. These approaches would counter the concern of Schriever et al. [33] that confounding factors in field studies preclude causal attribution.

Balancing pesticide reduction with food security and land management

Schriever et al. [33] have an important point that pesticide use needs to be evaluated in the context of food security and land management. Although no panacea, a substantial increase in the market share of organically grown staples and other produce would help mitigate this conflict [125]. While organic farming also relies on pesticides, an analysis of 9000 organic fields found that the probability of pesticide use was approximately 30% lower than in conventional agriculture, and the overall toxicity to fish approximately halved [126]. Another survey of 100 fields revealed that organically farmed land receives only half the number of active substances applied in conventional farming, with toxicity levels being about an order of magnitude lower [62]. Lastly, 55% of the active substances exclusively authorised for conventional agriculture carry health or environmental hazard statements, compared to only 3% of the substances permitted for organic agriculture [127].

Schriever et al. [33] point out that landscape context influences impacts of pesticides. Indeed, semi-natural habitat can buffer against pesticide effects as, for instance, reduced cropland dominance and rich floral resources weaken impacts on bumble bee colony growth and reproduction [113]. Exposure to neonicotinoids reduced wild bee development by 69% in mesocosms planted with oilseed rape in monoculture, but floral resources offset this strong effect in mesocosms with diverse wild-plant communities [128]. In this context, Schriever et al. [33] cite Redhead et al. [129] as evidence that allocating only 1–5% of farmland to conservation measures is sufficient to reconcile modern agriculture with biodiversity. This study actually shows that adding this level of high-quality habitat can improve bird and butterfly trends relative to comparable landscapes without such measures. It does not show that such low proportions alone can halt farmland biodiversity declines, and other studies suggest that approximately 20% semi-natural area is required to maintain biodiversity and ecosystem services in agricultural landscapes [130,131]. However, a global analysis of 681 crop fields spread across three continents revealed that while semi-natural habitats promote wild bee abundance and diversity, they do not fully mitigate pesticide-driven losses, suggesting that habitat restoration alone is insufficient to fully recover pollinator communities [103]. Therefore, an integrative approach addressing both pesticide reduction and habitat restoration is likely to yield the greatest gains.

Schriever et al. [33] argue that change to pesticide use could substantially reduce land-use efficiency and externalise environmental damage. The concerns are not entirely unjustified. However, practices promoting biodiversity through a suite of regenerative farming principles, including crop diversification, habitat creation, supported natural pest management, polycultures, and agroecological intensification, can substantially alleviate pesticide dependency without proportional yield loss [132,133]. In principle, such measures can be adopted worldwide [134]. Diversifying the agricultural landscape through reducing field size can multiply species richness on fields without reducing production levels [130,135]. Economically, approximately half of the income of farmers in EU Member States derives from subsidies. This means management priorities are fundamentally policy-determined [136,137] and suggests much scope for policy changes to guide agricultural practices. Finally, animal feed consumes at least one-third of global plant production, and an additional third is lost through food waste [138]. A reduction in meat consumption and food waste would therefore proportionally relieve pressure on agricultural land arising from yield declines under reduced pesticide use [139]. Policy interventions such as adjusting the taxation or subsidisation of plant- and animal-based agricultural products offer tractable ways to nudge consumption towards patterns that support ecosystem protection and recovery while safeguarding food security [140]. Clearly, multiple policy options exist to balance environmental and human health protection with food security and farmer subsistence.

Conclusions

Our critical appraisal of the evaluation of European pesticide risk assessment provided by Schriever et al. [33] reveals a range of meta-analyses and large-scale studies that provide evidence counter to these authors main claim that the current EU regulatory system is highly advanced and protective. Empirical evidence consistently demonstrates that current EU legislation suffers from substantial gaps between regulatory evaluations and real-world environmental outcomes. The discrepancies, which manifest in both exposure and effects assessment, reflect fundamental structural issues in how the EU regulatory framework currently conceptualises pesticide risk: through single-substance, single-stressor, heavily laboratory-based approaches that fall short of capturing ecosystem complexity (Fig. 1).

Closing the gap requires research to understand pesticide fate, behaviour and effects under real-world scenarios to support evidence-based improvements to current frameworks and their integration into evolving policy. This endeavour spans multiple domains and involves (i) expanding chemical and biological post-registration monitoring and experiments, both in soil and water, (ii) refining exposure prediction models so as to account for repeated applications and variable environmental conditions, (iii) addressing the joint toxicity of active substances applied in close proximity (time and space), (iv) considering the influence of other stressors (e.g. habitat degradation, invasive species, weather extremes) acting in concert with pesticides, (v) developing an assessment framework explicitly addressing community- and ecosystem-level effects which take species interactions and landscape context into account, and (vi) improving the interlinkage between major EU pieces of regulation.

Beyond technical improvements, regulatory reform should create scope for continuously integrating advances in the understanding of pesticide fate and effects. The yardstick guiding agricultural policy overall must be to minimise pesticide dependency through integrative strategies. Key to this objective is the combination of pesticide use reduction with habitat restoration, accompanied by incentives encouraging a sustainable agricultural production which ensures global food security and farmer subsistence.

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Figure 1 was created by Susanne Berger from Visual Science Communication.

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