Impacts of agricultural land use types on the biodiversity and health of river ecosystems: A large-scale analysis

Auswirkungen landwirtschaftlicher Nutzungstypen auf die Biodiversität und Gesundheit von Fließgewässerökosystemen: Eine großskalige Analyse

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1 Summary

1.1 English Summary

Biodiversity and the health of freshwater ecosystems is strongly impaired by human activities, compromising the stability of these ecosystems and the ecosystem services they provide. Global and European efforts to halt the biodiversity decline and protect ecosystem health were not very successful, especially for rivers, so that for less than ten percent of the German rivers good ecological status was reached in 2021. Present-day agriculture has been identified as the main driver for this deterioration, as evident from a multitude of studies. However, the agricultural effects differ between the organism groups and depending on environmental conditions like soil and climatic conditions. Moreover, and most importantly, agriculture is not uniform. The specific agricultural types and practices differ between regions, which in turn leads to differences in the intensity of agrochemical usage as suggested by many small-scale studies. Consequently, the magnitude of agricultural effects on biodiversity and health of river ecosystems most probably depends on agricultural types and practices and differs between regions. For the effective mitigation of these negative effects, several knowledge gaps need to be closed, which were addressed in six chapters, shortly described in the following.

First, the current knowledge on the effect of agriculture on river biota was summarized and analysed in a meta-analysis (Schürings et al., 2022). According to this meta-analysis described in the first chapter, agriculture has an overall medium to high negative effect on river biota, and results indicate that the effects of agriculture differ between agricultural types, practices, the organism groups, and biological metrics considered. Second, a pan-European dataset was used to establish an agricultural typology, based on agricultural production and agriculture-related freshwater pressure by nutrients, pesticides, water abstraction and hydromorphological alterations (Schürings et al., 2023). This chapter identified how agricultural types differ in their pressures exerted on freshwaters and shows that accounting for agricultural pressure intensity nearly doubles the correlation with the ecological status. Third, the effects of different agricultural types on the ecological status according to the EU Water Framework Directive (WFD) were investigated, using high resolution German-wide land use data, distinguishing between different crop types (Schürings et al., 2024a). The effects on the ecological status clearly differed between crop types, which typically are associated with different agrochemical application rates. Macroinvertebrates and macrophytes were most strongly affected by pesticide application intensive crops and diatoms were most affected by nutrient intensive crops. Fourth, the results presented in Markert et al. (2023) provided evidence that urban areas and different

agricultural crop types with typical agrochemical application rates are indeed related to the micropollutant concentrations monitored in rivers, which often exceeded Environmental Quality Standards. Fifth, crop type-specific differences in agrochemical application rates reported in literature were used to generate an agricultural intensity index (Schürings et al., 2024b). This index improved the correlative strength between present-day agriculture and the ecological status with most pronounced relations for macroinvertebrates in small mountain streams. Sixth, experiences from implementing environmental legislations like the WFD were used to advice for a successful implementation of the EU Nature Restoration Law (Hering et al., 2023). This final chapter highlights that joining restoration efforts with a shift to more sustainable agriculture, whose importance is reasoned in the previous chapters, would offer unprecedented opportunities for successful protection of ecosystem health.

In conclusion, this thesis provides overwhelming evidence for the negative effects of present-day agriculture on river biota, portraying influencing factors and highlighting strong relationships between agricultural effects on river biota and agrochemical application, particularly of pesticides. Therefore, to mitigate these effects, a transition of present-day agriculture to more sustainable practices, such as organic farming or agroecology is of vital importance. Such a transition would be beneficial both for the future viability of agriculture itself but also for the protection and restoration of healthy ecosystems, including the successful implementation of the European environmental legislation such as the Nature Restoration Law.

1.2 German Summary

Die Artenvielfalt und Gesundheit von Fließgewässerökosystemen sind stark durch menschliche Aktivitäten beeinträchtigt, was die Stabilität dieser Ökosysteme und die von ihnen erbrachten Ökosystemdienstleistungen stark gefährdet. Sowohl globale als auch europäische Bemühungen den Rückgang der Artenvielfalt einzudämmen und für den Schutz der Ökosysteme waren nicht sehr erfolgreich, insbesondere für Fließgewässer, so dass im Jahr 2021 nur weniger als zehn Prozent der deutschen Flüsse einen guten ökologischen Zustand erreichten. Die heutige Landwirtschaft wurde als Hauptursache für diese Verschlechterung identifiziert wie aus zahlreichen Studien hervorgeht. Jedoch unterscheiden sich die Auswirkungen der Landwirtschaft je nach Organismen und Umweltbedingungen wie Boden- und klimatischen Verhältnissen. Darüber hinaus ist die Landwirtschaft nicht einheitlich. Die spezifischen landwirtschaftlichen Typen und Praktiken unterscheiden sich zwischen Regionen, was wiederum zu Unterschieden in der Intensität und des Einsatzes von Agrochemikalien führt, wie es viele Fallstudien nahelegen. Zur wirksamen Minderung dieser negativen Auswirkungen müssen mehrere Wissenslücken geschlossen werden, deren Bearbeitung in sechs Kapiteln im Folgenden kurz beschrieben wird.

Im ersten Kapitel wurde das aktuelle Wissen über den Einfluss der Landwirtschaft auf die Flussbiota in einer Metaanalyse zusammengefasst und analysiert (Schürings et al., 2022). Gemäß dieser Metaanalyse hat die Landwirtschaft insgesamt eine mittlere bis hohe negative Wirkung auf die Flussbiota, und die Ergebnisse deuten darauf hin, dass sich die Auswirkungen der Landwirtschaft zwischen landwirtschaftlichen Typen, Praktiken, den betrachteten Organismengruppen, sowie biologischen Metriken unterscheiden. Zweitens wurde ein pan-europäischer Datensatz verwendet, um eine landwirtschaftliche Typologie zu erstellen, basierend auf landwirtschaftlicher Produktion und landwirtschaftsbedingten Belastungen auf Süßgewässer durch Nährstoffe, Pestizide, Wasserentnahme und hydromorphologischen Veränderungen (Schürings et al., 2023). In diesem Kapitel wurde identifiziert, wie sich landwirtschaftliche Typen in ihren Belastungsmustern auf Süßwasser unterscheiden, und es zeigt sich, dass die Berücksichtigung der Intensität landwirtschaftlichen Belastungen die Korrelation mit dem ökologischen Zustand nahezu verdoppelt. Drittens wurden die Auswirkungen verschiedener landwirtschaftlicher Typen auf den ökologischen Zustand gemäß der EU-Wasserrahmenrichtlinie (WRRL) untersucht, unter Verwendung hochauflösender landesweiter Landnutzungsdaten, mit Unterscheidung zwischen verschiedenen Anbaufrüchten (Schürings et al., 2024a). Die Auswirkungen auf den ökologischen Zustand unterschieden sich deutlich zwischen den Anbaufrüchten, die in der Regel mit unterschiedlichen Anwendungsraten

von Agrarchemikalien in Verbindung gebracht werden. Makroinvertebraten und Makrophyten wurden am stärksten von Anbaufrüchten mit intensivem Pestizideinsatz belastet, während Diatomeen hauptsächlich von nährstoffintensiven Anbaufrüchten beeinträchtigt waren. Viertens zeigten die Ergebnisse in Markert et al. (2023), das städtische Gebiete und verschiedene landwirtschaftliche Kulturpflanzentypen mit typischen Anwendungsraten von Agrochemikalien tatsächlich mit den in Flüssen beobachten Konzentrationen von Mikroschadstoffen in Verbindung stehen, die oft Umweltqualitätsstandards überschritten. Fünftens wurden, sich zwischen den Anbaufrüchten unterscheidende Anwendungsraten von Agrarchemikalien, verwendet, um einen landwirtschaftlichen Intensitätsindex zu erstellen (Schürings et al., 2024b). Dieser Index verbesserte die Korrelationsstärke zwischen der heutigen Landwirtschaft und dem ökologischen Zustand, wobei die deutlichsten Beziehungen zu Makroinvertebraten in kleinen Gewässern im Mittelgebirge festgestellt wurden. Sechstens wurden Erfahrungen aus der Umsetzung von Umweltgesetzgebungen wie der WRRL genutzt, um Empfehlungen für eine erfolgreiche Umsetzung und Implementierung des EU-Naturrestaurationsgesetzes (Nature Restoration Law) zu geben (Hering et al., 2023). Dieses abschließende Kapitel betont, dass die Verknüpfung von Restaurationsbemühungen mit einer Umstellung auf nachhaltigere Landwirtschaft, deren Bedeutung in den vorherigen Kapiteln begründet ist, beispiellose Chancen für einen erfolgreichen Schutz der Ökosystemfunktionen bieten würde.

Zusammenfassend liefert diese Arbeit überwältigende Hinweise für die negativen Auswirkungen von heutiger Landwirtschaft auf die Flussbiota, und identifiziert wichtige Einflussfaktoren, wie die Zusammenhänge zwischen den Auswirkungen der Landwirtschaft und dem Einsatz von Agrochemikalien, insbesondere Pestiziden. Daher ist zur Minderung dieser Auswirkungen ein Übergang von der heutigen Landwirtschaft zu nachhaltigeren Praktiken wie ökologischen Landwirtschaft oder Agrarökologie von entscheidender Bedeutung. Ein solcher Übergang wäre sowohl für die Zukunft von Landwirtschaft selbst als auch für den Schutz und die Wiederherstellung gesunder Ökosysteme von großer Bedeutung, einschließlich der erfolgreichen Umsetzung europäischer Umweltgesetzgebung wie des Naturrestaurationsgesetzes.

2 General introduction

2.1 Biodiversity and ecosystem health in crisis

Biodiversity across the planet is widely acknowledged as nature´s inherent insurance policy, playing a pivotal role for the health of ecosystems, including the maintenance of stability and efficiency of ecological processes (Loreau et al., 2021). It encompasses a wide range of life forms, spanning from microorganisms to animals and plants (Wilson, 1988). These diverse organisms play distinctive roles, collectively contributing to the preservation of ecological stability, enhancement of resilience, and assurance of the sustenance of all life on earth (Peterson et al., 1998; Donohue et al., 2016). Healthy ecosystems facilitate a multitude of ecological processes, including flow of energy (Barnes et al., 2018), cycling of nutrients (Zou et al., 2016), population dynamics (Kortsch et al., 2021), succession (Mori et al., 2017), and biogeochemical cycles (Kristensen et al., 2014). Ecosystem health stands as a cornerstone, not solely for the survival of the inhabiting organisms, whose intrinsic value necessitates their protection (Taylor, 2011), but is also essential for human well-being (Balvanera et al., 2006). Healthy and functioning ecosystems offer a wide array of benefits to humans, often referred to as ecosystem services (Benayas et al., 2009; Spangenberg et al., 2014). These services encompass provisioning services, like the provision of food, energy, and clean water (Sahle et al., 2019), and supporting services such as pollination (Schulp et al., 2014), soil quality preservation (Drobnik et al., 2018), waste remediation (Watson et al., 2016). Additionally, they encompass regulation services that maintain balance and stability, such as water purification (Piaggio & Siikamäki, 2021), climate regulation (Ma et al., 2019), and carbon sequestration (Beaumont et al., 2014) and cultural services like educational and recreational services (Hernández-Morcillo et al., 2013). Notably, the economic value of benefits derived from biodiverse natural ecosystems can far exceed their maintenance costs, potentially being 100 times greater (Balmford et al., 2002; Brink, 2009). The monetary value of ecosystems has been estimated to range between 125 trillion $US (Costanza et al. 2014) and 180 trillion $US per year (Boumans et al., 2002), far exceeding the global gross national product (GNP).

However, the world´s ecosystems face an imminent threat due to escalating human activities, resulting in biodiversity decline with rapidly increasing speed over the last decades. This concerning trajectory is underscored by the ongoing decline in nearly half of the species populations (Finn et al., 2023). On that note, Hallmann et al. (2017) provide a striking illustration of this decline, demonstrating a loss of more than 75 percent of flying insect biomass in protected areas over a 27-year span. The gravity of this situation is emphasized by comparing human-induced extinctions to the natural catastrophic events that caused the five most recent global mass extinctions in the past 500 million years (Tilman et al., 2017). Consequently, the health of the world's ecosystems is highly endangered (Valiente-Banuet & Verdú, 2013). The impact of human activities on biodiversity and ecosystem health spans various scales. More than 75 percent of the terrestrial land including freshwaters has been altered (Elis & Ramankutty, 2008), primarily due to deforestation (Pimm & Askins, 1995), urbanization (Concepción et al., 2015) and expansion and intensification of agriculture (Dudly & Alexander, 2017; De Graaff et al., 2019). These alterations introduce an array of stressors to the ecosystems, including pollution (Dumont et al., 2012; Toro et al., 2016), habitat fragmentation (Mullu, 2016) and morphological alterations (Fernandes et al., 2020), as well as overhunting (Peres et al., 2016) and overfishing (Jackson et al., 2001). Furthermore, the intrusion of partly invasive neobiota through travel and transport (Mollot et al., 2017), coupled with human-induced climate change leading to rising temperatures (Warren et al., 2011), and heightened frequency in extreme weather events (Ummenhofer & Meehl, 2017), has amplified the degradation of Earth´s ecosystems. This degradation extends to the terrestrial realm, affecting mammals (Ma, 1989; Shore & Douben, 1994; Poeta et al., 2017), insects (Skaldina & Sorvari, 2019; Fengolio et al., 2021), birds (Stanton et al., 2018; Wang et al., 2021), and soil biota (Veresoglou et al., 2015; Ullah et al, 2023). Similarly, organisms inhabiting the marine ecosystem (Fisk et al., 2005; Kakuschke & Prange, 2007; Leduc et al., 2013) and freshwaters (Roy et al., 2003; Leduc et al., 2013; Lemm et al., 2021; Wolfram et al., 2021) are strongly affected by the impact of human activities. In fact, freshwater ecosystems are even considered the most endangered ecosystem worldwide, declining faster than all other ecosystems (He et al., 2019; Almond et al., 2020). This underscores their central importance within the context of this thesis.

2.2 Current legislations fail to protect biodiversity and ecosystem health

To address this biodiversity crisis and safeguard ecosystem health and functioning, several global legislative agreements and treaties have been established. Examples include the Ramsar Convention on Wetlands in 1971 (Ramsar Convention, 1971), the World Heritage Convention in 1972 (Unesco, 1972), the Convention on International Trade in Endangered Species of Wild Fauna and Flora in 1973 (CITES, 1973), the Convention on Biological Diversity in 1992 (CBD, 1992) and the Sustainable Development Goals in 2015 (UNEP, 2015). Also, at the European scale several pieces of legislation have been introduced such as the Biodiversity Strategy, introduced in 1998 (European Commission, 1998), the Habitats Directive, introduced in 1992

(EEA, 1992), the Birds Directive, introduced in 2009 (EEA, 2009), the Forest Strategy, introduced in 2013 (European Commission, 2013a), the Marine Strategy Framework Directive, introduced in 2008 (EEA, 2008) and the Water Framework Directive (WFD), introduced in 2000 (European Commission, 2000). The WFD particularly targets the most endangered ecosystems of freshwaters, primarily focused on in this thesis. Notably, the WFD incorporates the world´s most intensive biological monitoring programme (EEA, 2018a), with more than 100,000 river bodies regularly monitored for the ecological status - a quality assessment for the structure and the health of surface water ecosystems (European Commission, 2000). The ecological status, a biodiversity-related criteria, reflects the response of the biological quality elements (BQEs) macrophytes, phytoplankton, phytobenthos, macroinvertebrates, and fish (supported by physicochemical and hydromorphological quality elements) to various stressors such as pollution, morphological alterations, or altered hydrological conditions (Hering, et al., 2006; EEA, 2018a).

However, despite these comprehensive efforts, until date the various directives were not very successful in halting biodiversity decline and protecting ecosystem health (Butchart et al., 2010; Tilman et al., 2017; Burns et al., 2021; Outhwaite et al., 2022), even in the face of some freshwater biodiversity gains in the 1990s and 2000s (Haase et al., 2023). The reasons for the legislation challenge encompass inadequate funding, limited human resources, suboptimal planning procedures and insufficient implementation capacities (Kruk et al., 2010; European Commission, 2016; Dieter et al., 2020; Pe'er et al., 2020), coupled with knowledge gaps (Mastrángelo et al., 2019; Xu et al., 2021) and conflicting interests that prioritize short-term economic gains (Marshall et al., 2007; Cortina-Segarra et al., 2021). For example, the WFD aims at reaching good ecological status in all European rivers by 2027, however the measures implemented were not very effective until date, with less than 40 percent of the European rivers in good ecological status in 2015 (EEA, 2018a). In Germany even more than 90 percent of the rivers still failed to reach good ecological status in 2021 (UBA, 2022), frequently attributed to agricultural land use in the catchment (Bieroza et al., 2021; Weisner et al., 2022). Despite efforts to incorporate environmental goals into agricultural legislation, such as the inclusion of Agri-Environmental Measures in the Common Agricultural Policy in 1992 (EEA, 1992), it appears that agricultural legislation still struggles to promote environmentally friendly farming practices. This issue may be linked to conflicts of interests, particularly short-term economic goals of the agrochemical industry and large-scale farmers (Pe'er, 2019; Navarro & López-Bao, 2019), obstructing ambitious legislation such as the recently rejected sustainable use regulation (European Commission, 2022a). As a result, present-day agriculture, which is frequently identified as the key driver for the deterioration of freshwater ecosystems, continues to have a strong impact on global biodiversity (Tilman et al., 2001; Stehle & Schulz, 2015; Wolfram et al., 2021).

2.3 Agriculture: The dilemma of growth

Agriculture has a history of more than 10,000 years and started with livestock production (Hartung, 2013) and the domestication of legumes and cereals, primarily in the Fertile Crescent stretching from modern-day Iraq to Israel (Salamini et al., 2002). This pivotal shift enabled the accumulation of surplus food and consequently the formation of permanent settlements (Lev-Yadun et al., 2000). Over the first 9,000 years, agriculture exhibited gradual expansion coupled with improvements in agricultural techniques. The development of all advanced civilization became tightly intertwined with superiority in agricultural practices productivity and surplus of food (Barker, 2006). Notably, Egypt refined irrigation systems (Mahmoud et al., 2019), while Asian advanced civilizations benefited from productive rice cultivation (Fuller, 2011). In Europe, the Roman Empire introduced farming methods such as crop rotation and animal-powered plows (Pleket, 1993). A notable change in agricultural practices transpired during the medieval period, roughly between the 10th and 16th century. This period witnessed the introduction of the heavy plow, allowing the cultivation of previously unworkable soils, as well as refined crop rotation systems with a transition from 2-course to 3-course rotations and the domestication of new crops enhancing yields (Evans, 2003; Andersen et al., 2016). The harnessing of wind and waterpower for milling and irrigation further amplified agricultural productivity (Lucas, 2005).

While agricultural growth until the 16th century was mainly based on agricultural land expansion, the subsequent agrarian revolution, and the following industrial revolution, spanning from the 16th to the 19th century, marked the shift towards agricultural intensification (Grantham, 1989; Pretty, 1991; Wallis et al., 2018). Scientific agricultural research emerged, accompanied by an array of technological innovations, such as increased availability of fertilizers and enhanced livestock breeds. The introduction of steam-powered machinery for tillage and threshing further increased food production, while drastically reducing labor demands (Evans, 1998). While the global food production further increased over time, the most significant surge at the global scale was observed in course of the green revolution between 1960 and 2000. In this period high-yielding crop varieties were introduced, mainly grown in monocultures (Everson & Gollin, 2003) with high application rates of synthetic fertilizers and pesticides (Pimentel, 1996). This led to an unprecedented increase of the global food production by a factor of more than two within four decades (Evenson & Gollin, 2003), so that agriculture now successfully feeds billions of people (Borlaug, 2002). Since then, agriculture has continued to intensify to meet the growing food demand driven by further population growth, and due to geopolitical crises (Sohag et al., 2023). Also, the purpose of agricultural production has changed drastically. While agriculture solely targeted to feed humans and ensure food security in the past, now the major part of agricultural land is used to feed animals, coinciding with the drastic increase of meat consumption (Godfray et al., 2010), and a substantial part of agricultural land is used for energy production from biomass (Reilly & Paltsev, 2009). These trends are likely accompanied by further increases in agrochemical usage and decreased habitat heterogeneity (Benton et al., 2003; Gibbs et al. 2009; Geiger et al., 2010; Meehan et al., 2011).

However, this intensification and expansion of agriculture comes at a high cost. Over the course of 10,000 years, agriculture has transformed landscapes through activities like deforestation (Williams, 2003; Salinger, 2007). Especially recent cropland expansions have only marginally increased the yields, while inflicting considerable harm to biodiversity (Lark et al., 2020). Moreover, the adverse effects of agricultural intensification are conspicuous. Intensive agriculture compromises human health, particularly for farmers with skin contact to agrochemicals (Kumar et al., 2012; Elahi et al., 2019), but also consumers through dietary changes leading to obesity and related diseases (Horrigan et al., 2002; Brownell & Horgen, 2004). However, the environment is even more strongly impaired by the present-day agriculture. The massive increase in agricultural production, particularly livestock production contributes to significant carbon dioxide emissions (Snyder et al., 2009) and habitat loss caused by deforestation (Bodo et al., 2020). In fact, the dietary shift of humans to more meat consumption results in three to four-fold higher environmental burden as compared to a vegan diet (Scarborough et al., 2023). Pollution from pesticides and nutrients adversely impacts numerous terrestrial and aquatic organisms (Reinert et al., 2002; van Meter et al., 2019), collectively driving rapid biodiversity decline (Dudley & Alexander, 2017). Ultimately, the present-day agriculture even jeopardizes its own existence, exploiting and eroding soils (Pimentel & Burgess, 2013), while widespread monocultures become increasingly vulnerable to environmental changes such as climate change, due to genetic erosion, which narrows the spectrum of high-yielding crop varieties (Khoury et al., 2022).

2.4 Agricultural effects on river biota

Rivers are affected by a multitude of stressors (Vörösmarty et al., 2010), with land use among the major drivers, which is often referred to as substantial cause for freshwater deterioration (Hughes & Vadas, 2021; Lemm et al., 2021; Chen et al., 2023). Addressing agricultural burden for river ecosystems is an especially complex task as compared to dealing with explicit point source pollution (O´Shea, 2002), often originating from urban and industrial sources, and hydromorphological alterations, which have more frequently become focal points for restoration efforts (Haase et al., 2013). In fact, the major challenge lies in addressing the diffuse pollution caused by agricultural practices in the catchment (Burkart, 2007; Collings & McGonigle, 2008; De Vito et al., 2020), especially given the vast extent of agricultural land use covering nearly 50 percent of the world´s habitable land (Ritchie & Roser, 2019).

Agricultural activities have been shown to cause nutrient runoff into rivers (Withers & Lord, 2002), increasing the primary production and fluctuations and temporally reduction of oxygen levels (Sabater et al., 2000, Nijboer & Verdonschot, 2004; Desmet et al., 2011). This ultimately causes eutrophication (Almeida et al., 2018) with direct effects on diatoms and macrophytes (O´Hare et al., 2018) and indirect effects on macroinvertebrates and fish (Hering et al., 2006). Nutrient surplus favors species that feed on algae detritus and dead plants, while oxygen depletion leads to declines in sensitive taxa (Lange, 2014), owing to interspecific competition and secondary saprobic load (Gieswein et al., 2017). However, a moderate nutrient influx can have positive impacts on macroinvertebrate abundance and richness in nutrient-deficient rivers due to elevated overall productivity (Matthai et al, 2010; Piggot et al., 2012; Piggott et al., 2015). River biota, especially macroinvertebrates and fish are also negatively influenced by the influx of fine sediments, which obstructs the interstitial spaces of natural stream beds muddying rivers (Larsen & Ormerod, 2010; Jones et al., 2012; Foucher et al., 2015; Gieswein et al., 2019). This sediment influx disrupts the exchange of oxygen between the free-flowing and interstitial water, compromising habitat conditions for various species (Wagenhoff et al., 2012, Burdon et al., 2013, Elbrecht et al., 2016).

Alongside nutrients and fine sediment also pesticides drift into rivers, especially after heavy rainfall events (Bereswill et al., 2012), with strong toxic effects on river organisms (Liess et al., 2008). Depending on pesticide application rates (Andert et al., 2015) and the presence of riparian vegetation (Palt et al., 2023), varying quantities of pesticides enter rivers, exerting detrimental effects on river biota. However, the severity of impact varies among pesticide groups (herbicides, fungicides, and insecticides) and different organism groups. For instance, macrophytes and diatoms are primarily impacted by herbicides like Atrazine, Hexazinone, Metazachlor and Iofensulforon (Solomon et al., 1996, Mohr et al., 2007; Fernández-Naveira et al., 2016; Ribeiro et al., 2019; Bighiu et al., 2020). Reports concerning the effects of fungicides and insecticides on primary producers are relatively scarce (Stang et al., 2016). Herbicides have been mainly shown to indirectly impact macroinvertebrates, based on differences in vegetation (Prosser et al., 2016; Pleasants & Oberhausner, 2012). Conversely, fungicides and particularly insecticides exert strong direct negative effects on macroinvertebrates (Wernecke et al., 2019), particularly shown for neonicotinoids and pyrethroids (Anderson et al. 2015; Wurzel, 2020). While fish rarely experience lethal pesticide effects (Schäfer, et al., 2011; Nowell et al., 2018), sublethal chronic effects are documented, especially following heavy rain fall events (Schäfer et al., 2012; Belenguer et al., 2014). Consequently, while all river organisms are affected by pesticides, the effect appears especially pronounced for macroinvertebrates, coinciding findings of Liess et al. (2021), identifying pesticides to be the most relevant stressor for sensitive macroinvertebrates.

Beyond pollution stress, agriculture also influences river biota through stressors originating from competition of land between agriculture and natural river dynamics. Agriculturally induced river straightening increased flow velocity and removal of riparian vegetation deprives river from shading, increasing water temperature (Feld & Hering, 2007; Haidekker & Hering, 2008). Intensive river management further impairs river organisms (Baattrup-Pedersen et al., 2018; Baczyk et al., 2018) with negative effects on macrophytes, diatoms, macroinvertebrates, and fish. Additionally, agricultural water abstraction burdens river ecosystems (Bolpagni and Piotti, 2016, Sabater et al., 2018) by reducing dilution capacity, and causing decreasing groundwater levels and desiccation events (Foster & Gustodio, 2019; Romero et al., 2019).

Agriculture, however, is anything but uniform. The effects of agricultural practices not only differ between organism groups but also strongly depend on the agricultural practices. Those agricultural practices vary depending on climate (Metzger et al., 2005; Reidsma et al., 2007), socioeconomic conditions (Kuemmerle et al., 2008), and soil types (Johnston et al, 2009), as well as on the individual farmers and farm size (Ricciardi et al., 2021). Agricultural practices particularly differ between crop types (Hénault et al., 1998), spatial configurations of agricultural land, such as between mosaic farming and monoculture farming (Figuerola et al., 2015) and the individual cultivation intensity, reflected in agrochemical application rates (Britz & Witzke, 2014; Andert et al., 2015).

A multitude of small-scale studies underscored the strong disparities in effects across different agricultural types and intensities (Sluydts et al., 2009; Weijters et al., 2009; Rega et al., 2020). Distinct crop types differ in the application of pesticides and nutrients on fields (Dachbrodt-Saaydeh et al., 2021; Britz & Witzke, 2014), and several studies showed differential intensity and crop type-specific effects (Wasson et al., 2010; Bereswill et al., 2012; Wang et al., 2013; Abdi et al., 2021). However, until date most large-scale studies primarily considered the sheer percentage of agriculture or the distinction between arable land and grasslands (Del Tánago et al., 2012; Feld et al., 2016; Davis et al., 2022) to investigate agricultural effects, likely because high-resolution land use data, differentiating between crop types, became available only recently (Griffiths et al., 2019; Blickensdörfer et al., 2022). Consequently, in the light of the discernable differences between crop types and management intensities found by many small-scale studies as shown above, there is an urgent need to account for these differences on a larger scale. Such an approach can yield pivotal evidence for legislation, especially concerning the renewal of the post-2027 Common Agricultural Policy, ensuring a balance between feeding humanity and protecting biodiversity and ecosystem health.

2.5 Research motivation

As demonstrated above, compelling evidence suggests detrimental effects of present-day agriculture on river ecosystems, the most endangered ecosystems worldwide, and for effective mitigation of these negative impacts, several knowledge gaps need to be addressed. To identify factors influencing agricultural impacts on river biota, the agricultural effects reported in literature should be assessed and the influencing factors on these effects, as well as knowledge gaps should be compiled to be answered in the following. As reasoned above, agriculture is not uniform, but differing strongly in types, practices, and intensities, wherefore deriving an agricultural typology with relations to freshwater impacts can improve understanding on pathways of agricultural stress. Additionally, in depth investigation of the differing effects between agricultural crop types can shed light on the cause-effect relations between agriculture and river biota and can help to associate the crop type-specific most relevant stressors, which in turn enables mitigation measures. To increase evidence for the crop type-specificity in exerted freshwater stress, relating land use types with micropollutants monitored in the river bodies can be helpful to distinguish agricultural sources from other sources such as urban areas. This allows also to directly link reported crop-specific pesticide applications with the pesticides monitored in rivers. Then, investigating the organism- and stream type-specific effects of agricultural cultivation intensity of different crop types, i.e., based on different application rates of agrochemicals, can offer crucial insight for management and legislation. Lastly, the gathered knowledge can be used to advise the implementation of the European Union´s Nature Restoration Law to halt biodiversity decline and allow for nature restoration in a continent predominantly covered by agriculture. Against this background, this thesis aimed at:

(1) Compiling the current knowledge of agricultural effects on river biota to analyse if agricultural effects are consistently negative at a global scale and identifying influencing factors as well as knowledge gaps.

(2) Establishing a pan-European agricultural typology with relations to production intensity and pressures exerted on rivers to identify regions of similar agricultural practices, potentially better explaining relations between agriculture and the ecological status.

(3) Comparing agriculture with forests and urban areas in their effect on the ecological status and analysing the effects of different agricultural crop types on macroinvertebrates, macrophytes and diatoms at a German-wide scale.

(4) Relating agricultural and urban land use types with river micropollutants monitored under the Water Framework Directive to examine pathways of pesticides industrial chemicals and pharmaceuticals and identify Environmental Quality Standards exceedances and crop-specific pesticide application, monitored concentration concurrences.

(5) Identifying the potential improvement of the relations between agriculture and the ecological status when accounting for agricultural intensity based on crop-specific pesticide and nutrient application rates across different stream type groups and identifying the most important agricultural stressors for macroinvertebrates, macrophytes and diatoms.

(6) Drawing from experiences from implementing legislation like the WFD to advise on how to effectively halt biodiversity decline and facilitate nature restoration with a successful implementation of the EU Nature Restoration Law and outlining benefits from joint efforts with agricultural transition in a continent predominantly covered by agricultural land use.

3 Published and submitted articles

An assemblage of six separate articles thoroughly explored the objectives of this thesis. They have been either already published, accepted, or have been submitted for publication, as specified below:

Chapter 1: Schürings, C., Feld, C. K., Kail, J., & Hering, D. (2022). Effects of agricultural land use on river biota: a meta-analysis. *Environmental Sciences Europe, 34*(1), 124.

Chapter 2: Schürings, C., Globevnik, L., Lemm, J. U., Psomas, A., Snoj, L., Hering, D., & Birk, S. (2023). River ecological status shaped by agricultural land use intensity across Europe. Manuscript submitted for publication in *Water Research* on September 19th, 2023.

Chapter 3: Schürings, C., Kail, J., Kaisjer, W., & Hering, D. (2024a). Effects of agriculture on river biota differ between crop types and organism groups. *Science of the Total Environment*, *912*, 168825.

Chapter 4: Markert, N., Schürings, C., & Feld, C. K. (2023). Water Framework Directive micropollutant monitoring mirrors catchment land use: Agricultural and urban sources revealed. Manuscript submitted for publication in *Science of the Total Environment* on November 30th, 2023.

Chapter 5: Schürings, C., Hering, D., Kaijser W., & Kail, J. (2024b). Assessment of cultivation intensity can improve the correlative strengths between agriculture and the ecological status in rivers across Germany. *Agriculture, Ecosystems and Environment*, *361*, 108818.

Chapter 6: Hering, D., Schürings, C., Wenskus, F., Blackstock, K., Borja, A., Birk, S., Bullock, C., Carvalho, L., Dagher-Harrat, M. B.,Lakner, S., Lovrić, N., McGuinnes, S., Nabuurs, G-J., Sánchez-Arcilla, A., Settele, J., & Pe'er, G. (2023). Securing success for the EU Nature Restoration Law. Manuscript accepted for publication in *Science* on 28th November, 2023.

A declaration of author contribution precedes each article.

Chapter 1

Effects of agricultural land use on river biota: a meta-analysis

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REVIEW **Open Access**

Effects of agricultural land use on river biota: a meta-analysis

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Abstract

Agriculture, the world's most dominant land use type, burdens freshwater biodiversity with a multitude of stressors such as diffuse pollution and hydromorphological alteration. However, it is difficult to directly link agricultural land use with biota response as agricultural stressors can also originate from other causes. Also, there is evidence for positive and negative effects of agriculture on organisms, agricultural impact differs strongly with the biological metric and study region considered and agricultural impact differs among practice and type, which in turn affects different organism groups with varying severity. Against this background, our study aimed at assessing, if agricultural land use has a consistent effect on river biota. We conducted a systematic review of the literature, which yielded 43 studies and 76 relationships between agriculture and aquatic organism groups. The relationships were subjected to a meta-analysis using Hedge's g to calculate the standardized mean difference of effects. Overall, we detected a medium to strong effect $g = -0.74$ of agricultural land use on freshwater biota, only marginally influenced by study design, river type and region. Strong differences in biota response could be observed depending on the biological metric assessed, with ecological quality indices of agricultural impairment performing best. Sensitive taxa declined with agricultural impact, while tolerant taxa tended to benefit. In addition, the biota response differed among agricultural types and practices and organism group, with macroinvertebrates showing the strongest effect. Our results quantify the effects of agriculture on riverine biota and suggest biological metric types for assessing agricultural impact. Further research is needed to discriminate between agricultural types and account for intensity.

Keywords: Benthic invertebrates, Diatoms, Farming, Fish, Macrophytes, Metrics, Review, Streams

Introduction

Agriculture is the world's dominant land use type covering approximately 51 million km^2, which accounts for nearly 50% of the world's habitable land [64]. More than 75% of this area is used for meat and dairy production, which accounts for less than 20% of the human calorie supply [1]. While large parts of the agricultural areas in developed regions like Europe are already characterized by farm management of high intensity [76], further expansion and intensification of agriculture is to be expected, given the increasing demand for food and the raising share of meat diet [33, 71]. The need for industrial crops arising from the growing world population further adds to agricultural intensification [83]. This steadily increasing intensity of agricultural land use causes biodiversity loss in many organism groups, such as birds [22], mammals [40] and insects [9].

Similarly, the biodiversity of rivers draining agricultural land is impaired. Agriculture affects river biota through a variety of stressors, particularly the influx of nutrients [73], agrochemicals [45] and fine sediment [43], as well as alteration of river morphology [78] and intensive river maintenance [7]. The detrimental effect of these stressors has been shown in a multitude of studies: Nutrient influx impacts macrophytes [58], insecticides cause impairment of macroinvertebrates [13], fish are affected by fine sediment influx [43], hydromorphological alteration impairs

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macroinvertebrates [35], while river management impairs various organism groups [10]. Often, agriculture is identified as the most relevant driver for the deterioration of river biota [72, 78]. However, several uncertainties obstruct general conclusions about the effects of agriculture on river biota.

First, stressors such as diffuse pollution and hydromorphological impairment can also be caused by other drivers like urbanization, flood protection, hydropower or navigation [23]. It therefore appears impossible to separate the effects of agriculture in case of multi-driver situations [17, 82]. Agricultural impacts clearly differ with distance to river shores and riparian vegetation [19]. Additionally, there is also evidence for the positive effects of agricultural land use on biota, such as riverine macrophyte diversity [6]. The effect of agriculture potentially differs from the biological metric used for describing the assemblage of an organism group. While the macrophyte species investigated by Baattrup-Pedersen et al. [6] benefit from nutrient influx, other species suffer from associated light deprivation and are consequently suppressed [39, 58]. Similar differences can be observed for macroinvertebrates: sensitive species of the order Ephemeroptera, Plecoptera or Trichoptera are suppressed by more tolerant filter feeders and grazers [46] and fluctuations in oxygen availability [42]. Taxa tolerant to anthropogenic disturbances are expected to be less impaired by agricultural land use [79] and therefore, metrics focusing on the quantity of organisms, such as the number of taxa, are likely to be less sensitive to agricultural stressors compared to metrics focusing on sensitive species [74]. Also, fish species tolerant to sedimentation have been found to benefit from agricultural land use [36], while benthic and substrates spawning species are impaired [14].

Second, the biotic response depends on the share of agricultural land use in river catchments [20], but also on agricultural types and practices. Cornfield farming, especially when close to riverbanks, can cause massive fine sediment influx combined with strong phosphorous enrichment [68]. Intensive livestock farming in the river catchment can result in nitrate influx into rivers after strong rainfall events [54], while the pasture is less problematic for riverine biodiversity [12]. However, the effect of individual stressors caused by agriculture clearly differs between organism groups. Insecticides, for instance, are more detrimental to invertebrates [4] compared to the aquatic flora, which tends to recover rapidly from pesticide exposure [11, 44] and to fish, for which acute toxic effects are rare and rather chronic effects are observed [57, 67]. The nutrient influx also mainly affects macroinvertebrates followed by fish [81], while aquatic flora may potentially even benefit [18, 52]. Fine sediment influx altering the habitat conditions is more detrimental for invertebrates, fish and diatoms compared to macrophytes [41, 43]. River management affects macrophytes most directly, while macroinvertebrate and fish are often indirectly affected by habitat alteration [51]. In conclusion, there is evidence that all the above-mentioned pathways can affect macroinvertebrates, but only some of them are relevant for fish, macrophytes and diatoms. Finally, the effects of agriculture on river biota may differ between spatial scales [2], study regions [53] and river types [29], soil type [21] or riparian vegetation [61].

Against this background, the main objective of this meta-analysis is to assess if agricultural land use has a consistent effect on river biota across ecoregions in different climate zones, independent of the individual study design. We analyse if there are consistent differences between organism groups, to which degree the effect is context specific and if there are generalizable stressor pathways. More specifically, we hypothesized that:

- Agricultural land use affects river biota negatively and is widely from study design, region and organism group. Agricultural impacts on river biota can be better assessed with metrics of species composition (ecological quality indices) compared to metrics of species richness.
- The biota response differs with agricultural types and practices present in the river catchments and in areas close to the river, with macroinvertebrates affected more strongly compared to fish, macrophytes and diatoms.

Methods

Literature search

We conducted a systematic literature search with a search string including the attributes country, freshwater ecosystem type and agricultural stressor. We only considered studies published between January 1990 and May 2022, as we expected both agricultural practice and biodiversity data to have changed over time and reported agricultural effects on freshwater biota to be less standardized in the years before, or untypical for today's situation (e.g. addressing pesticides that are meanwhile prohibited). To further restrict the variability of impacts and responses, we limited our search to references from subpolar, temperate and subtropical regions in North America, Europe and Oceania.

The following search string was used for an all-database literature search in the Web of science:

(europe* OR albania* OR austria* OR belarus OR belgium* OR bosnia* OR bulgaria* OR croatia* OR cyprus OR czech* OR denmark* OR England* OR UK OR estonia* OR finland* OR france OR germany* OR greece OR hungary* OR ireland* OR italy* OR itali* OR latvia* OR lithuania* OR luxemb* OR macedonia* OR malta* OR moldova* OR montenegro OR netherlands* OR norway* OR poland* OR portugal OR romania* OR slovakia* OR slovenia* OR serbia* OR sweden* OR switzerland OR spain OR ukraine OR "new Zealand" OR australi* OR oceanic OR USA* OR "united states" OR "north america" OR U.S* OR canada).

(Topic) and (stream* OR river* OR watershed* OR catchment* OR floodplain*).

(Topic) and (diatom* OR macrophy* OR fish OR phytobenth* OR phytoplan* OR macrobenth* OR macrozoobenth* OR "aquatic plant" OR "aquatic larvae" OR "water plant" OR invertebrate*) (Topic) and (agri* OR farm* OR agronom* OR cultiv* OR cropl* OR pasture OR livestock OR ranching).

(Topic) not (marin* OR sea* OR ocean*).

From the initial population of 6484 papers, we screened the titles and abstracts and excluded studies that:

- did not focus on agricultural impacts,
- did not report effects on river biota,
- did not include data from Europe, North America or Oceania,
- did not provide empirical data,
- provided a review, lacking the original data required for a meta-analysis.

This initial screening resulted in 387 studies that met all criteria and that were subjected to a full-text scan, to identify studies qualified for a meta-analysis, that is studies that provided the data required to calculate the effect size (see below). We excluded studies, for which no comparable effect sizes could be calculated because they either were lacking a control group or a gradient in agricultural land use, or because of missing data that could not be retrieved. Eventually, the meta-analysis was performed with 43 studies, from many of which several different effect sizes were derived.

Data extraction

From the selected articles, we extracted data from narrative descriptions, tables and figures (using webplot digitizer: https://apps.automeris.io/wpd/index.de_DE.html, last accessed on June 23rd, 2022) to calculate effect sizes. More specifically, we compiled data on sample size, mean, variance and correlation coefficients, or used the raw data provided to calculate these values. If important test statistics were not directly retrievable, we tried to estimate mean values following Wan et al. [80] and standard deviations (SD) following Higgins and Green [38]:

$$\text{mean} = \frac{q1 + m + q3}{3} \tag{1}$$

where m = median, q = quartile

$$SD = \sqrt{n}\frac{(\text{upper 95\%CI} - \text{lower 95\%CI})}{3.92} \tag{2}$$

where n is the sampling size and CI is the confidence interval.

Furthermore, we extracted information on the organismic group (macroinvertebrates, fish, macrophytes and diatoms), agricultural types and practices and the type of response metric. Abundance metrics entailed counts and biomass of specimens, while richness metrics included species richness and number of taxa. Quality metrics refer to ecologically relevant assessment indices (e.g. ecological quality ratios, indices of biological integrity or taxa scores). The analysed studies discriminated between livestock farming, grassland, arable land and a mixture (or not further defined), which we refer to as "Livestock" including livestock farming and grassland, "Arable land" and "Mixture" including mixture and not further defined agriculture. Further, we distinguished metrics of "tolerant" and "sensitive" taxa according to their tolerance towards pollution [49, 63]. Disregarding in-group variability, Oligochaeta, Chironomidae and Diptera were classified as tolerant taxa, while sensitive taxa include Ephemeroptera, Plecoptera and Trichoptera (EPT). Finally, we extracted information on geographical region, time of fieldwork, river type, river size and the prevailing agricultural stressors. We gathered information on study design, agricultural type and practices (including the spatial scale addressed: riparian zone or entire catchment) and whether agriculturally used areas were compared to forests, best attainable or other control areas or if an agricultural gradient was reported.

Statistical analysis

We used Hedge's g, a measure of standardized mean difference, to quantify biota response to agricultural land use [8]. Hedge's g is a modified version of Cohen´s d, corrected for small or unequal sample sizes. Hedge's g compares the means of the treatment (agricultural land use) and control groups (X_T and X_C, respectively), which are standardized by dividing the pooled standard deviation (SD_p) and multiplied by a correction factor J, to avoid sample size bias.

$$g = \left(\frac{X_T - X_C}{SD_p}\right) \cdot J \tag{3}$$

The pooled standard deviation was calculated using the sample sizes of control and treatment groups (n_T and

n_C) and the squared standard deviations of the respective groups (SD^2_T and SD^2_C):

$$SD_p = \sqrt{\frac{(n_T - 1)SD_T^2 + (n_c - 1)SD_C^2}{n_c + n_T - 2}}. \quad (4)$$

The correction factor J was calculated with the sample sizes:

$$J = 1 - \frac{3}{4(n_C + n_T - 2) - 1} \quad (5)$$

If studies reported correlation coefficients (r) rather than means and standard deviations, they were transformed to Hedge's g following Borenstein et al. [8]:

$$g = \left(\frac{2r}{\sqrt{1 - r^2}}\right)J \quad (6)$$

An effect size = |1| indicates a deviance of one standard deviation between the treatment and the control group. Positive and negative effect sizes indicate higher and lower metric values, respectively, in the treatment group, while an effect size = 0 indicates no effect. Following Cohen [16], an effect size of ≤|0.2| is considered as small, a value of |0.5| as medium and a value of ≥|0.8| as a large effect.

To test Hypothesis 1 (general effects of agricultural land use on biota and quality metrics assessing agricultural impact best), we ran a meta-analysis with the metafor package in R [77]. We also ran a multi-level meta-analysis with the function rmv.ma to account for potential high heterogeneity which is typical for meta-analyses in ecology [70] using the Study ID as a random effect. Subsequently, we used meta-regression and subgroup analysis to investigate dominant co-variables (e.g. metric category, geographical region, ecoregion or river type and size), referred to as moderators, and hence, further explain heterogeneity. To test Hypothesis 2 (effects of agricultural types and practices on different organism types), we used measures of agricultural land use types, agricultural stressors reported, as well as the spatial basis (riparian or catchment land use) and the organism groups as moderators.

Publication bias

Studies with large and strongly significant results are more likely to be published as opposed to studies reporting weak and insignificant effects. This misbalance of published effects is called publication bias, which requires thoughtful consideration in meta-analyses. We used a funnel plot including data augmentation following Duval [24] to discover asymmetry of effect sizes and their variances and to estimate the number of potentially missing studies. In an unbiased dataset, the association should result in a funnel-shaped graph with most studies located within the funnel area. Additionally, we applied Egger´s regression test to analyse the funnel asymmetry [25]. Finally, we applied the fail-safe n test as a second measure to detect publication bias, calculating the number of effect sizes needed to reduce the significance level to non-significance. A study is considered robust and thus publication bias is negligible, when the fail-safe number is greater than 5 k + 10 (k = number of effect sizes). We also tested for time-lag bias, arising when initially published studies have larger effects than those of later studies often reported in ecological meta-analyses [75].

Results

Overview of studies and results

A total of 43 studies were distributed between Europe, North America and Oceania with 18, 16 and 9 studies, respectively. The effects on macroinvertebrates were reported by 43 studies, 11 studies reported effects on fish, 3 studies on macrophytes and 11 studies on diatoms (multiple assignments possible). While 75% of the 43 studies named nutrient influx as a major stressor, 50% listed fine sediment, followed by morphological alteration and pesticide influx with 40–30%, respectively (multiple assignments possible). Concerning study design, 28 studies compared control and treatment, while 15 studies reported effects as a correlation coefficient. In total, 76 extracted effect sizes were used for the overall meta-analysis, with 43 cases belonging to the metric category "Richness & Abundance metrics" and 33 to "Quality metrics".

The overall effect size pooled from all 76 cases was $g = -0.74$ [CI − 1.01; − 0.48] with $Q = 660$, a total variability of $I^2 = 93\%$ and an estimated total heterogeneity of $Tau^2 = 1.1$, showing strong heterogeneity following [37]. Because the 76 effect sizes were retrieved from 43 studies only, we applied a multilevel meta-analysis with the Study ID as a random effect yielding no change in overall effect size. Nearly, all heterogeneity is explained by the effect size level ($Tau^2_{Level\ 2} = 1.08$) with the study size level explaining less than 1 percent ($Tau^2_{Level\ 3} < 0.00$).

Conducting meta-regression analysis to identify moderators explaining heterogeneity based on the Akaike Information criterion [3], we were able to explain $R^2 = 54.3\%$ of the heterogeneity using four moderators: metric category, geographical region, agricultural types and organism types.

Effects of agriculture on river biota and metrics describing them best

Individually accounting for $R^2 = 40.4\%$ of heterogeneity, "metric category" was the most important moderator (Fig. 1): Richness & Abundance metrics had a $g = -0.16$ [CI − 0.50; 0.17], which is considered a non-significant low effect, while for Quality metrics a very large effect was observed: $g = -1.41$ [CI − 1.66; − 1.17]. We also applied multilevel meta-analysis rendering a small change of the subgroup effect sizes to $g = -0.18$ [CI − 0.92; 0.57] for Richness & Abundance metrics and $g = -1.47$ [CI − 1.81; − 1.14] for Quality metrics with $\text{Tau}^2_{\text{Level 2}} = 0.50$, $\text{Tau}^2_{\text{Level 3}} = 0.14$. The attempt to investigate, which studies caused study level heterogeneity ($\text{Tau}^2_{\text{Level 3}}$) was not successful. In conclusion, the effect sizes did not change from the two-level meta-analysis for the overall meta-analysis and only marginally for the subgroups and most heterogeneity was found on the effect size level. Against this background, we limited our further analysis and the graphical representation to a two-level meta-analysis to reduce complexity.

Further subgroup analysis suggests that agricultural effects are stronger in the northern hemisphere compared to the southern rendering strong negative effects in the northern hemisphere and no effects in the southern hemisphere (Fig. 2). Only small differences were observed between ecoregions with slightly stronger effects in the subtropical compared to temperate and subpolar regions (Fig. 3). No major differences were observed for study design, river types and size, suggesting similar effects for lowland and mountain rivers (Table 1).

Biota response to different agricultural types and practices

The comparison between different agricultural types suggests that arable land affects river biota more strongly than livestock farming. Geographically, most studies from North America and Europe report the effects of arable land use, while the studies from Oceania almost exclusively report livestock effects (Fig. 2). No major differences were observed between catchment- and buffer land use (Table 1) and meta-regression with the number of stressors reported did not explain any heterogeneity (not shown). Small differences in the response of different organism types could be observed with macroinvertebrates affected most. We observed a large effect for macroinvertebrates, a medium effect for fish and macrophytes and only a small effect for diatoms (Fig. 4).

Sensitive vs tolerant taxa

We conducted another smaller meta-analysis with the 16 studies reporting effects for either pollution-sensitive or pollution-tolerant macroinvertebrate taxa (Fig. 5). We observed strong differences in effect size between those ($g = -0.6$ [CI − 1.22; − 0.70] vs $g = 1.51$ [CI 0.66; 2.34], with sensitive taxa being strongly impaired and tolerant taxa benefiting.

Publication bias

The Egger´s regression test suggests no significant publication bias ($z = -0.56$, $p = 0.58$), and likewise the funnel plot only suggests a small lack of studies with positive effects (empty circles, Fig. 6). Also, no time-lag bias could be found and the fail-safe number of 7730 is greater than 5 k + 10 (> 390) and therefore highly significant. Consequently, the results of the meta-analysis can be considered robust.

Discussion

Effects of agriculture on river biota and metrics describing them best

We expected that agricultural land use is negatively related to riverine biota, irrespective of region and organism group considered (Hypothesis 1). This expectation was supported by a medium to a strong negative overall effect size of $g = -0.74$ [− 1.21; − 0.62] [16], despite differences among geographical regions and ecoregions. The observed strong heterogeneity between the effect sizes most likely resulted from the combination of several studies of different geographical regions with varying agricultural land use, considered target organisms and effect endpoints.

The positive effects reported in the literature can be partly explained by the type of indicator considered. Fitzpatrick et al. [31] addressed the response of fish species richness, which is known to increase with river size [34], as agricultural land cover is also correlated to catchment size; this may explain the positive relationship between catchment agricultural area and fish richness. Likewise, the positive impact of agriculture on diatoms reported by Zheng et al. [85] was observed for overall species richness, while the authors reported a decline of pollution-sensitive taxa along a gradient of agricultural area. Similar relationships were found for extensive pasture in New Zealand, which showed positive effects for overall species richness but negative effects for more

(See figure on next page.)

Fig. 1 Effect size of agricultural land use on different organism groups accounting for different metric types. Shown are individual effect sizes per study, pooled effect sizes for two subgroups divided by metric type and an overall pooled effect size with 95% confidence intervals. Effect sizes are significant if 95% CI does not overlap the vertical zero line. Richness = species-, taxa richness; abundance = density, mass, count; *ASPT* Average core er taxa; *IBI* Index of biotic integrity, *EPT* Ephemeroptera, Plecoptera, Trichoptera, *EQR* Ecological quality ratio, *O/E* Observed/Expected, *SPEAR* Species at risk, *WQ* sensitive Water quality sensitivity index

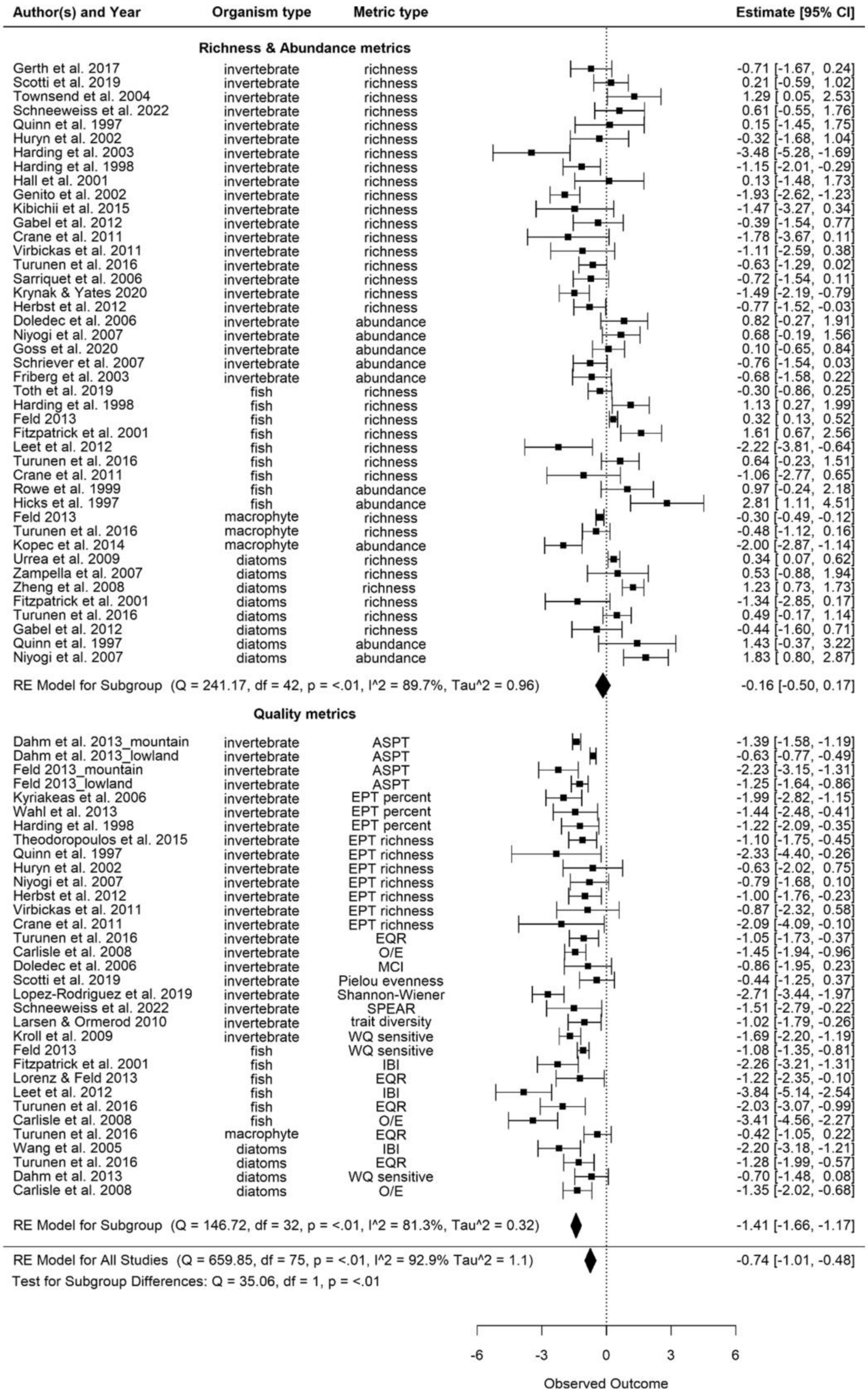

Fig. 1 (See legend on previous page.)

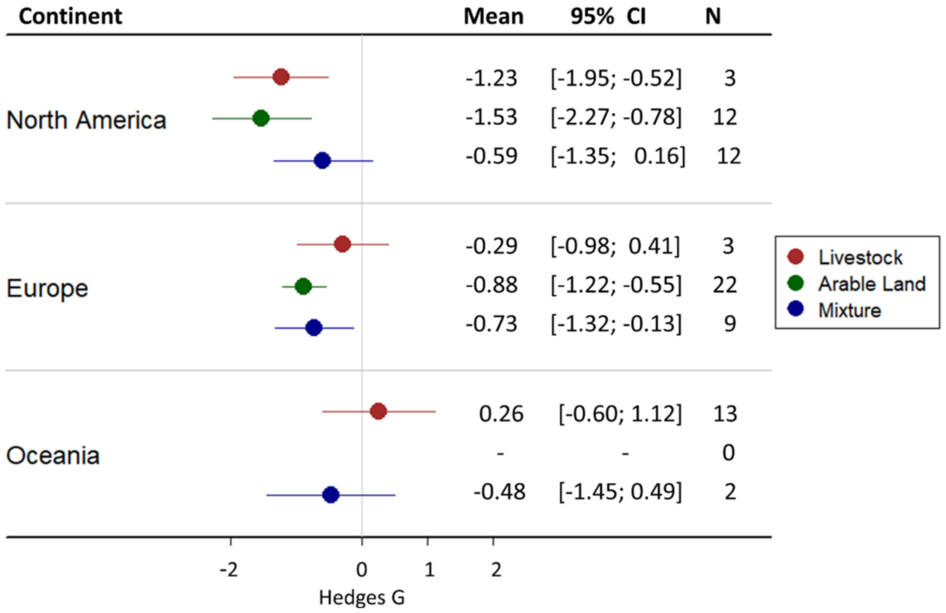

Fig. 2 Mean effect sizes of different agricultural types on river biota in different continents. Shown are mean summary effect sizes (mean), 95% confidence intervals (95% CI) and number of effect sizes (*N*). Effects are significant if 95% CI does not overlap the vertical zero line

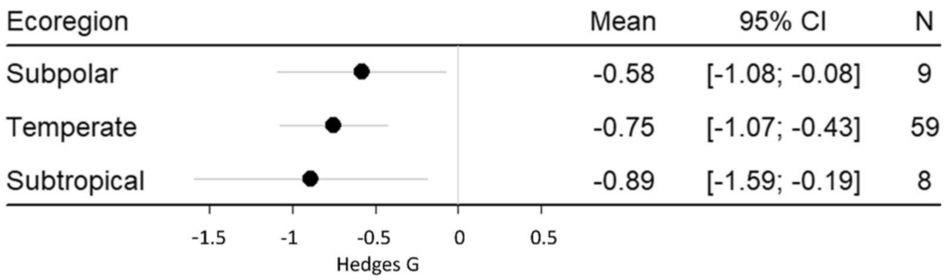

Fig. 3 Mean effect sizes of agricultural land use on river biota in different ecoregions. Shown are mean summary effect sizes (mean), 95% confidence intervals (95% CI) and number of effect sizes (*N*). Effects are significant if 95% CI does not overlap the vertical zero line

pollution-sensitive taxa (e.g. [56, 62]. These findings match our expectation that overall species richness is a poor indicator of agricultural stress because the loss of stress-sensitive species may be balanced by stress-tolerant ones. Consequently, agricultural impact on riverine biota is better reflected by metrics of species composition (quality metrics) as compared to metrics of species richness/abundance.

This was further supported by the subgroup analysis showing a very strong negative effect for quality metrics, while we only found a small negative effect for richness/abundance metrics (Fig. 1). Additional support was obtained by the analysis of sensitive *vs.* tolerant taxa, which showed a strong negative effect of agriculture for sensitive taxa, but a strong positive effect for tolerant taxa (Fig. 6). Together, these findings suggest a stronger land use effect on community compositional metrics. Compositional metrics used to assess the impact of agriculture should include sensitive taxa, because those are the first to respond with a decline in their abundance or richness. Stress-tolerant species, however, may increase due to subsidy effects by nutrient enrichment [60, 84] or due to the release from competition with sensitive taxa [46]. These opposing effects on sensitive and tolerant taxa also explain why metrics of whole-community richness performed much weaker. Likewise, [30] found that the loss of sensitive taxa due to hydromorphological degradation may be

Table 1 Subgroup analysis for river types, river size, spatial basis and study design

	Mean	95% CI	*N*
River type			
Mountain	− 0.72	[− 1.30;− 0.15]	17
Lowland	− 0.76	[− 1.09;− 0.42]	45
River size			
Small	− 0.72	[− 1.24;− 0.21]	29
Large	− 0.68	[− 1.09;− 0.27]	27
Spatial basis			
Catchment	− 0.75	[− 1.04;− 0.46]	69
Buffer	− 0.70	[− 1.07;− 0.34]	7
Study design			
Control vs impact	− 0.78	[− 1.12;− 0.45]	51
Gradient	− 0.68	[− 1.11;− 0.25]	25

Shown are mean summary effect sizes (mean), 95% confidence intervals (95% CI) and number of effect sizes (*N*)

largely balanced by more tolerant ones. Thus, metrics of whole-community richness are likely to be insufficient for the analysis of agricultural stress.

Consequently, the observed positive effects in Oceania (Fig. 2) can be partly explained by the number of richness metrics reported compared to quality metrics. Only four effect sizes belong to the category of quality metrics with eleven belonging to richness metrics. The geographical differences can be further explained by different types and practices of agriculture (compare Hypothesis 2 below). The differences between ecoregions reflect climate conditions. Rivers in arid zones, for example in the Mediterranean, are frequently impaired by water abstraction pressures [53] and by particularly intense land use close to the rivers [55] leading to multiplying stressed biota [69]. Subpolar regions, on the other hand, are known for lower crop production and livestock populations [26, 27].

Biota response to agricultural types and practices

We expected that types and practices of agriculture in the river catchments would be reflected in the biota response (Hypothesis 2). This expectation was supported by differing effects between arable land and livestock farming (Fig. 4), with arable land impairing river biota more strongly. Those effects differed across continents, with agriculture in Northern America most affecting river biota most followed by Europe and surprisingly positive effects in Oceania (partly explained by the metric category assessed as described above). The positive effect in Oceania can be further explained as most studies assessed the effects of extensive pasture farming and no studies with arable land use were incorporated in this analysis (Fig. 2). Although most studies did not report more detailed information on agricultural types and practices, differences in biota response across continents are likely to be linked to different agricultural practices. In particular, agriculture in North America is generally known for large scale highly intensive machinery farming with higher nutrient input concentrations compared to Europe and Oceania [59].

Several studies report strongly differing effects of agricultural types and practices with livestock farming in river catchments resulting in nitrate influx [54], or corn production causing massive fine sediment influx [68]. The fact that we did not find an effect for livestock farming can be explained by a multitude of studies with extensive pasture from Oceania, which is considered less influential on river biota [12]. Additionally, the differences between arable land and livestock farming can be explained by pesticide usage. Unlike livestock farming arable land probably mirrors their exponentially increasing use [72] causing severe pressure on pesticide-sensitive species [50]. Although we could not investigate this further as in-depth information on the agricultural types was lacking, we expect for the effects of agricultural types to differ based on crop specific

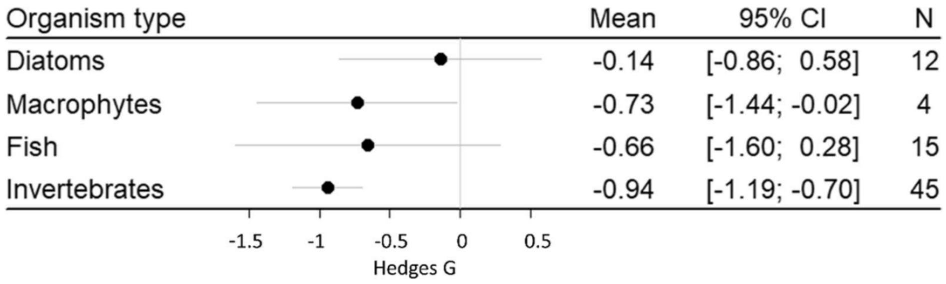

Fig. 4 Mean effect sizes of agricultural land use on different organism types. Shown are mean summary effect sizes (mean), 95% confidence intervals (95% CI) and number of effect sizes (*N*). Effects are significant if 95% CI does not overlap the vertical zero line

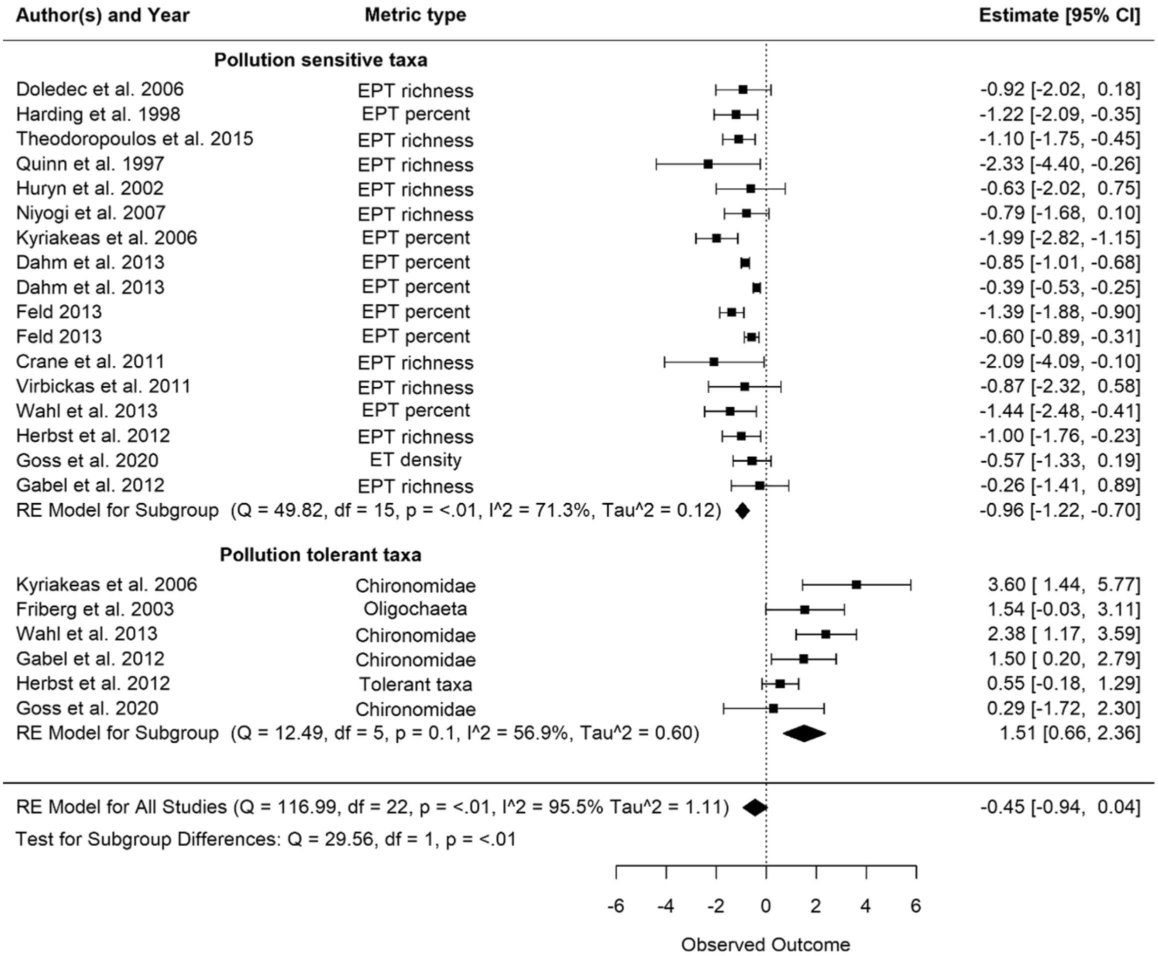

Fig. 5 Effect size of agricultural land use on sensitive vs. tolerant taxa. Shown are individual effect sizes per study, pooled effect sizes for two subgroups divided by metric type and an overall pooled effect size with 95% confidence intervals. Effect sizes are significant if 95% CI does not overlap the vertical zero line. *EPT* Ephemeroptera, Plecoptera and Trichoptera; *ET* Ephemeroptera and Trichoptera

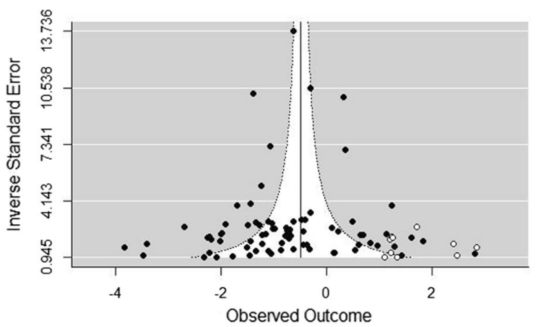

Fig. 6 Funnel plot with effect size against inverse standard error. Empty circles are effect sizes expected when no publication bias was present

pesticide treatment [5]. However, 75% of the studies named nutrient influx as major stressor followed by fine sediment, morphological alteration, and pesticide influx with 50%, 40% and 30%, respectively. The presumed importance of pesticides can be explained by complex analysis to assess pesticide pressures (e.g. [15]. Additionally, several studies focusing on pesticides could not be incorporated into this analysis lacking comparable effect sizes or predictors (e.g. [15, 66].

We also expected stronger responses for macroinvertebrates compared to fish, macrophytes and diatoms. While macrophytes were only slightly stronger affected compared to fish and macrophytes, a strong difference could be found in comparison to diatoms as shown in the subgroup analysis (Fig. 5). This coincides with the reported predominant stressors and invertebrates suffering from all, nutrients [48], pesticides [28], fine sediment [32] and morphological alterations for instance resulting in water temperature rise [35].

Study implications

This study shows clearly that freshwater biota are impaired by agricultural land, with small differences based on geographical region, ecoregion or organism types, but results vary strongly based on the metric used. Quality metrics encompassing various ecologically relevant assessment indices reflect stronger effects than metrics focusing on richness or abundance. The obvious reason for these differences is the differential effects of stressors on individual taxa, that is not all taxa are equally impaired, with some tolerant species even benefiting from less competition or enhanced food availability. For future assessment of agricultural impacts, we suggest the use of metrics considering the tolerance of organisms, ideally metrics that are specific for individual stressor types caused by agriculture [32].

Agricultural stress is likely depending on the soil and climate conditions and agricultural types and practices, partly reflected in the ecoregions with stronger agricultural effects in the subtropical region compared to temperate and subpolar regions, potentially caused by water stress. Large parts of heterogeneity we could not explain are likely to be situated in the intensity and type of agriculture, in particular cropland densities [20], pesticide use [5], fertilizer use and biomass production [47]. Hence, in order to truly account for the impact of agricultural land use on river biota, further systematic investigations on the role of agricultural types, intensities and spatial arrangement are needed, not disregarding interactions with riparian vegetation as hinted by Palt et al. [61].

The present meta-analysis could only render hints suggesting the strongest agricultural impacts in North America, which is known for large-scale high intensity agriculture and in general stronger impacts for arable land compared to livestock farming. Still, the lower effects of livestock farming need to be taken with care, as they mostly receive concentrated feed imported from other regions [65]. Hence, parts of the environmental impact are spread, and animal production is less connected to the local river conditions.

Accordingly, after several years of research on agricultural impacts on freshwater biota, it is still not possible to quantitatively discriminate between agricultural practices, intensities, and types (such as individual crop types) and their interaction with riparian buffers. This is a significant knowledge gap that obstructs tailor-made solutions for minimizing the effects of agriculture on river biota. Obviously, these solutions are urgently required, given the overall strong impact of agriculture. Therefore, we call for further research to directly link agricultural types and intensities with biota response to derive management directives. Approaches could include but are not limited to accounting for crop-specific pesticide and nutrient application rates, different management practices including organic farming and mosaic farming or the occurrence of monocultures.

Supplementary Information

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Additional file 1. References of studies incorporated in meta-analysis.

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Author contributions

CS: Conceptualization, investigation, literature review and analysis, drafting and writing of manuscript. CKF: Conceptualization and review, editing. JK: Conceptualization and review, editing. DH: Conceptualization and review, editing. All authors read and approved the final manuscript.

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Availability of data and materials

The dataset used and/or analysed during the current study is available from the corresponding author upon reasonable request.

Declarations

Ethics approval and consent to participate

Not applicable.

Consent for publication

Not applicable.

Competing interests

All authors declare no competing interests.

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- Conception – 60%
- Conduction of experimental work – not applicable
- Data analysis – 100%
- Species identification – not applicable
- Statistical analysis – 100%
- Writing the manuscript – 80%
- Revision of the manuscript – 50%

Signature of the Doctoral Candidate Signature of the Doctoral Supervisor

Chapter 2

River ecological status is shaped by agricultural land use intensity across Europe

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River ecological status is shaped by agricultural land use intensity across Europe

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Abstract

Agriculture impacts the ecological status of freshwaters through multiple pressures such as diffuse pollution, water abstraction, and hydromorphological alteration, strongly impairing riverine biodiversity. The agricultural effects, however, likely differ between agricultural types and practices. In Europe, agricultural types show distinct spatial patterns related to intensity, biophysical conditions, and socioeconomic history, which has been operationalised by various landscape typologies. Our study aimed at analysing whether incorporating agricultural intensity enhances the correlation between agricultural land use and the ecological status. For this, we aggregated the continent’s agricultural activities into 20 Areas of Farming-induced Freshwater Pressures (AFFP), specifying individual pressure profiles regarding nutrient enrichment, pesticides, water abstraction, and agricultural land use in the riparian zone to establish an agricultural intensity index, and related this intensity index to the river ecological status. Using the agricultural intensity index nearly doubled the correlative strength between agriculture and the ecological status of rivers, as compared to the share of agriculture in the sub-catchment (based on the analysis of more than 50,000 sub-catchment units). Strongest agricultural

pressures were found for high intensity cropland in the Mediterranean and Temperate regions, while extensive grassland, fallow farmland and livestock farming in the Northern and Highland regions, as well as low intensity mosaic farming, featured lowest pressures. The results provide advice for pan-European management of freshwater ecosystems and highlight the urgent need for more sustainable agriculture. Consequently, they can be also used as a basis for European Union-wide and global policies to successfully halt biodiversity decline such as the post-2027 renewal of the Common Agricultural Policy.

1. Introduction

European rivers are burdened by multiple anthropogenic stressors (Birk et al., 2020), including point source pollution (Bouraroui & Grizzetti, 2011), diffuse pollution (Mellander et al. 2018) and hydromorphological alterations (Vaughan & Ormerod, 2010; Fernandes et al., 2020). They are also impaired by increasing temperature (Strayer & Dudgeon, 2010), water scarcity (Arenas-Sánchez et al., 2016) and invasive species (Gallardo et al., 2016). All these stressors affect the health of river ecosystems, so that only about 40 percent of the more than 100,000 regularly monitored European river water bodies reached good ecological status in 2015 (EEA, 2018a).

Agriculture, Europe´s most dominant land use covering nearly 40% of its terrestrial surface (Eurostat, 2022c), has been held responsible for the above-mentioned stressors (Moss, 2008), including diffuse pollution from excess use of fertilizers (Grizzetti et al., 2017) and pesticides (Liess et al., 2021), unsustainable water abstraction (Zal et al., 2017) and hydromorphological alterations of the channel bed, riparian vegetation and floodplains (Baattrup-Pedersen et al., 2018). There is strong empirical evidence for negative agricultural effects on river biota (Schürings et al., 2022). While nutrients can lead to eutrophication effects (Almeida et al., 2018), pesticides have the potential to cause lethal and sublethal toxic effects to aquatic organisms (Schäfer, 2019), both deteriorating the water quality. Water abstraction results in a decrease of dilution capacity, further decreasing water quality (Baccour et al., 2021). In addition, low water levels reduce lateral connection and eventually causes loss of riparian habitat (Bond et al., 2019) and desiccation of rivers, mainly occurring in the Mediterranean (Karaouzas et al., 2018) but also increasingly observed in other parts of Europe (Zal et al., 2021). Hydromorphological alteration changes the structure and availability of habitats for aquatic organisms, resulting in extensive community turnover or species loss (Elosegi & Sabater, 2013). Hence unsurprisingly, agriculture has been suggested as a key driver for the deterioration of freshwater ecosystems (Tilman et al., 2001; Tang et al., 2021; Wolfram et al., 2021), which are considered among the most degraded ecosystems in the world (Ferreira et al., 2019).

Generally, the agricultural activities in the European Union (EU) are characterized by high farm management intensity, i.e., homogenized landscapes such as monoculture and mechanized production systems (spatial configuration), high nutrient input and pesticide treatment (input intensity) and high biomass production (output intensity) (Levers et al., 2018; Schreiner et al., 2021). However, European agricultural practices show distinct spatial patterns across the continent, related to diverse environmental conditions (mainly climate and biogeography), as well as the economic and socio-political settings at local, regional and national level (Metzger et al., 2005; Kuemmerle et al., 2008). These spatial patterns result in different types of agriculture, which represent different agricultural usage intensities.

Such differences in agricultural types and management intensities have been shown to exert different levels of pressure on streams and rivers. Corn farming, for instance, is associated with high sediment-bound phosphorus influx into rivers (Secchi et al., 2011), vineyards and vegetable cultivation burden streams with periodic pesticide loading (Schulz, 2001; Andert et al., 2015), and livestock farming adjacent to rivers can cause nutrient and fine sediment pollution (Wilson & Everard, 2018). The effects of these different farming practices on river biota have been documented in various studies at local to national scales (e.g. Weijters et al., 2009; Wasson et al., 2010, Schürings et al., 2024b). To date, however, the effects of differences in agricultural types and intensities on river biota have never been analysed at a continental scale. Such analyses were hampered by the lack of synthesis of data on agriculture, its pressures and ecological effects. The Water Framework Directive, which is based on Europe´s largest biological monitoring programme, has made consistent information on the ecological status available at the EU-scale (EEA, 2018a). The ecological status indicates deviations of biotic communities (benthic invertebrates, fish, macrophytes and phytobenthos) from natural, undisturbed conditions (Birk et al., 2012). Moreover, the European farming landscape has been classified into specific types of agriculture differing in land cover and management intensity (Levers et al., 2018). These spatial patterns presumably relate to patterns of environmental pressures exerted from agricultural activities and allow for analysing of the ecological response.

Analyses at the continental scale have shown that climatic and biogeographical factors play a central role for biota and often interact with land use effects (Brucet et al., 2013; Feld et al., 2016). This suggests to consider biogeographical regions in the assessment of land use effects, as it has been implemented in comparative studies of ecological status in Europe (Poikane et al., 2014). Ecological effects on riverine biota also differ between types of ecosystems, so that river types should always be taken into account (Lyche Solheim et al., 2019; Lemm et al., 2021). For instance, today's biota of smaller streams are more vulnerable than those of large rivers due to stronger land-water-interface and reduced dilution-capacity effects (Leps et al., 2015). This vulnerability holds particularly for Mediterranean intermittent streams (Smeti et al., 2019). More heterogeneous land use patterns in upland areas make biota in rivers in hilly and low mountainous regions react more strongly to pressure gradients than in lowland rivers (Feld, 2013; Li et al., 2018). Accounting for biogeographical as well as river typological factors is therefore indispensable for analysing the farming-induced ecological effects on a European scale.

In this study we identified how the European types of agriculture differ in their intensity of farming-induced pressures exerted on freshwaters and ultimately on ecological status. These pressures include diffuse pollution (i.e. nutrients and pesticides), water abstraction for irrigation and agricultural land use in the river floodplain, a proxy for agriculture-induced hydromorphological alterations. Adopting the farming landscape typology of Levers et al. (2018) and the Biogeographical Regions of Europe (Roeckaerts, 2002), pressure levels were allocated to these types of agriculture, which allowed to delineate 20 Areas of Farming-induced Freshwater Pressures (AFFP) with characteristic farming-induced freshwater pressure-profiles to establish an agriculture intensity index. We related this agricultural intensity index with the ecological status in different European river types covering more than 50,000 hydrological sub-catchments.

In particular, we addressed the question if accounting for agricultural intensity does improve the correlative strength between agricultural land and the ecological status and how this differs between different types of rivers.

2. Methods

2.1 Data basis

2.1.1 Spatial reference

The study was built on the 'European catchments and rivers network system' (ECRINS), i.e., a geographical information system of the European hydrographical sub-catchments organised from a layer of 104,684 so-called 'Functional Elementary Catchments' (FECs) with an average size of 60 ± 72 km² (EEA, 2012). The FEC-level represents the spatial unit, at which all data used in this study were processed. All data referring to different spatial units, for instance NUTS (Eurostat, 2021) or E-HYPE sub-basins (Lindström et al., 2010), were transferred into FEC-level (Globevnik et al., 2017). For 51,625 FECs covering nearly 80 percent of EU, data on freshwater stressors and the ecological status were available, which were used for the subsequent analysis. For better comprehensibility, we refer to the FECs as ‘river catchments’ in the sections below.

2.1.2 Existing landscape classifications

Two existing landscape classifications were used in this study: (1) The land-system Archetypes (Levers et al., 2018) and (2) the Biogeographical Regions of the Habitats Directive of the European Community (Roekaerts, 2002). The land-system Archetypes were derived using Self Organising Maps and evaluated in an expert workshop. They were mapped at the pan-European scale (covering 27 Member States of the European Union, including the UK but excluding Croatia) on the basis of selected land-cover (e.g. cropland or forest) and land-use indicators (e.g. nitrogen input, livestock density, or crop harvesting) for the year 2006. These archetypes describe landscapes featuring similar patterns of land cover and management intensity (input and output intensity), for instance high intensity cropland, medium-intensity livestock farming, low-intensity mosaic and fallow farmland.

The six Biogeographical Regions of Roeckaerts (2002) were summarized into three groups used in all subsequent analysis: ‘Temperate’, ‘Mediterranean’, and ‘Northern and Highlands’, matching the geographical intercalibrations groups relevant in ecological status classification (Poikane et al., 2014). They consider the coarse climatic differences between the areas, which influences the agricultural systems: The Temperate Region generally exhibits favourable climatic conditions for productive agriculture. Climate-induced water scarcity is the decisive factor for the Mediterranean Region, and the Northern and Highland Region comprises less-favourable areas for agricultural production due to wet and cold climatic conditions (Metzger et al., 2005).

2.1.3 Pressure indicators

The four pressure indicators chosen in this study match the agricultural pressures from agriculture on the aquatic environment, outlined in the report on water and agriculture by the

European Environmental Agency (EEA, 2021b) to represent the major farming-induced freshwater pressures.

Diffuse pressures: nutrients

To estimate the effects of farming-induced nutrient pollution on freshwater ecosystems, the parameter nitrogen surplus, which is the difference between nitrogen input (e.g. fertilisers, feed) and output (e.g. animal and plant products) was used. It was calculated using the CAPRI (Common Agricultural Policy Regional Impact Analysis) modelling system (Britz and Witzke, 2014), a global economic model for agriculture with a regionalised focus for Europe, which uses regional and national data and is based on official EUROSTAT statistics. The CAPRI nitrogen balances relevant for this study were estimated for the year 2012 with a spatial resolution of 1 km, on the basis of four components: (1) Export of nutrients by harvested material per crop, depending on regional crop patterns and yields, (2) output of manure, depending on the animal type, (3) input of mineral fertilizers, based on national statistics at sectoral level and (4) a model for ammonia pathways. The model outputs provided indicators of regional farm, land and soil nitrogen-budgets and nitrogen-flows of the agricultural sector at the European scale (Leip et al., 2011). The indicator 'nitrogen surplus on agricultural areas' was selected as a proxy for nutrient pollution pressure, aggregated at river catchment-level.

Diffuse pressures: pesticides

To quantify the effects of pesticides on the freshwater ecosystems, the chronic multi-substance Potentially Affected Fraction (msPAF) was used, derived from Europe-wide integrated exposure and effect modelling for the year 2013, based on about 23,000 sub-catchments with an average size of 252 km² (van Gils et al., 2020). The msPAF specifies the potential share of the aquatic species community affected by pesticide toxicity. The model includes two components: (1) A spatio-temporally resolved model for emissions and fate-transport of chemicals driven by a hydrological model (van Gils et al., 2020), yielding Europe-wide daily predicted environmental concentrations (freely dissolved part) of 332 pesticides (Table S1) in water bodies to obtain a "real-life" mixture exposure scenario for each river catchment, and (2) species sensitivity distributions (SSD) based on effect models considering chronic non-observed-effect concentrations (NOEC) of each studied chemical as effect endpoint (Posthuma et al., 2019). Combining (1) and (2) yields the mixture toxic pressure metric msPAF (de Zwart & Posthuma, 2005; Posthuma et al., 2020), being an estimate of the likelihood (values between 0 and 1) of direct effects of chemical exposure to effect-endpoints of aquatic organisms such as growth and reproduction (Posthuma et al., 2019). In this study, the msPAF-NOEC based on 99th percentile predicted environmental concentrations of the daily concentration estimates was used, representing an acute toxic stress level exceeded at four days per year.

Hydrological pressure: water abstracted for irrigation

To consider the effect of crop irrigation, the annual volume of water abstracted for irrigation (acquired for the year 2015) was compiled (Zal et al., 2017) for whole Europe, except for Belarus, Russia and Ukraine indexed per hectare of irrigated crop land. The data are intermediate model outputs from transformation and downscaling processes estimated at the river catchment level from the pan-European water quantity accounting exercise of the European Environmental Agency, using the Nopolu software (https://www.naldeo.com/environment/water-resources-management-116). Monthly values were summed up to the total annual amounts and transformed to cubic metres per hectare, dividing the annual water abstracted in each river catchment by the irrigated crop area in the respective river catchment.

Hydromorphological pressures: agricultural land use in potential river floodplain

To incorporate the hydromorphological alteration caused by agriculture, the area of agricultural land located in the potentially flood-prone areas was calculated as an average of the years 2011 to 2013 (EEA, 2020). It was derived from two spatial layers, (1) the JRC flood hazard map (100-year return period) for Europe, compiled with the flood model 'LisFlood' (Bates & De Roo, 2000; Alfieri et al., 2014) and (2) the Copernicus Potential Riparian Zone layer compiled with data from the Copernicus Land Monitoring Service (EEA, 2015; CLMS, 2019). This proxy-indicator, also used in the report of the European Environmental Agency on water and agriculture (EEA, 2021b), allows for an estimate of various farming-induced pressures on the freshwater ecosystems. It can be interpreted as the probability of hydrological and morphological alterations to surface waters due to agricultural activities (e.g. drainage, straightening, and embarkments) in the floodplain area.

2.1.4 River types

Following the river typology of Lyche Solheim et al. (2019) synthesising the river typologies of the EU member states, we grouped the European rivers of the European catchments and rivers network system into 12 broad river types. The types are based on river size, altitude, and catchment biogeography, i.e., the main typological factors defined by the Water Framework

Directive. In the rare cases when more than one river type was located in a river catchment, the type at the catchment´s outlet was selected.

2.1.5 Ecological status

The ecological status, an assessment of the quality of the structure and health of the surface water ecosystems (European Commission, 2000), reflects the effect of pressures on the different river organisms: macroinvertebrates, fish, and aquatic flora (macrophytes and phytobenthos). The observed species taxonomic composition is compared to type-specific undisturbed reference conditions (Wallin et al., 2003), resulting in an Ecological Quality Ratio (EQR) classified into five classes (high, good, moderate, poor, bad). Data on the ecological status were available for 51,625 river catchments from the second River Basin Management Plan reports for the year 2010-2015 supplemented by data on the first reports (2006-2009), where national data were missing to increase coverage (Lemm et al., 2021). The monitoring was performed by the EU member states and results were reported to the European Environmental Agency (EEA, 2012; EEA, 2020). For river catchments with more than one assessed river, the river body at the river catchment´s outlet was selected, assuming, that the catchment stress increases downstream. The classification of the ecological status depended on the number of organism groups measured for each river catchment. Approximately 90% of the river catchments incorporated macroinvertebrates, 66% included diatoms, 50% involved fish and 33% incorporated macrophytes. Hence, in most cases, multiple organism groups contributed to the status classification and the overall ecological status classification for a river catchment was determined by the organism group with the worst status following the “one out, all out” principle (European Communities, 2005). Consequently, river catchments with more than one organism group assessed are more likely to be classified as a lower status compared to river catchments with only one organism group assessed.

2.2 Data analysis

To relate agricultural intensity with the ecological status, we first generated homogenous groups of agricultural land use. Then we applied them at the river catchment-level to establish an ‘agricultural intensity index’ and to assign each river catchment a value of agricultural intensity, which we subsequently related to the ecological status.

2.2.1 Delineation of Areas of Farming-induced Freshwater Pressures (AFFP) with assigned pressure-profiles relevant for freshwater ecosystems

As a first step to establish the agricultural intensity index, we generated homogenous groups of agricultural land use (i.e. Areas of Farming-induced Freshwater Pressures -AFFP), featuring similar agricultural production and farming-induced freshwater pressure intensities. For this, existing landscape classifications were combined and associated with the pressure data in four steps as described below:

First, from the 15 different land-system archetypes provided in 3 km raster-cells of Levers et al. (2018), the twelve agriculture-related archetypes were selected for further processing. These were combined with the three large European regions resulting in 12 x 3 = 36 'regionalized archetypes'. The regionalized archetypes were then transferred to the river catchment-level, i.e., for each catchment the percentage cover of the different regionalized archetypes was calculated.

Second, to calculate reliable pressure-profiles, only river catchments, for which a specific regionalized archetype was dominant (i.e. ≥66.6% river catchment coverage) were chosen. Median and interquartile ranges were used to compare the pressure-ranges of the four pressure indicators (nutrients, pesticides, water abstraction, hydromorphological alterations) in each regionalized archetype. For this, the pressure levels were ranked, ranging from 1 ('very low') to 4 ('high') according to their medians referring to either existing classification of pressure intensity for nitrogen surplus (Rega et al., 2020), pesticides (van Gils et al., 2020), or expert judgement for water abstraction and floodplain agricultural land use (Table 1). These rankings for individual pressure-ranges established a 'pressure-profile' for each regionalized archetype. More sophisticated approaches such as min-max transformation did not improve the classification performance, owing to outliers misrepresenting the differences between the river catchments, so we chose this straightforward approach.

Table 1: Pressure indicators and levels ranked into four classes of freshwater pressure.

Presure indicator	**Unit**	**Very low (1)**	**Low (2)**	**Medium (3)**	**High (4)**
Nitrogen surplus	kg/ha/year	≤ 20	> 20 – 35	> 35 – 50	> 50
Potentially Affected Fraction of species by pesticides	--	≤ 0.15	> 0.15– 0.30	> 0.30 - 0.50	> 0.50
Water abstracted for irrigation	m^3/ha/year	≤ 1,000	> 1,000 – 3,500	> 3,500 – 6,000	> 6,000
Hydromorphological alterations (Agricultural land use in the floodplain)	%	≤ 50	> 50 – 65	> 65 – 80	> 80

Third, as for several regionalized archetypes only few river catchments with dominant coverage (≥66.6 %) could be identified, these were grouped based on similar cultivation and pressure-profiles, but not across the large European regions, to conserve regional differences. Exceptionally, fallow farmland and extensive grassland were each merged across the large European regions, because the pressure levels were very similar and only minor regional differences were expected, to limit the total number of AFFP for reasons of clarity and comprehensibility (Table S2). This allocation resulted in a total of 20 AFFP, representing large-scale landscape units of similar agricultural land use and farming-induced pressures on freshwater ecosystems.

Fourth, to summarize the farming-induced pressures exerted on the freshwater ecosystem across Europe, a cumulative pressure index was calculated for each AFFP with river catchment-coverage of ≥66.6% (n = 17,099). We used the pressure levels of the four pressure indicators (nutrients, pesticides, water abstraction, hydromorphological alteration) as specified in Table 1 (for extensive grassland water abstraction was set to 'very low'). Then a cumulative pressure index was calculated for each AFFP by summing up the numerical values of all individual pressure levels divided by the total number of pressures. For each 3km raster cell of Levers et al. (2018), the individual pressure indicators (Figures S1-S4) and the respective cumulative pressure of the different AFFP were mapped, resulting in a pan-European pressure map with relations to agricultural production and presumed freshwater stress.

2.2.2 Relations with river ecological status

To answer our research question, whether accounting for agricultural intensity does improve the correlative strength of agricultural land and ecological status and whether this correlation differs between river types, we quantified the agricultural land use in the different river catchments in two different ways and subsequently assessed the correlative strength with Spearman rank correlations. First, for each river catchment all AFFP were merged to calculate the percentage area in the catchment covered by any type of agricultural land use (ignoring information on their cultivation intensities and respective pressure indices), referred to as 'percentage of agriculture' in the following. Second, the percental cover of the area in the river catchments covered by the different AFFP were weighted based on the previously calculated pressure index and subsequently merged, resulting in what is referred to as 'agricultural intensity index'. The 51,625 river catchments were subdivided based on the twelve different river types of Lyche Solheim et al. (2019), while for river catchments including more than one river type, the river type at the outlet of the catchment was selected. For each of the twelve river types and all river types combined, both agriculture and the agricultural intensity index were related to the ecological status, using Spearman rank correlation, and calculating confidence intervals based on Fisher transformation (Fisher, 1915):

$$95\%\ \mathrm{CI} = \tanh\left(\operatorname{atanh}\rho \pm \frac{1.96}{\sqrt{n-3}}\right)$$

Where ρ is Spearman´s Rho and n is the sample size.

Additionally, we fitted Generalized Linear Mixed Models (GLMMs) with the gamlss package in r (v5.2-0, Rigby & Stasinopoulos, 2005). For this, we first transformed the values of the ecological status ranging between 1 and 5 to values between 0 and 1 using the equation (x-1)/4. Then we set up the GLMMs with a logit link as well as beta distribution, allowing for zero-one inflation (BEINF) to account for the values between zero and one. We built one model with the agricultural intensity index and another model with the percentage of agriculture as fixed effect and used the European member state, to account for possible differences in sampling and the river type (following Lyche Solheim et al., 2019) as random factors. For both models, 70 percent of the data were bootstrapped for 1000 iterations, each to calculate a mean-pseudo-R^2 (from here on referred to as R^2) and standard deviation. The models were checked visually for residual distribution against predicted values, yielding centered averages and symmetrical distributions. These two methods were used to jointly investigate the increase in the correlative strength based on incorporating agricultural intensity of production and presumed freshwater stress.

3. Results & Discussion

3.1 Areas of Farming-induced Freshwater Pressures (AFFP)

The delineated 20 AFFP portray the diversity of the agricultural land system in Europe. While Northern Europe is mostly covered by forests, the other areas are mainly covered by AFFP of different size and character. The AFFP feature a broad range of pressure-profiles demonstrating clear geographic distributions that reflect the various agricultural practices in Europe (Table 2). The cumulative pressure index shows highest values for the Mediterranean intensive cropland, located in specific regions of Greece, Italy and Spain (Figure 1). The intensively farmed regions of the western part of the Temperate region, including France, Germany, Denmark, and UK, also feature high index levels. Agricultural areas with low pressure are located in Eastern European countries and the Northern and Highland regions.

Highest nitrogen surplus levels (Figure S1) are found in the western part of the Temperate region (France, Germany, Denmark, England), which is dominated by intensive agriculture (Rega et al., 2020). Low levels are found in the Northern and Highland regions as well as the eastern part of the Temperate region, where low-intensity agriculture is dominating, related to the different socioeconomic history of collectivisation and later de-intensification, with smaller nutrient application rates compared to western Europe (Rozelle & Swinnen, 2004; Jepsen et al., 2015). For the pesticide pressure (Figure S2), the highest levels are found in the Mediterranean region, where higher application rates may be applied due to the warmer, pest-friendly climate (Ippolito et al., 2015; Tang et al., 2021). A high pesticide pressure level was also found in the Temperate region, which is likely related to intensive agricultural practices (for the same reasons as for nitrogen surplus) and the frequency of crops that require high pesticide applications in conventional agriculture. Intensive agricultural practices are mostly associated with high hydromorphological pressures (Figure S3) examined by the degree of agricultural land use in the floodplain. Agriculture is frequently located in floodplains, because of favourable soil fertility and water availability, the latter particularly relevant in the Mediterranean region (Verhoeven & Setter, 2010; Auerswald et al., 2019). The agricultural water abstraction pressure (Figure S4) was almost exclusively observed in the Mediterranean region, where the dry climatic conditions more often lead to water scarcity stress (Metzger et al., 2005; Zal et al., 2021). This regional phenomenon contributes to a pronounced multi-pressure situation in the Mediterranean (Segurado et al., 2018). Livestock farming is associated with rather low local impact on freshwater ecosystems, except in the case of Temperate high intensity agriculture, where it is largely characterised by factory farming (van Arendonk & Liinamo, 2003). However, this must be regarded with care, as for instance micropollutants such as pharmaceuticals originating from livestock (Osorio et al., 2016) are not regarded in this study and livestock pressure is spread to other regions by relying on imported concentrated feed, causing high greenhouse gas emissions and water usage (Gerber et al., 2010). Overall, the farming intensities of the underlying archetypes (Levers et al., 2018), are well reflected in the individual and cumulative pressure levels of the different AFFP, suggesting a clear relationship between agricultural intensity and the environmental pressures exerted to freshwaters.

Table 2: The 20 Areas of Farming-induced Freshwater Pressures (AFFP) ranked according to the cumulative agricultural pressure index (high to low), including their pressure-profiles and number of river catchments (covered ≥66.6% by a single AFFP) used for the calculation; note that the water abstraction pressure levels were not assigned to the AFFP 'Extensive grassland' (total number of FECs = 17,099).

AFFP	**Pressure index**	**Nitrogen pressure**	**Pesticide pressure**	**Hydromorphological pressure**	**Water abstraction**	**Number of river catchments**
Mediterranean high intensity cropland	3.75	*medium*	*high*	*high*	*high*	68
Mediterranean medium intensity cropland	3.50	*low*	*high*	*high*	*high*	689
Temperate high intensity cropland	3.25	*high*	*high*	*medium*	*low*	2202
Mediterranean low intensity cropland	3.25	*very low*	*high*	*high*	*high*	377
Mediterranean large-scale permanent cropland	3.00	*very low*	*high*	*medium*	*high*	823
Temperate high intensity livestock farming	3.00	*high*	*low*	*high*	*very low*	1623
Temperate medium intensity cropland	3.00	*medium*	*medium*	*high*	*low*	2112
Temperate high intensity mosaic farming	2.75	*high*	*medium*	*medium*	*very low*	674
Mediterranean livestock farming	2.50	*low*	*medium*	*low*	*medium*	79

Northern and Highland high intensity cropland	2.25	*medium*	*low*	*medium*	*very low*	183
Northern and Highland medium intensive cropland	2.25	*medium*	*low*	*medium*	*very low*	83
Temperate low intensity livestock	2.25	*medium*	*low*	*low*	*low*	1012
Temperate low intensity cropland	2.25	*low*	*medium*	*medium*	*very low*	910
Northern and Highland low intensity cropland	2.00	*low*	*low*	*medium*	*very low*	48
Temperate low intensity mosaic farming	2.00	*low*	*low*	*low*	*low*	1199
Mediterranean mosaic farming	1.50	*very low*	*very low*	*very low*	*medium*	292
Northern Highland mosaic farming	1.50	*low*	*very low*	*very low*	*low*	339
Extensive grassland	1.00	*very low*	*very low*	*very low*	-	3037
Northern and Highland livestock farming	1.00	*very low*	*very low*	*very low*	*very low*	180
Fallow Farmland	1.00	*very low*	*very low*	*very low*	*very low*	1169

3.2 Accounting for agricultural intensity enhances the link between agricultural land use and the ecological status.

The correlative strength of the relationship between the ecological status and agriculture is higher when considering the agricultural intensity index compared to simply using the percentage of agriculture in a catchment (Figure 2). For all river types, the correlative strength of the intensity index showed higher correlations, and confidence intervals did not overlap between the intensity index and percentage of agriculture for any of the river types except very large rivers and siliceous large lowland rivers. For all river types combined, the correlation of the intensity index of Spearman $\rho = 0.30$ was nearly twice as high compared to simply using the percentage of agriculture (ρ = 0.16). Similarly, the results of the GLMMs showed clear differences between the model with the intensity index and using the percentage of agriculture ($R^2 = 0.18 \pm 0.00$ (SD) and 0.15 ± 0.01 (SD), respectively). The correlations and results from the GLMMs jointly show the high potential of accounting for the agricultural intensity when assessing agricultural effects at the continental scale, upscaling findings at smaller scales (e.g. Schürings et al., 2024a). Regarding the different river types, the strongest correlative strength was observed for the mountain rivers and the Mediterranean intermittent rivers with Spearman ρ = 0.39 and 0.41, respectively, while it was small for very large rivers and large rivers in the lowlands. Highland and glacial rivers featured only a small to medium correlation of ρ = 0.17 and 0.07, respectively. Small calcareous lowland rivers and small siliceous mid-altitude rivers feature strongest differences in correlative strength between the two ways to quantify agriculture, where the agricultural intensity index´s correlation with the ecological status exceeded the percentage of agriculture´s more than five times. The higher correlations for mountain rivers can be understood in comparison to lowland rivers, which no longer exhibit a full gradient of environmental conditions due to ubiquitous, long-lasting and often intensive agricultural land use (Feld, 2013). The higher correlation for the intermittent Mediterranean rivers is likely related to the combined effects of water stress with the other pressures (Skoulikidis et al., 2017; Karaouzas et al., 2018). The mostly low correlations for large rivers indicate the generally lesser influence of agricultural land use in larger catchments (Li et al., 2018).

3.3 Methodological reflections

The correlative strength of the agricultural intensity index clearly exceeded findings from prior studies with Europe-wide datasets. Lemm et al. (2021), for instance, found a correlation of agriculture with the ecological status of ρ = 0.23. This highlights the potential of considering the different agricultural types. The higher correlations of some smaller-scale analyses (e.g. Schürings et al., 2024a), with relations between agricultural intensity and the ecological status of up to $R^2 = 0.43$ at the German scale, however, seem to indicate a shortcoming in our coarse-scale data, with an average river catchment size of 60 km². Reasons include a European land use grid of 3km compared to the German land use data of 10m resolution, and less comparability of the ecological status assessment between different EU member states, but also missing information on local differences (e.g. organic farming promoting lower pesticide burden; Geisen et al., 2021). As our pressure data was modelled for each river catchment at the European scale, we had to improve the spatial accuracy by assigning pressure data only to those river catchments, for which a given AFFP covered ≥ 66.6 percent of the river catchment. This approach may have led to the misestimation of some AFFP that are more fragmentary distributed (such as livestock farming). In addition, the results might be affected by a temporal mismatch between the underlying data with land use data from 2006 (Levers et al., 2018), pressure data from 2011-2015 and ecological status data from 2006-2015. However, land use and river ecology do not seem to have been subject to strong dynamics between 2006 and 2015, so that no substantial effects of a possible temporal mismatch can be assumed (EEA, 2012; EEA, 2020; Ivits-Wasser et al. 2019). Our analyses were thus based on the best available continental data sets for the synthesis of agriculture, its pressures on freshwaters and the corresponding ecological effect. Once fine resolution data is available for the whole of Europe, future studies will certainly benefit from inclusion of additional agricultural pressures, more sophisticatedly derived pressures from more detailed input data (van Gils et al., 2020), more detailed information on soil conditions (Dobbie & Smith, 2003), slope (Cambien et al., 2020), riparian vegetation (Palt et al., 2023), sampling site based river network analysis (Büttner et al., 2020), and improved river typologies (Jupke et al., 2022; Jupke et al., 2023).

4. Conclusion

Based on existing landscape typologies, we delineated 20 Areas of Farming-induced Freshwater Pressures (AFFP), which show characteristic agriculture-driven pressure profiles. Unlike previous large-scale studies that distinguished only between few broad categories of agricultural land use to investigate effects on freshwaters (e.g. Wasson et al., 2010), the AFFP are an amalgamation of the agricultural archetypes and large European biogeographic regions, which explicitly incorporate production intensity (nitrogen input and biomass output) and feature corresponding environmental pressure levels. They can serve as a tool to compare case studies across different agricultural regions, identify pressure hotspots caused by agricultural activity, and allow to relate agricultural intensity and environmental pressures. The results identify regions of large agricultural pressures for river biodiversity and show that accounting for agricultural intensity can strongly increase the correlative strength. This information can be used to provide advice for the pan-European management of freshwater ecosystems, in particular suggesting regions for extensification or sustainable intensification along concepts of land-sharing and land-sparing (Postma-Blaauw et al., 2010; Rockström et al., 2017; Grass et al., 2019). It also renders urgently needed evidence for EU-wide environmental policies such as reforming the Common Agricultural Policy to halt biodiversity loss (Pe’er et al., 2022) with an effective post-2027 renewal. The high environmental burden of agricultural land use and the strong increase in correlative strength when accounting for agricultural intensity highlights the urgent need for agricultural transition to more sustainable practices such as organic farming or permaculture, particularly in regions with high agricultural pressures.

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Data availability

The data that support the findings are openly available in Zenodo at https://zenodo.org/record/8141684

Figures

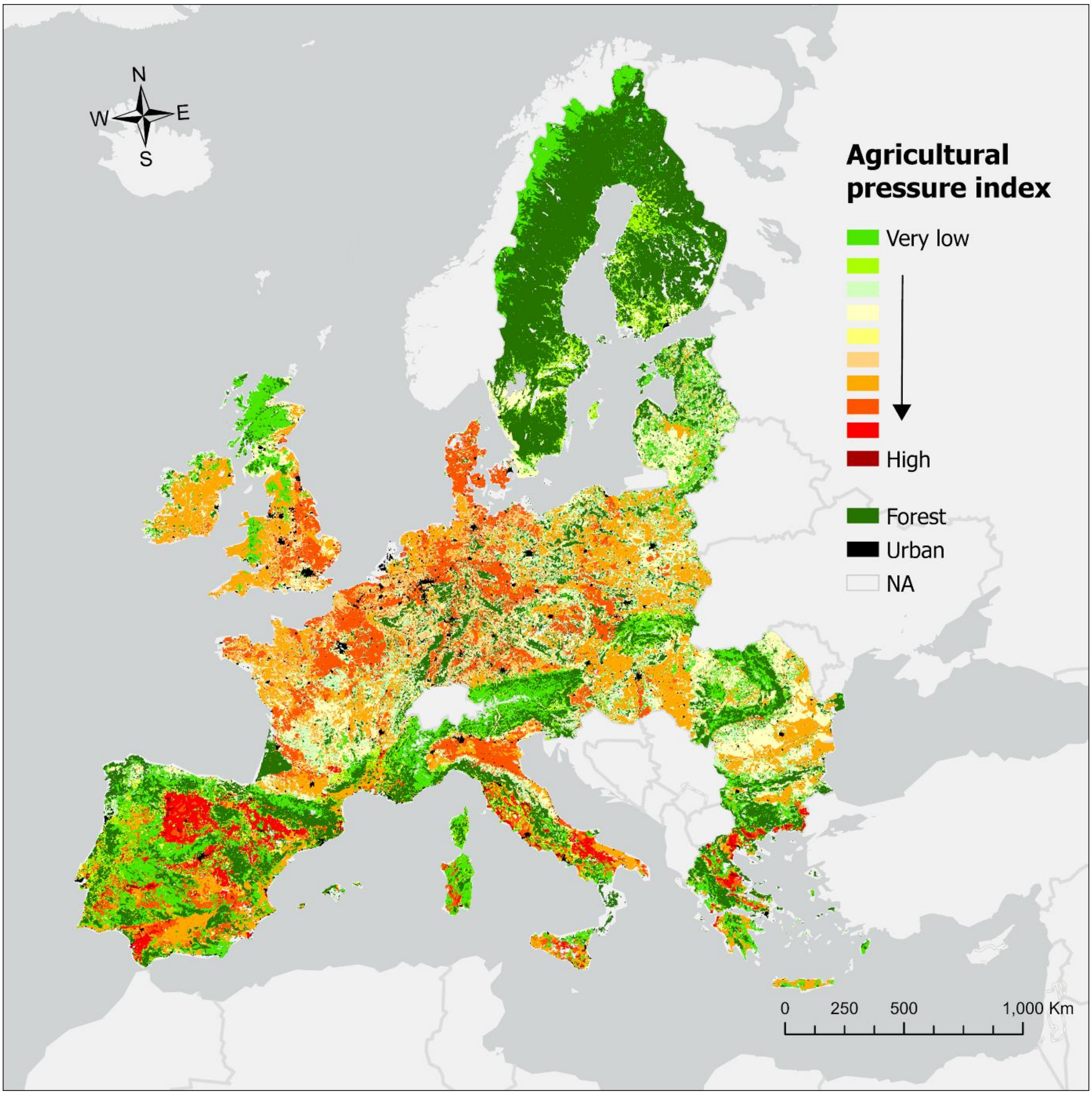

Figure 1: Cumulative agricultural pressure index classifying the average intensity of multiple agricultural pressures (nutrients, pesticides, water abstraction, hydromorphological alterations) on water bodies in Europe at the 3km scale.

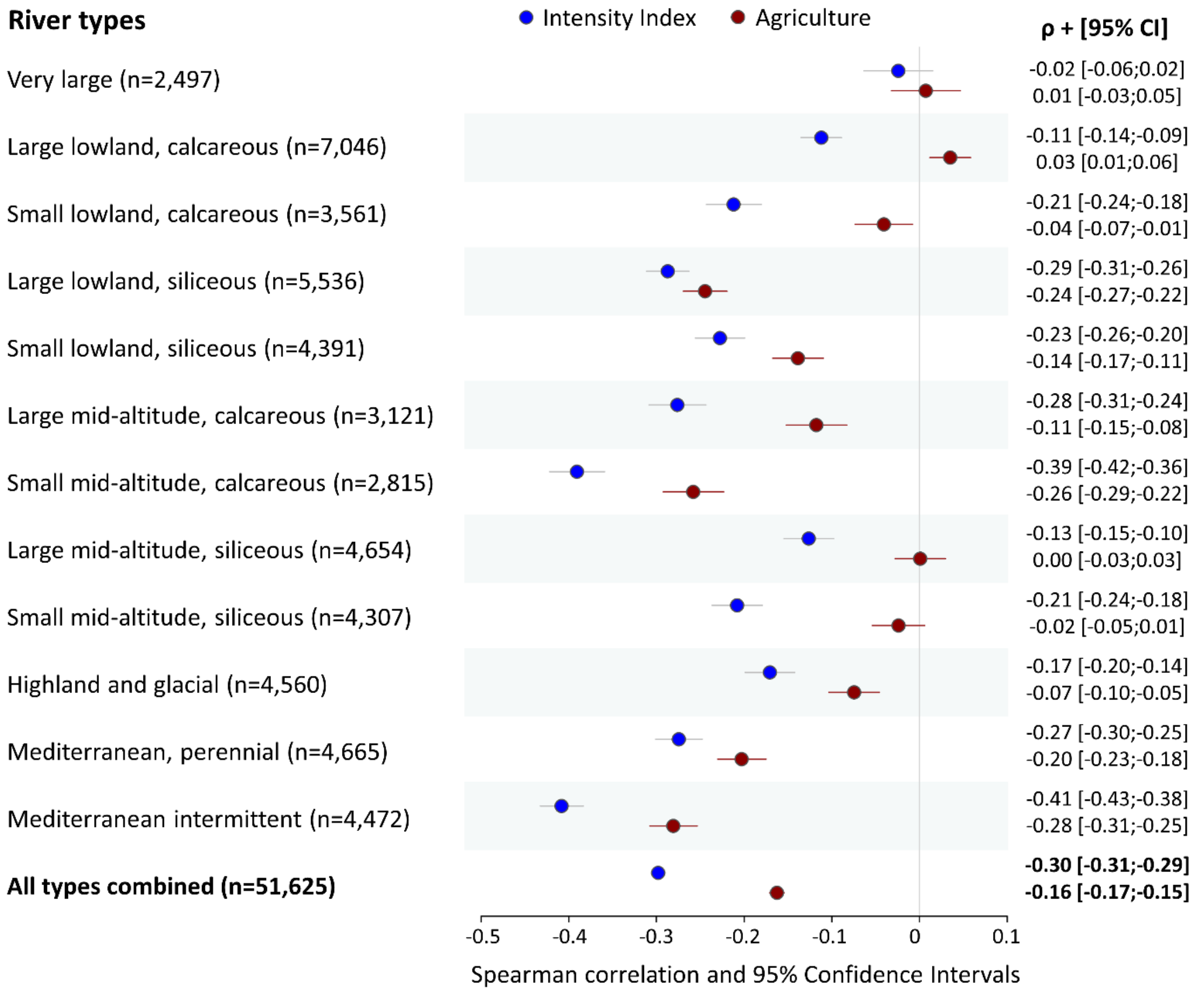

Figure 2: Spearman rank correlation between the ecological status and the two ways to quantify agriculture, the agricultural intensity index in blue and the percentage of agriculture in dark red, across the twelve different river types and all rivers combined. Shown are Spearman ρ and 95% confidence intervals.

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Supplementary Material to

River ecological status is shaped by agricultural land use intensity across Europe

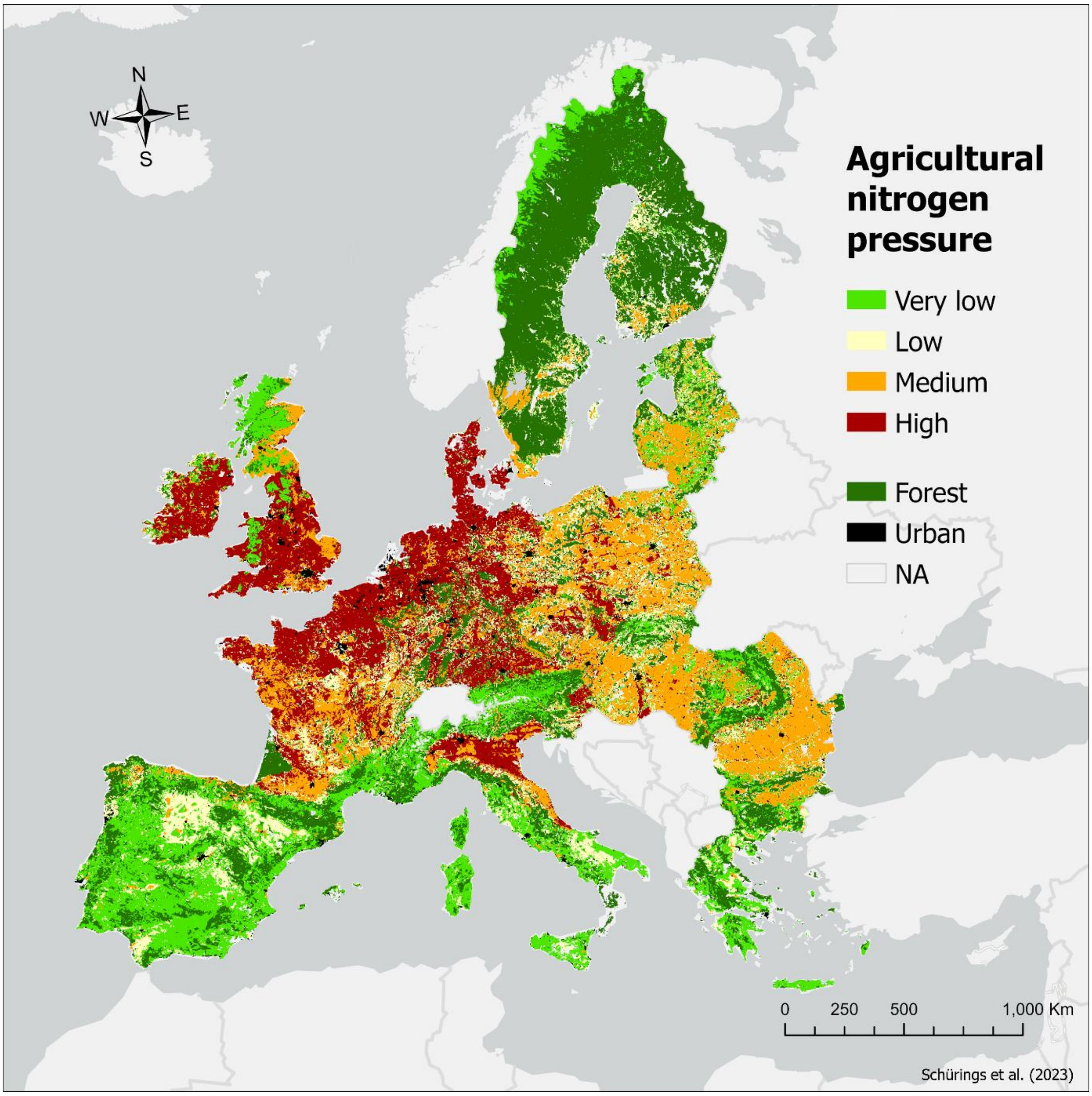

Figure S*1*: Average intensity of agricultural nitrogen pressure on water bodies in Europe and the 3km scale.

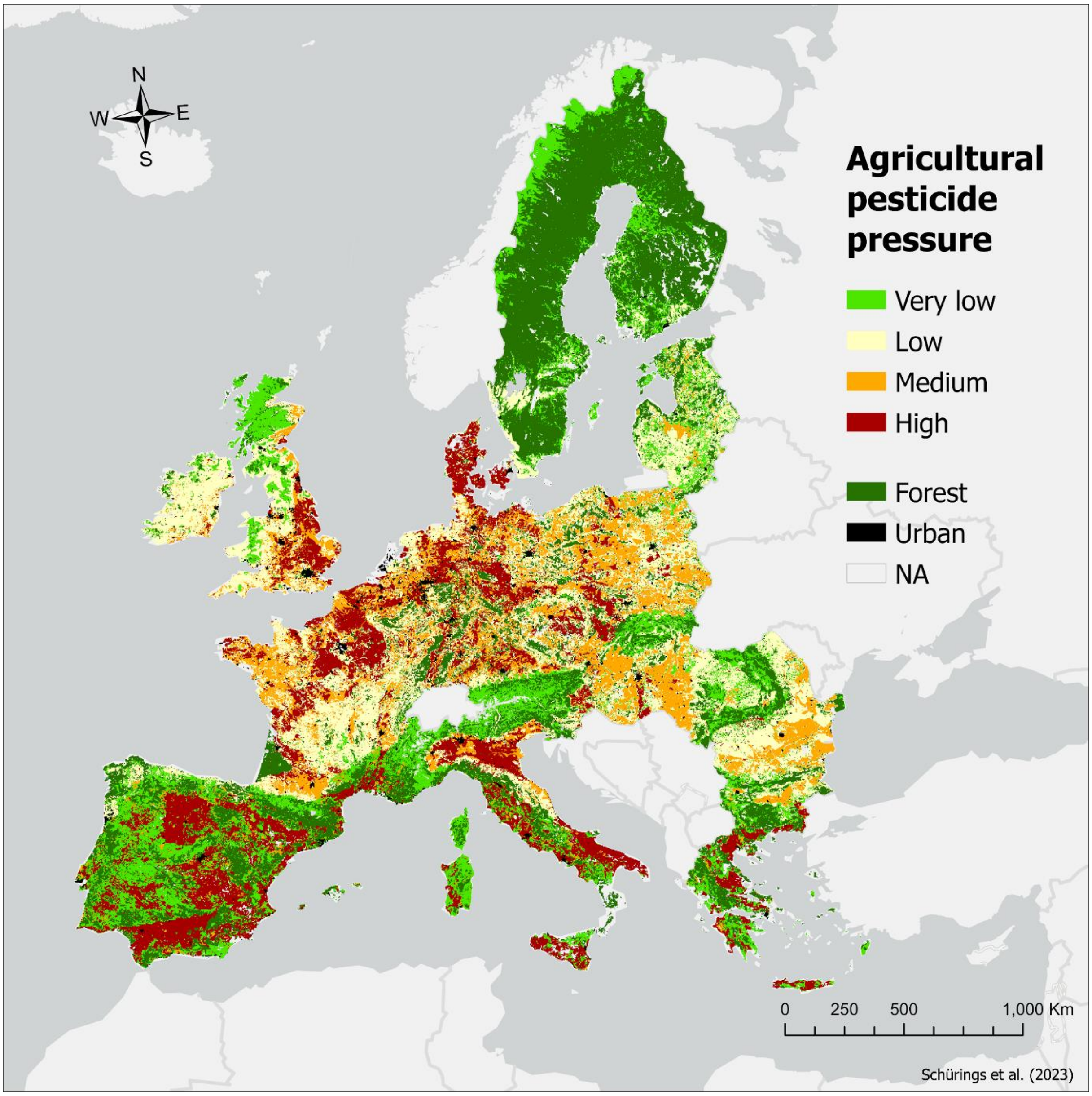

Figure S2: Average intensity of agricultural pesticide pressure on water bodies in Europe and the 3km scale.

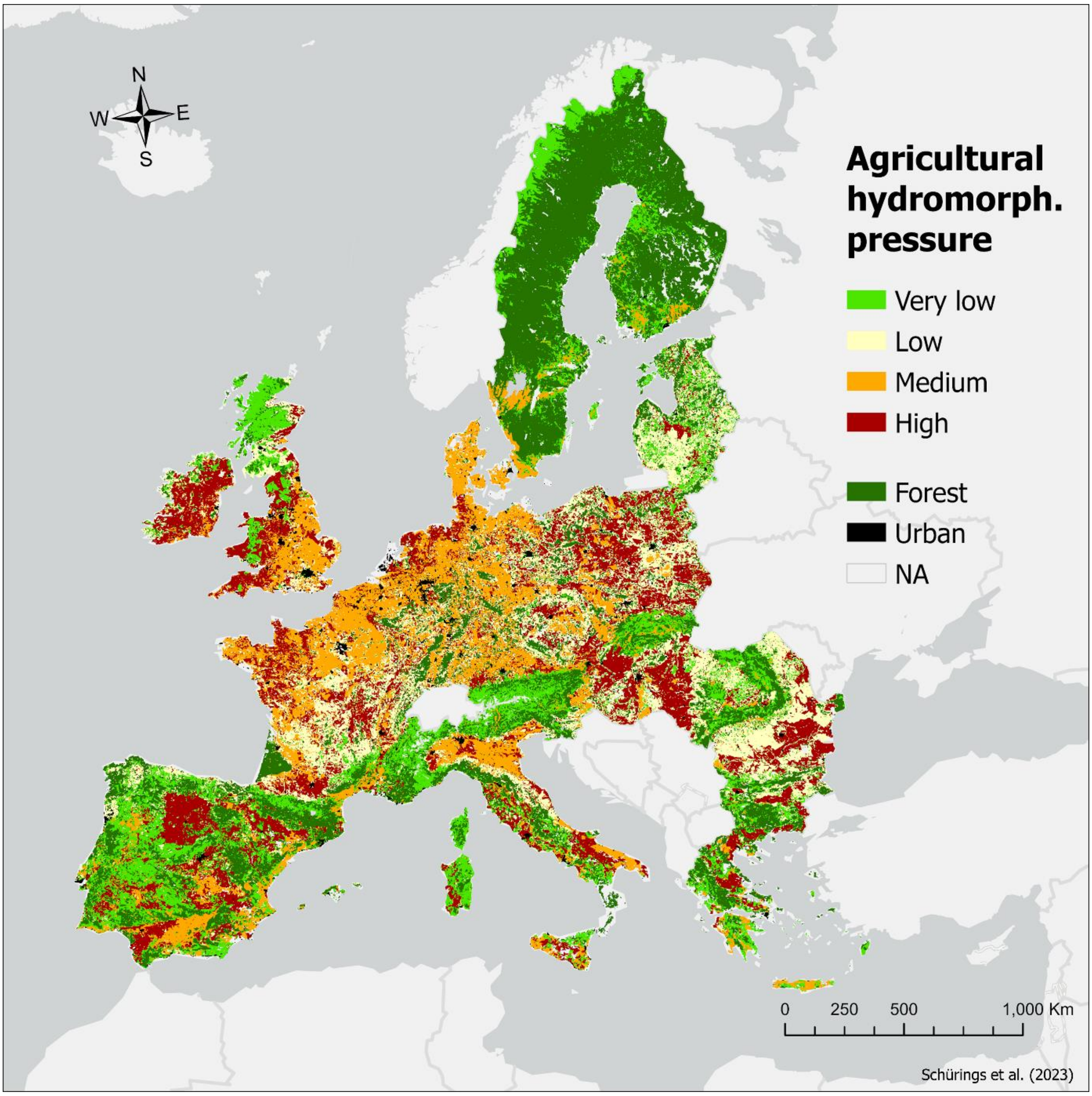

Figure S3: Average intensity of agricultural hydromorphological pressure on water bodies in Europe and the 3km scale.

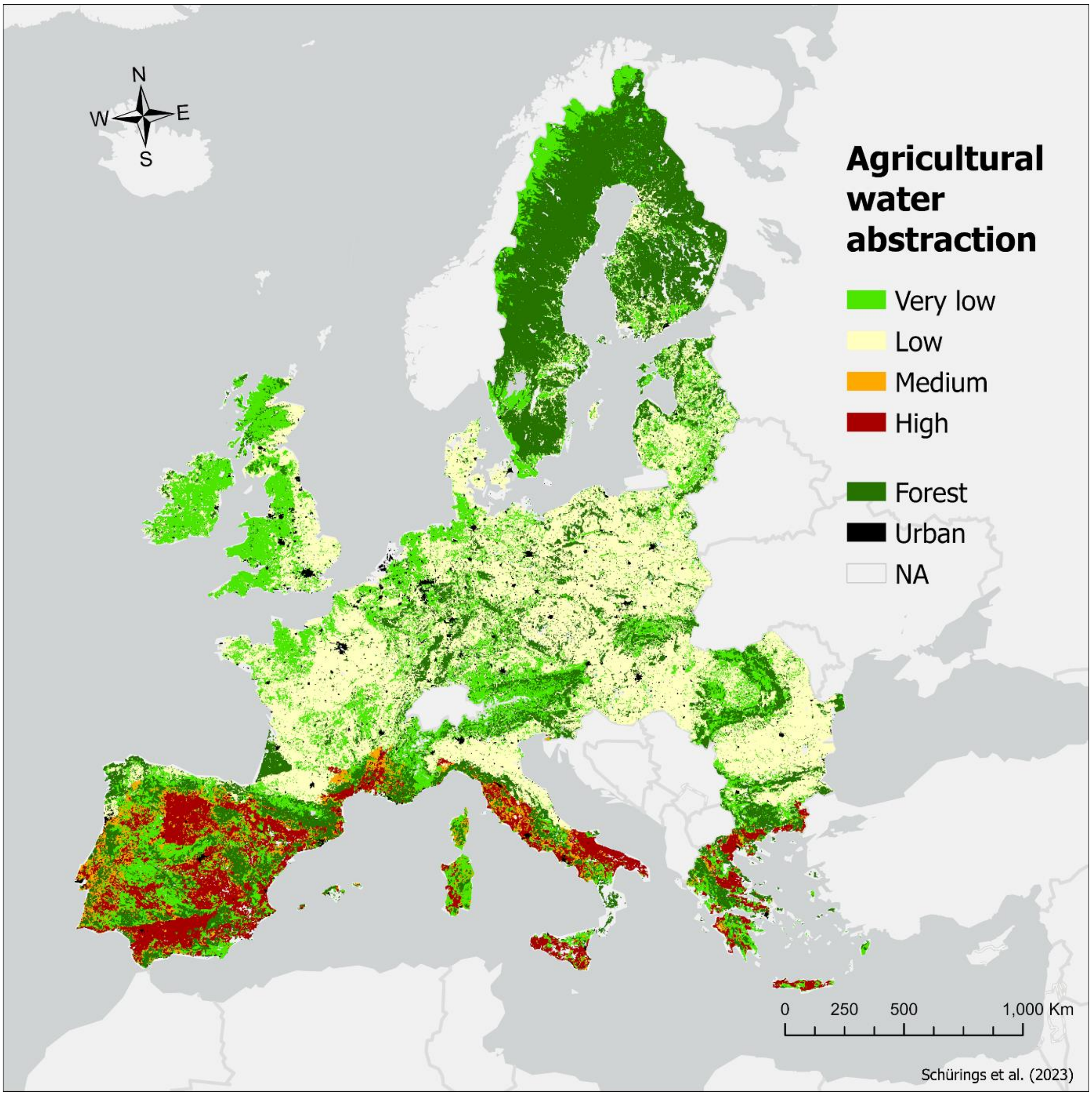

Figure S4: Average intensity of agricultural water abstraction on water bodies in Europe and the 3km scale.

Table S1: Pesticides (n = 332) underlying the “real-life” mixture exposure scenario for each FEC as a basis for the mixture toxic pressure metric msPAF.

CAS Number	Name
101200-48-0	Tribenuron-methyl
104206-82-8	Mesotrione
112410-23-8	Tebufenozide
119168-77-3	Tebufenpyrad

123312-89-0	Pymetrozine
131983-72-7	Triticonazole
137-26-8	Thiram
149961-52-4	Dimoxystrobin
16672-87-0	Ethephon
1861-40-1	Benfluralin
19937-59-8	Metoxuron
2310-17-0	Phosalone
2540-82-1	Formothion
29973-13-5	Ethiofencarb
33629-47-9	Butralin
39515-41-8	Fenpropathrin
51218-45-2	Metolachlor
55335-06-3	Triclopyr
5915-41-3	Terbuthylazine
668-34-8	Fentin
71626-11-4	Benalaxyl
76674-21-0	Flutriafol
82560-54-1	Benfuracarb
86-50-0	Azinphos-methyl
950-37-8	Methidathion
10605-21-7	Carbendazim
10453-86-8	Resmethrin
132-66-1	Naptalam
3689-24-5	Sulfotep
15299-99-7	Napropamide
49866-87-7	Difenzoquat
67564-91-4	Fenpropimorph
95617-09-7	Fenoxaprop
101205-02-1	Cycloxydim
10552-74-6	Nitrothal-isopropyl
113136-77-9	Cyclanilide
119446-68-3	Difenoconazole
123-33-1	Maleic hydrazide

133-06-2	Captan
138261-41-3	Imidacloprid
150-68-5	Monuron
16752-77-5	Methomyl
18691-97-9	Methabenzthiazuron
2032-65-7	Methiocarb
23103-98-2	Pirimicarb
2593-15-9	Etridiazole
300-76-5	Naled
33693-04-8	Terbumeton
39807-15-3	Oxadiargyl
5234-68-4	Carboxin
55-38-9	Fenthion
60168-88-9	Fenarimol
67129-08-2	Metazachlor
72178-02-0	Fomesafen
786-19-6	Carbophenothion
82657-04-3	Bifenthrin
87820-88-0	Tralkoxydim
95266-40-3	Trinexapac-Ethyl
470-90-6	Chlorfenvinphos
87392-12-9	S-Metolachlor
298-00-0	Parathion-Methyl
4685-14-7	Paraquat
15457-05-3	Fluorodifen
50594-66-6	Acifluorfen
74738-17-3	Fenpiclonil
100646-51-3	Quizalofop-P-ethyl
101-21-3	Chlorpropham
1071-83-6	Glyphosate
113158-40-0	Fenoxaprop-P
1194-65-6	Dichlobenil
125116-23-6	Metconazole
133-07-3	Folpet

139-40-2	Propazine
153719-23-4	Thiamethoxam
1689-83-4	Ioxynil
188425-85-6	Boscalid
21087-64-9	Metribuzin
23135-22-0	Oxamyl
26225-79-6	Ethofumesate
301-12-2	Oxydemeton-methyl
34014-18-1	Tebuthiuron
40487-42-1	Pendimethalin
52-68-6	Trichlorfon
55512-33-9	Pyridate
60207-90-1	Propiconazole
67306-00-7	Fenpropidin
72490-01-8	Fenoxycarb
79277-27-3	Thifensulfuron-methyl
83055-99-6	Bensulfuron-methyl
886-50-0	Terbutryn
95465-99-9	Cadusafos
52315-07-8	Cypermethrin	Zeta-cypermethrin
62-73-7	Dichlorvos
314-40-9	Bromacil
5902-51-2	Terbacil
18181-80-1	Bromopropylate
51276-47-2	Glufosinate
76578-12-6	Quizalofop
110488-70-5	Dimethomorph
101-42-8	Fenuron
107534-96-3	Tebuconazole
114-26-1	Propoxur
120068-37-3	Fipronil
126833-17-8	Fenhexamid
13360-45-7	Chlorbromuron
140923-17-7	Iprovalicarb

15545-48-9	Chlorotoluron
1689-84-5	Bromoxynil
1897-45-6	Chlorothalonil
2164-17-2	Fluometuron
23560-59-0	Heptenophos
2642-71-9	Azinphos-ethyl
30560-19-1	Acephate
34123-59-6	Isoproturon
41083-11-8	Azocyclotin
52888-80-9	Prosulfocarb
5598-13-0	Chlorpyrifos-Methyl
60-51-5	Dimethoate
67747-09-5	Prochloraz
7287-19-6	Prometryn
79538-32-2	Tefluthrin
83164-33-4	Diflufenican
88671-89-0	Myclobutanil
96489-71-3	Pyridaben
58-89-9	Lindane
75-99-0	Dalapon
555-37-3	Neburon
7003-89-6	Chlormequat
24579-73-5	Propamocarb
51707-55-2	Thidiazuron
76738-62-0	Paclobutrazol
139528-85-1	Metosulam
101463-69-8	Flufenoxuron
1085-98-9	Dichlofluanid
114369-43-6	Fenbuconazole
120928-09-8	Fenazaquin
13071-79-9	Terbufos
134098-61-6	Fenpyroximate
141776-32-1	Sulfosulfuron
156052-68-5	Zoxamide

1698-60-8	Chloridazon
19044-88-3	Oryzalin
21725-46-2	Cyanazine
23564-05-8	Thiophanate-methyl
28249-77-6	Thiobencarb
3060-89-7	Metobromuron
34256-82-1	Acetochlor
41394-05-2	Metamitron
52918-63-5	Deltamethrin
563-12-2	Ethion
63-25-2	Carbaryl
68694-11-1	Triflumizole
731-27-1	Tolylfluanid
79622-59-6	Fluazinam
834-12-8	Ametryn
90717-03-6	Quinmerac
99105-77-8	Sulcotrione
94-74-6	MCPA
82-68-8	Quintozene
1134-23-2	Cycloate
7786-34-7	Mevinphos
26644-46-2	Triforine
58138-08-2	Tridiphane
77732-09-3	Oxadixyl
141517-21-7	Trifloxystrobine
1014-69-3	Desmetryn
1113-02-6	Omethoate
115-32-2	Dicofol
121552-61-2	Cyprodinil
13121-70-5	Cyhexatin
13457-18-6	Pyrazophos
142459-58-3	Flufenacet
1563-66-2	Carbofuran
1702-17-6	Clopyralid

1912-24-9	Atrazine
2212-67-1	Molinate
23947-60-6	Ethirimol
28434-01-7	Bioresmethrin
330-54-1	Diuron
34681-10-2	Butocarboxim
41483-43-6	Bupirimate
53112-28-0	Pyrimethanil
56-38-2	Parathion
63284-71-9	Nuarimol
6923-22-4	Monocrotophos
732-11-6	Phosmet
79983-71-4	Hexaconazole
83657-24-3	Diniconazole
919-86-8	Demeton-S-methyl
99129-21-2	Clethodim
67375-30-8	Alpha-cypermethrin
94-81-5	MCPB
1420-07-1	Dinoterb
10311-84-9	Dialifos
27314-13-2	Norflurazon
61213-25-0	Flurochloridone
78587-05-0	Hexythiazox
149979-41-9	Tepraloxydim
2164-08-1	Lenacil
10265-92-6	Methamidophos
111479-05-1	Propaquizafop
116-06-3	Aldicarb
122-14-5	Fenitrothion
131341-86-1	Fludioxonil
13593-03-8	Quinalphos
143390-89-0	Kresoxim-Methyl
1582-09-8	Trifluralin
173159-57-4	Foramsulfuron

1918-00-9	Dicamba
22224-92-6	Fenamiphos
23950-58-5	Propyzamide
2921-88-2	Chlorpyrifos
330-55-2	Linuron
35367-38-5	Diflubenzuron
42874-03-3	Oxyfluorfen
55179-31-2	Bitertanol
57018-04-9	Tolclofos-methyl
640-15-3	Thiometon
69377-81-7	Fluroxypyr
74051-80-2	Sethoxydim
80-33-1	Chlorfenson
84087-01-4	Quinclorac
93-65-2	Mecoprop
99-30-9	Dicloran
87674-68-8	Dimethenamid
94-82-6	2,4-DB
1967-16-4	Chlorbufam
13171-21-6	Phosphamidon
29091-05-2	Dinitramine
62850-32-2	Fenothiocarb
79241-46-6	Fluazifop-P-butyl
161050-58-4	Methoxyfenozide
3813-05-6	Benazolin
102851-06-9	Tau-fluvalinate
111988-49-9	Thiacloprid
116255-48-2	Bromuconazole
122-34-9	Simazine
131807-57-3	Famoxadone
136426-54-5	Fluquinconazole
145701-23-1	Florasulam
15972-60-8	Alachlor
1746-81-2	Monolinuron

1918-16-7	Propachlor
2275-23-2	Vamidothion
24017-47-8	Triazophos
29232-93-7	Pirimiphos-Methyl
33089-61-1	Amitraz
35554-44-0	Imazalil
43121-43-3	Triadimefon
55219-65-3	Triadimenol
57646-30-7	Furalaxyl
64902-72-3	Chlorsulfuron
69806-34-4	Haloxyfop
74070-46-5	Aclonifen
81777-89-1	Clomazone
841-06-5	Methoprotryne
94125-34-5	Prosulfuron
94-75-7	2,4-D
76-87-9	Fentin hydroxide
115-29-7	Endosulfan
2439-01-2	Quinomethionate
13194-48-4	Ethoprophos
29104-30-1	Benzoximate
66215-27-8	Cypromazine
81335-37-7	Imazaquin
208465-21-8	Mesosulfuron-methyl
10004-44-1	Hymexazol
103055-07-8	Lufenuron
111991-09-4	Nicosulfuron
116-29-0	Tetradifon
122453-73-0	Chlorfenapyr
13181-17-4	Bromofenoxim
13684-56-5	Desmedipham
14816-18-3	Phoxim
161326-34-7	Fenamidone
175013-18-0	Pyraclostrobin

1929-77-7	Vernolate
22781-23-3	Bendiocarb
24934-91-6	Chlormephos
298-02-2	Phorate
333-41-5	Diazinon
36734-19-7	Iprodione
50471-44-8	Vinclozolin
55283-68-6	Ethalfluralin
57837-19-1	Metalaxyl
66230-04-4	Esfenvalerate
70124-77-5	Flucythrinate
74223-64-6	Metsulfuron-methyl
82097-50-5	Triasulfuron
85509-19-9	Flusilazole
94361-06-5	Cyproconazole
71751-41-2	Abamectin
51630-58-1	Fenvalerate
120-36-5	Dichlorprop
2764-72-9	Diquat
14214-32-5	Difenoxuron
33820-53-0	Isopropalin
66332-96-5	Flutolanil
83121-18-0	Teflubenzuron
101-05-3	Anilazine
104040-78-0	Flazasulfuron
112281-77-3	Tetraconazole
118134-30-8	Spiroxamine
122931-48-0	Rimsulfuron
131860-33-8	Azoxystrobin
13684-63-4	Phenmedipham
148-79-8	Thiabendazole
163515-14-8	Dimethenamid-P
17804-35-2	Benomyl
19666-30-9	Oxadiazon

2303-17-5	Tri-allate
25311-71-1	Isofenphos
298-04-4	Disulfoton
3347-22-6	Dithianon
3878-19-1	Fuberidazole
50563-36-5	Dimethachlor
55285-14-8	Carbosulfan
57966-95-7	Cymoxanil
66246-88-6	Penconazole
709-98-8	Propanil
759-94-4	EPTC
82558-50-7	Isoxaben
86479-06-3	Hexaflumuron
944-22-9	Fonofos
25057-89-0	Bentazone
70630-17-0	Metalaxyl-M
122-42-9	Propham
2797-51-5	Quinoclamine
15165-67-0	Dichlorprop-P
42576-02-3	Bifenox
66841-25-6	Tralomethrin
91465-08-6	Lambda-Cyhalothrin

Table S2: Allocation of the 36 'regionalized archetypes' based on the archetypes of Levers et al. (2018) and the biogeographical regions of the Habitats Directive (Roeckaerts, 2002) to the 20 Areas of Farming-driven Freshwater Impacts.

Regionalized Archetypes	**Areas of Farming-induced Freshwater Pressures (AFFP)**
Temperate high intensity cropland	Temperate high intensity cropland
Temperate high intensity arable cropland	
Temperate large-scale permanent cropland	
Temperate medium intensity arable cropland	Temperate medium intensity cropland
Temperate low intensity arable cropland	Temperate low intensity cropland
Temperate high intensity livestock farming	Temperate high intensity livestock farming
Temperate medium intensity livestock farming	
Temperate low intensity livestock farming	Temperate low intensity livestock farming
Temperate high intensity agricultural mosaic	Temperate high intensity mosaic farming
Temperate low-intensity mosaic	Temperate low intensity mosaic farming
Northern and Highland high intensity cropland	Northern and Highland high intensity cropland
Northern and Highland large-scale permanent cropland	
Northern and Highland high intensity arable cropland	
Northern and Highland medium intensity arable cropland	Northern and Highland medium intensity cropland
Northern and Highland low intensity arable cropland	Northern and Highland low intensity cropland
Northern and Highland high intensity livestock farming	Northern and Highland livestock farming
Northern and Highland medium intensity livestock farming	
Northern and Highland low intensity livestock farming	

Contributions to publications

Cumulative Dissertation of Christian Schürings

Author contributions

Titel: *River ecological status is shaped by agricultural land use intensity across Europe*

Authors: Schürings, C., Globevnik, L., Lemm, J. U., Psomas, A., Snoj, L., Hering, D., & Birk, S.

Contributions:

- Conception – 60%
- Conduction of experimental work – not applicable
- Data analysis – 70%
- Species identification – not applicable
- Statistical analysis – 100%
- Writing the manuscript – 90%
- Revision of the manuscript – 60%

Signature of the Doctoral Candidate Signature of the Doctoral Supervisor

Chapter 3

Effects of agriculture on river biota differ between crop types and organism groups

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Effects of agriculture on river biota differ between crop types and organism groups

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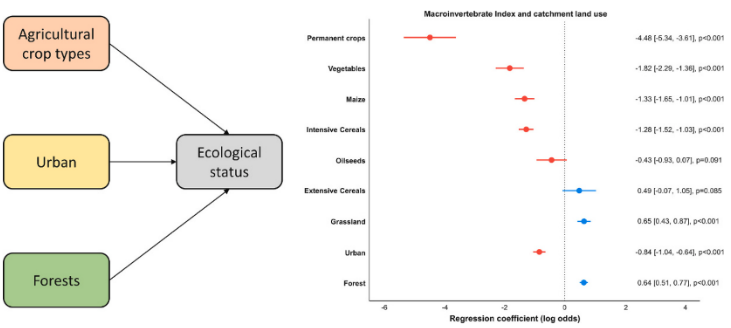
HIGHLIGHTS

- Effects of catchment agriculture exceed urban effects on river biota in magnitude.
- Discrimination between crop types reveals clear differences in biota response.
- Macroinvertebrates are most strongly impaired by pesticide intensive crops.
- Diatoms respond most strongly to the fertilization intensity of crop types.
- Macrophyte response is less clear and likely depending on river hydromorphology.

GRAPHICAL ABSTRACT



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ABSTRACT

While the general effects of agricultural land use on riverine biota are well documented, the differential effects of specific crop types on different riverine organism groups, remain largely unexplored. Here we used recently published land use data distinguishing between specific crop types and a Germany-wide dataset of 7748 sites on the ecological status of macroinvertebrates, macrophytes and diatoms and applied generalized linear mixed models to unravel the associations between land use types, crop types, and the ecological status. For all organism groups, associations of specific crop types with biota were stronger than those of urban land use. For macroinvertebrates and macrophytes, strong negative associations were found for pesticide intensive permanent crops, while intensively fertilized crops (maize, intensive cereals) affected diatoms most. These differential associations highlight the importance of distinguishing between crop types and organism groups and the urgency to buffer rivers against agricultural stressors at the catchment scales and to expand sustainably managed agriculture.

1. Introduction

Agriculture is constantly intensifying to fulfill the needs of the world's growing population (Foley et al., 2005), a process that is accompanied by massive biodiversity decline. Besides terrestrial mammals, birds (Joppa et al., 2016) and flying insects (Hahn et al., 2015), the biota of rivers draining agricultural areas is also strongly impaired (Schürings et al., 2022).

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Agriculture is frequently considered the most important driver for the deterioration of river biota (e.g. Wolfram et al., 2021). Calls for reducing agricultural impacts on rivers are manifold, particularly in parts of Europe where the majority of rivers do not meet the quality targets set by legislation; e.g., in Germany about 90 % of the river water bodies fail to meet the target of 'good ecological status according to the EU Water-Framework Directive' (UBA, 2022). On the global scale, the Convention on Biological Diversity adopted the need of pollution reduction and agricultural transition in targets 7 and 10 of the Kunming-Montreal global biodiversity framework (CBD, 2022). The implementation of targeted environmental protection measures, however, is restricted by complex and contradictory legislation, land availability, and also because large-scale studies on stressors related to specific crop types and how these agricultural stressors affect different organism groups are rare, which would allow to draw more general conclusions for larger areas.

The first link in this cause-effect chain has already been investigated, but mainly in small-scale experimental studies. They found clear differences in stress on freshwaters arising from different crop types. While extensive grassland exerts less stress on river biota (Blake et al., 2012), corn farming close to rivers causes high nutrient and fine sediment influx (Secchi et al., 2011; Jwaideh et al., 2022) and permanent crops like orchards, vineyards, and vegetables are known for high pesticide treatment and runoff, particularly consisting of fungicides and insecticides (Dachbrodt-Saaydeh et al., 2021). More specifically, pesticides cover a range of substances with different modes of action, including insecticides that may directly eradicate macroinvertebrates (Schulz and Liess, 1999), herbicides that may affect the growth of algae (Lorente et al., 2015) and higher plants (Ribeiro et al., 2019) once flushed into rivers and subsequently the feeding conditions of macroinvertebrates, and finally fungicides that may modify leaf litter breakdown in streams and thus the food availability for invertebrates (Artigas et al., 2012).

The second link in this cause-effect chain has also already been investigated in large-scale studies, and there is clear empirical evidence that specific stressors which are potentially caused by agriculture affect river biota and that these effects differ between organism groups. Fine sediments (Davis et al., 2022) mainly affect macroinvertebrates (Davis et al., 2022) and Liess et al. (2021) identified pesticides applied on arable land as the most dominant stressor for vulnerable aquatic insects in lowland streams. In contrast, nutrients are more strongly associated with macrophytes (aquatic plants) and diatoms (O'Hare et al., 2018). River morphology (Kaijser et al., 2022) and river management (Bączyk et al., 2018) are crucial variables for the occurrence of macrophytes, whereas morphological alterations directly impair macroinvertebrates (Urbanič, 2014).

However, typical stressors resulting from agriculture can also originate from other land use types, and contradictory results have been reported about whether agricultural or urban land use exerts higher stress on river biota. Streams draining urban areas are also frequently impacted by both hydromorphological alterations and nutrient pollution (Weitere et al., 2021), and, in some cases, also by pesticide pollution (Gerecke et al., 2002). As additional stressors are common, e.g., micropollutants and flash floods caused by impervious land cover (e.g. Zhou et al., 2022), urban areas are frequently expected to exert higher stress on freshwater biota compared to agricultural land use (e.g. Böhmer et al., 2004). In contrast, recent studies suggest agriculture, in particular cropland, as a main driver of freshwater biota deterioration (Gerecke et al., 2002; Neumann et al., 2002; Stehle and Schulz, 2015). Most of these large-scale empirical studies only distinguished between rather broad land use categories and mainly used gross agricultural land use types, often only distinguishing between crop- and grassland (e.g. Gieswein et al., 2017; Davis et al., 2022). Differential effects of specific crop types on river biota have rarely been considered in large-scale empirical studies yet (Wasson et al., 2010), as high-resolution land use data distinguishing specific crop types became available only recently (e.g. Blickensdörfer et al., 2022).

Against this background, a large dataset of 7748 sampling sites for macroinvertebrates, 2905 sites for macrophytes and 3402 sites for diatoms from rivers across Germany was used to test if cropland exerts higher stress on river biota compared to other land use types (especially urban land use, but also grassland and forests), and to investigate the specific associations between different crop types and individual organism groups (macroinvertebrates, macrophytes, diatoms). We hypothesized that (1) urban land use shows weaker associations with riverine biota compared to cropland and larger associations compared to grassland, while forests have positive associations. (2a) The associations between different crop types and the organism groups strongly differ due to cultivation practices such as crop-specific pesticide and nutrient application rates. (2b) Invertebrates are more sensitive to pesticide-intensive crops, while crops that are intensively fertilized are more strongly affecting macrophytes and diatoms.

2. Methods

2.1. Biological data

Data on biological samples (n = 7748 for macroinvertebrates, n = 2905 for macrophytes, n = 3402 for diatoms) taken between 2010 and 2019 were acquired from all German federal states, except the Saarland. The biological samples were taken with standardized methods used for the ecological status assessment according to the EU Water Framework Directive. Macroinvertebrates were sampled with a multi-habitat sampling method (Haase et al., 2004); we calculated the river-type specific multimetric invertebrate index (MMI) with the resulting species-level taxa lists using the online tool PERLODES (https://www.gewaesser-bewertung-berechnung.de/index.php/perlodes-online.html). The MMI is a combination of biodiversity-related indices and was designed to reflect the impact of various stressors like hydromorphological degradation, altered hydrology, and impacts of land use (Böhmer et al., 2004; Hering et al., 2006). The German Fauna Index is the main metric included in the MMI, which mainly assesses the presence of river-type specific indicator species. Depending on the river-type, usually two to four additional core metrics are included in the MMI like the share of Ephemeroptera, Plecoptera and Trichoptera (%EPT), the share of rheophilic species, or the number of Trichoptera species. Macrophytes and diatoms were sampled following Schaumburg et al. (2012) and the species-level taxa lists were processed using the online tool PHYLIB (https://www.gewaesser-bewertung-berechnung.de/index.php/phylib-online.html) to calculate multimetric indices for macrophytes and diatoms. The multimetric index for macrophytes is mainly based on the quantity of river-type specific indicator species, while the MMI for diatoms is based on both river-type specific indicator species and additional metrics mainly regarding nutrients, salinization, or acidification. We used a standardized version of all three multimetric indices ranging from 0 (worst) to 1 (best) to enable later comparison. Metric selection and thresholds between quality classes differ between the 30 different German river-types, which account for differences in ecoregion, altitude, geology, and river size (Pottgiesser and Sommerhäuser, 2008). If sites have been sampled multiple times, we choose the sampling date closest to 2018 (one year after the land use data were recorded; see next paragraph).

2.2. Catchment land use

For each sampling site, we quantified catchment land use. Upstream catchments (watersheds) were automatically delineated with ESRI ArcView 3.3 based on a digital elevation model (DEM, 10 m resolution) and visually checked. We used agricultural land use data for the year 2017 provided by Blickensdörfer et al. (2022) that were derived through random forest classification of Sentinel-2, Landsat 8, and Sentinel-1 data at a resolution of 10 m. For each catchment, the percentage cover of 23 agricultural crop types distinguished by Blickensdörfer et al. (2022) was quantified with ESRI ArcGIS Pro 2.9.0 and Spyder (Phyton 3.7). We added urban areas and forests for the year 2016 from the land use data of Griffiths et al. (2019) and processed it likewise. The time mismatch of crop maps (2017) and biological data (2010–2019) is likely partly compensated by the sheer size of the dataset (7748 sites). Moreover, results from analysis with land use data from 2018 differed only marginally (except for oilseeds) and would not change the general interpretation (Table S1). Although crops rotate between years for individual fields, crop compositions tend to vary only slightly on larger spatial scales. However, farmers might have responded to extreme weather conditions such as excessive rainfall in 2018 (Blickensdörfer et al., 2022). Consequently, we chose to use 'the year of biological data' as a random factor in our models.

To decrease collinearity between predictors, avoid overfitting, and increase model stability, the 23 different crop types were grouped based on similarities in cultivation practices and assumed similarities in impacts on freshwater biota (Table S2), in particular fertilizer (Britz and Witzke, 2014) with highest application rates for maize and pesticide usage (Andert et al., 2015; Dachbrodt-Saaydeh et al., 2021). While crop-specific pesticide application rates are not available at a Germany-wide scale, Dachbrodt-Saaydeh et al. (2021) reported crop-specific pesticide treatment frequency indices, also referred to as the number of full doses (Sattler et al., 2007; Ferguson and Evans, 2010). Pesticide treatment frequencies for the different crop types ranged from 0.4 to 5.2 for herbicides, 0 and 24.8 for fungicides, and 0 and 5.3 for insecticides in 2017 (Dachbrodt-Saaydeh et al., 2021). For permanent crops and vegetables, the highest pesticide treatment was reported, particularly concerning insecticides and fungicides, while the weakest pesticide applications were found for maize, on which exclusively herbicide application was reported (Dachbrodt-Saaydeh et al., 2021). The resulting land use groups were named "Intensive Cereals", "Extensive Cereals", "Oilseeds", "Permanent crops", "Maize", "Vegetables", "Grassland", "Urban" and "Forest", with mean percentage cover ranging between 1 and 35 % (Fig. S1). For general comparison of agriculture with urban areas, grassland, and forests, we combined all crop land use groups into a gross land use group "cropland".

2.3. GLMM (generalized linear mixed model)

To quantify the associations between the land use groups and the ecological status, we fitted generalized linear mixed models (GLMMs) using the 'gamlss' package in r-studio (v5.2-0; Rigby and Stasinopoulos, 2005). The GLMMs were set up with a zero-one inflated beta regression (BEINF) and a logit link to decrease the dependency of the effect sizes on the gradient length (compare Mack et al., 2022). Prior to model fitting, we assessed collinearity with the variance inflation factor (VIF). Yielding values <2.5, we assumed small collinearity and kept all variables for the analysis. Also, only negligible spatial autocorrelation was observed (Figs. S2-S4), which was consequently not regarded in the models.

We built two models for each organism group. The general model contained the four gross land use groups (cropland, grassland, urban, and forest) as fixed variables. The specific model contained the nine more detailed land use groups (intensive cereals, extensive cereals, oilseeds, permanent crops, maize, vegetables, grassland, urban, and forest) as fixed variables (Table S2). Additionally, four random variables were used in all models, namely "river type" following Pottgiesser and Sommerhäuser (2008) (to account for differences in river size and altitude), "year of biological sampling" (to account for variations in land use between years), "federal state" (to account for possible differences in sampling), and "category" (indicating whether the sampling site was considered natural or 'heavily modified' in accordance with Article 4 of the EU Water Framework Directive). A visual check of the residual distribution against the predicted values and each variable yielded centered averages and symmetrical distributions. Finally, regression coefficients (log odds) including 95 % confidence intervals were calculated using the r-package 'parameters' (v0.12.0; Lüdecke et al., 2020).

3. Results

3.1. Associations between river biota and cropland vs. other land use categories

Our first hypothesis (stronger negative associations between cropland and river biota compared to urban land use) was largely supported by the results of the organism group-specific general models (Table S3). For macroinvertebrates, the difference was most pronounced, with the regression coefficient for cropland (−1.31) being >40 % larger compared to urban land use (−0.92). Moreover, this difference can be considered reliable because the 95 % confidence intervals did not overlap, with −0.73 to −1.12 for urban land use and −1.17 to −1.45 for cropland. For macrophytes, regression coefficients differed only by 10 %: −0.95 [−1.19;−0.71] for cropland and −0.83 [−1.20;−0.45] for urban land use. For diatoms, the magnitude of land use associations were smaller, and although they differed by 40 %, the confidence intervals did overlap with regression coefficients of −0.63 [−0.73;−0.53] and −0.44 [−0.60;−0.28] for cropland and urban land use, respectively. Grassland and forest tended to have small negative or even positive regression coefficients (Table S2).

3.2. Associations between river biota and different crop type groups

Hypothesis 2a (crop types differ in their associations with the ecological status) was partly supported by the specific models (Figs. 1–3). For macroinvertebrates, the regression coefficients of the six crop land use groups differed most strongly, ranging between log odds of −4.48 for permanent crops and 0.49 for extensive cereals, i.e. spanned a range of 4.97. Moreover, 95 % confidence intervals did not overlap, and differences were therefore reliable for the three groups of 'permanent crops' vs. 'vegetables, maize and intensive cereals' vs. 'oilseeds and extensive cereals' (Fig. 1). For macrophytes, the log odds only ranged between −2.83 for permanent crops and −0.48 for extensive cereals (range = 2.35), and confidence intervals strongly overlapped for all crop type groups (Fig. 2). For diatoms, the range was even smaller, with regression coefficients from −0.99 for maize to 1.05 for extensive cereals (range = 2.04). However, confidence intervals of vegetables and extensive cereals did not overlap with 'maize, intensive cereals, and oilseed', indicating that at least these differences were reliable (Fig. 3).

The organism groups clearly differed in their sensitivity to the crop type groups, as predicted by hypothesis 2b (Figs. 1–3). For macroinvertebrates, the pesticide-intensive permanent crops and vegetables showed the strongest negative associations with regression coefficients of −4.48 and −1.82, respectively. The intensively fertilized crops 'maize and intensive cereals' had smaller negative regression coefficients of −1.33 and −1.28, respectively, while oilseeds and extensive cereals had no clear associations. In contrast to the crops, grassland had a reliable positive association with a regression coefficient of 0.65 (Fig. 1). For macrophytes, a similar pattern was observed (Fig. 2). Pesticide-intensive permanent crops and vegetables had the strongest negative association with regression coefficients of −2.83 and −1.71 respectively, while intensive cereals associated with high nutrient application rates also had a relatively large negative regression coefficient of −0.92. However, associations were smaller in magnitude compared to macroinvertebrates, and differences were less pronounced given that confidence intervals overlapped. For diatoms, a different pattern was observed (Fig. 3): Intensively fertilized crops like maize and intensive cereals, as well as oilseeds, showed the largest negative associations with regression coefficients of −0.99, −0.77 and −0.75, respectively. In contrast, permanent crops and vegetables associated with high pesticide application rates showed no reliable relationship with confidence intervals overlapping the zero line. However, associations were smallest compared to macrophytes and macroinvertebrates, and most confidence intervals overlapped.

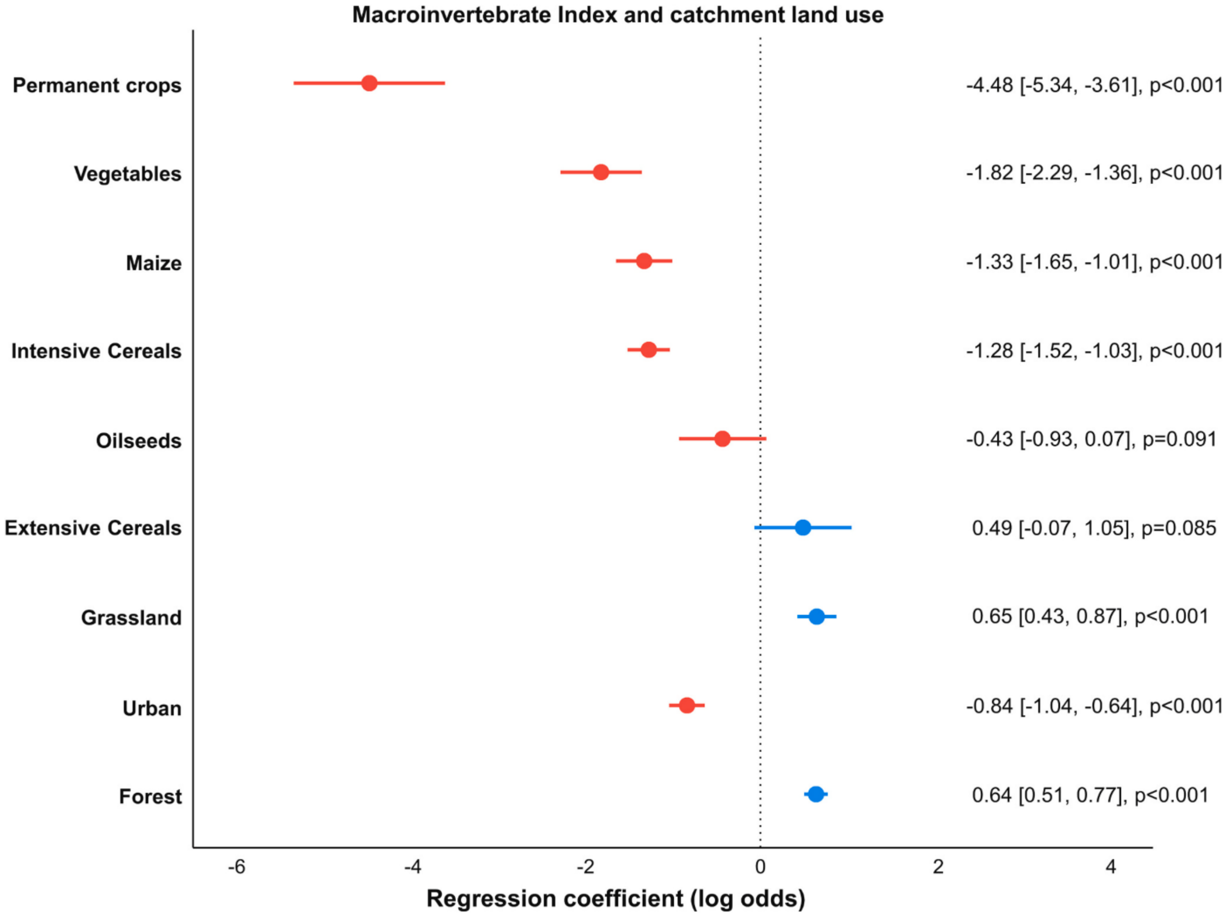

Fig. 1. Zero one inflated GLMM with land use groups as fixed effects and the macroinvertebrate multimetric index as response (n = 7748). Plot shows the regression coefficients of the different catchment land use groups (Intercept = 0.00 [−0.12, 0.13]).

4. Discussion

The land use data recently published by Blickensdörfer et al. (2022) allowed to investigate the associations between specific crop types and different organism groups in much more detail compared to previous large-scale studies, which were restricted to the effect of cropland in general on river biota. The results clearly showed different associations between cropland land use at the catchment scale and the ecological status of macroinvertebrates, macrophytes and diatoms depending on the specific crop types. Given that these crop types typically differ in respect to application rates of fertilizers and pesticides, these results likely mirror the underlying effects of agrochemicals.

4.1. Associations between river biota and cropland vs. other land use categories

Our analysis underlines the differential associations between different types of catchment land use and river biota. The expectation of stronger associations of cropland compared to urban land use (hypothesis 1) was largely supported. This coincides with other recent studies suggesting agriculture to be the dominant stressor for river biota (Wolfram et al., 2021; Liess et al., 2021). Stressors characteristic for arable land (e.g. nutrients resulting from fertilizer use and pesticides) appear more severe on a nationwide scale, than typical stressors resulting from urban land use, such as sewage overflows, flash floods related to impervious land cover, and micropollutants (Weitere et al., 2021) and urban pesticides (Gerecke et al., 2002) and nutrient stress (Müller et al., 2002) appear less relevant (Neumann et al., 2002). Naturally, stressors originating from arable land and urban areas overlap; e.g., parts of the agrochemicals found in wastewater treatment plants adjacent to urban areas originate from urban sources such as roads, railroads, and urban green spaces. However, the substantial pesticide contribution is likely to result from farm courtyard draining to urban point sources (Gerecke et al., 2002; Neumann et al., 2002). Forests, the potentially natural vegetation in almost all parts of Germany, had an overall positive association with river biota, as also observed in other studies (e.g. Goss et al., 2020).

4.2. Associations between river biota and different crop type groups

The crop type groups strongly differed in their associations with river biota (hypothesis 2a), likely caused by varying agricultural practices, in particular nutrient (Britz and Witzke, 2014) and pesticide application rates (Andert et al., 2015; Dachbrodt-Saaydeh et al., 2021). Pesticide application rates differ between permanent crops and extensive crops like maize by a factor of up to 15 (Dachbrodt-Saaydeh et al., 2021), while nutrient applications on maize exceed nutrient extensive crops such as permanent crops up to ten-fold (Britz and Witzke, 2014). These differences suggest stronger effects for pesticide- and nutrient-intensive crops compared to grassland (Blake et al., 2012) or crops such as rye and oat with much lower pesticide and fertilizer applications (Schulz et al., 2013; Mushtaq and Mehfuza, 2014). The relations between different

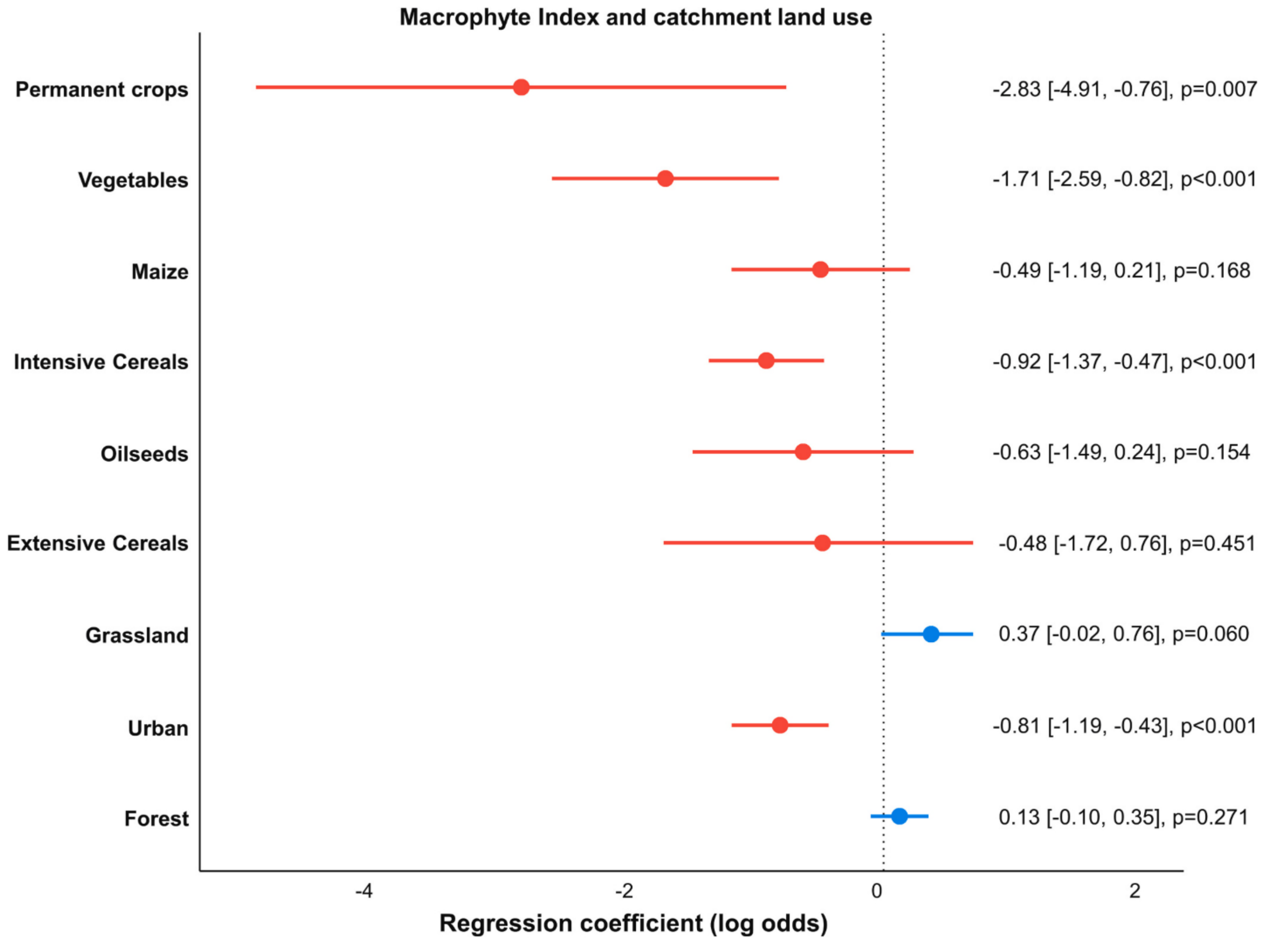

Fig. 2. Zero one inflated GLMM with land use groups as fixed effects and the macrophyte multimetric index as response (n = 2905). Plot shows the regression coefficients of the different catchment land use groups (Intercept = −0.16 [−0.36, 0.05]).

crop types and exerted stressors, suggested by the associations between land use and biota response, are further supported by preliminary analysis relating different crop types with pesticides monitored in rivers (unpublished results) and relating the different crop type groups with the saprobic index, a proxy for eutrophication and the subsequent biomass development (Table S4).

Also, as expected in hypothesis 2b, the organisms differ in their sensitivity to the crop type groups. For macroinvertebrates, the hierarchical order of the association magnitudes between the crop type groups and biota response mirrors the pesticide application rates identified by Dachbrodt-Saaydeh et al. (2021) with strongest associations for permanent crops and vegetables, for which highest insecticide and fungicide application rates were reported. These findings suggesting the importance of pesticides for macroinvertebrates are in line with findings of Liess et al. (2021). Nutrients seem less important, and fine sediments are likely less crop-type-dependent. Although fine sediments impair macroinvertebrates (Gieswein et al., 2019), their dynamics are strongly dependent on hydromorphological factors (Urbanič, 2014) and stressor interactions (Piggott et al., 2012). Moderate nutrient concentrations may even benefit macroinvertebrates (Piggott et al., 2012), while effects turn negative as soon as oxygen depletion resulting from the decomposition of plant biomass is involved (Weitere et al., 2021). Nutrient concentrations might be too low in most German streams to cause these effects. Macrophytes also appear to respond most strongly to pesticide-intensive crops, coinciding with mesocosm studies finding strong effects of agricultural herbicides (e.g. Ribeiro et al., 2019), while O'Hare et al. (2018) contrastingly suggested strong nutrient sensitivity. However, large confidence intervals hinder interpretation, and other factors such as river morphology (Kaijser et al., 2022) or river management (Bączyk et al., 2018) may be more important for macrophyte occurrence. For diatoms, the associations likely mainly relate to nutrient applications, with the strongest associations for the most nutrient-intensive crop type groups, maize and intensive cereals (Britz and Witzke, 2014) and weaker associations for the insecticide- and fungicide-intensive permanent crops and vegetables. Although the overall pesticide application rates are lower for the crop types with strongest diatom response associations (maize, intensive cereals, oilseeds), the share of herbicides applied is higher (Andert et al., 2015; Dachbrodt-Saaydeh et al., 2021). These findings highlight the importance of eutrophication effects for diatoms and match the literature (O'Hare et al., 2018), accompanied by potential effects of herbicides (Debenest et al., 2010). Other than for macroinvertebrates and macrophytes, grassland showed a negative association with diatoms, as most grassland is fertilized in addition to nutrients from livestock (Mouri and Aisaki, 2015). Overall, the crop-type specific differences were most pronounced for macroinvertebrates, while macrophytes and diatoms showed large confidence intervals or smaller regression coefficients, respectively. This coincides with findings from Schürings et al. (2022) suggesting macroinvertebrates are most strongly affected by agriculture. Though we did not investigate causal relationships, our results provide support for the

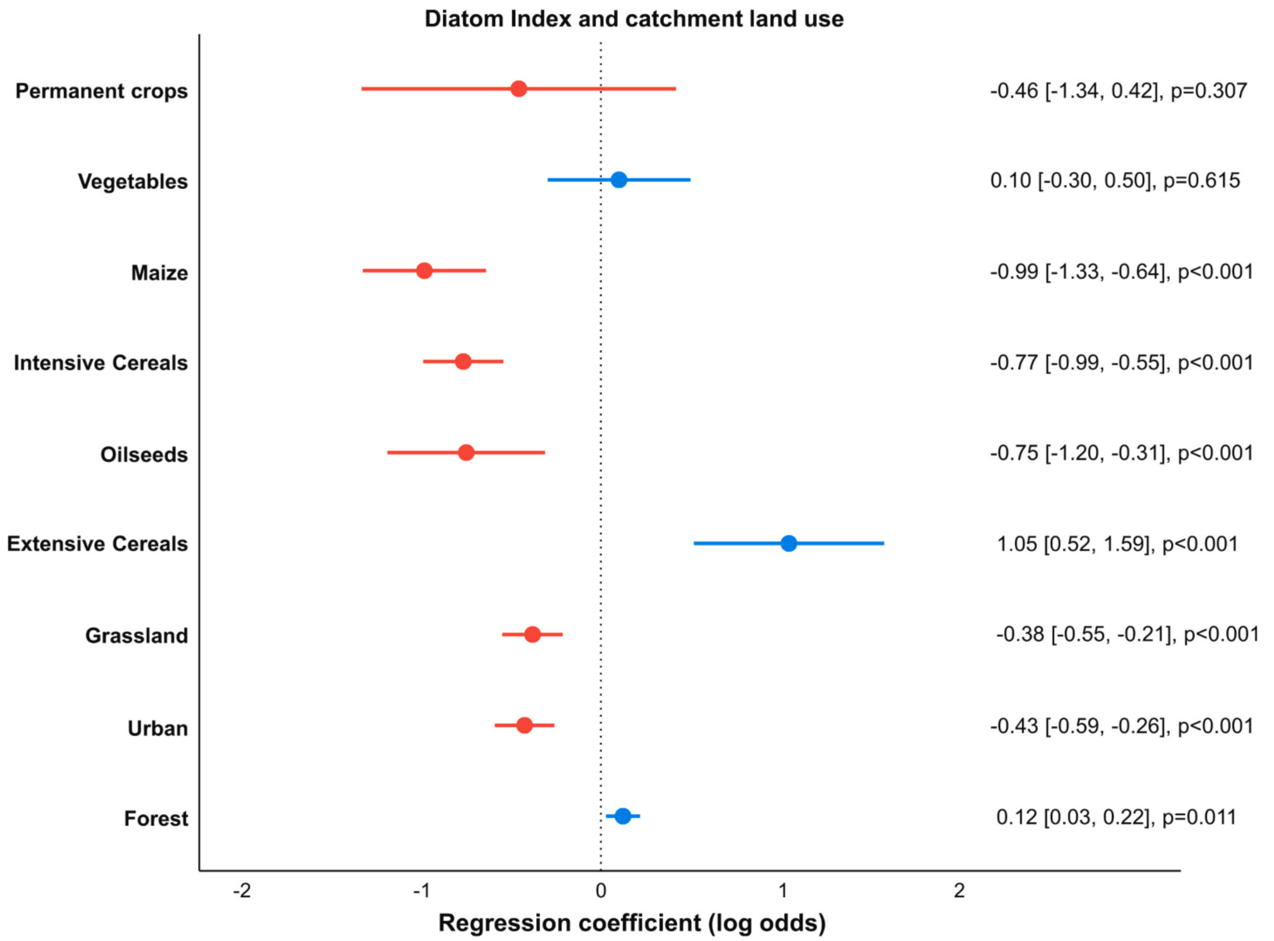

Fig. 3. Zero one inflated GLMM with land use groups as fixed effects and the diatom multimetric index as response (n = 3402). Plot shows the regression coefficients of the different catchment land use groups (Intercept = −1.25 [−1.34, −1.17]).

detrimental effects of pesticide-intensive crops.

To prevent misinterpretation of the positive regression coefficients of extensive cereals and grasslands, which are at a similar magnitude as forests, it is important to consider that all regression coefficients in one model are interdependent. Large shares of extensive cereals and grasslands in a catchment cause lower percentages of more harmful land use types. Moreover, grassland incorporates various types that we could not discriminate: pasture with livestock effects (Mouri and Aisaki, 2015) and (natural) grasslands (Blickensdörfer et al., 2022). Extensive cereals such as rye and oat grow on poor and acid soil and are, besides being main crops, also used as intermediate cover crops, facilitating nutrient retention and reducing soil erosion. They are also known for their allelochemical properties, suppressing weeds, so fewer herbicides and plowing of land are needed (Schulz et al., 2013; Gataneh et al., 2021).

5. Conclusion

The results suggest against using oversimplified approaches to account for agricultural effects such as the sheer cover of arable land and grassland as a proxy for agricultural stress. This study indicates strong differences in agricultural associations on river biota, mainly depending on crop types, which should be accounted for when assessing land use stress. Using multiple organism groups best reflects agricultural impacts. The strong associations of pesticide-intensive crop types call for the implementation of buffer strips to reduce pesticide inputs and, ideally, the implementation of more pesticide-free, environmentally friendly farming practices such as organic agriculture or permaculture to improve the ecological status of rivers.

CRediT authorship contribution statement

Christian Schürings: Conceptualization, Methodology, Writing- Original draft preparation, Writing- Reviewing and Editing.
Jochem Kail: Conceptualization, Methodology, Writing- Rewriting and Editing, Supervision.
Willem Kaijser: Methodology, Writing- Reviewing and Editing.
Daniel Hering: Conceptualization, Methodology, Writing- Reviewing and Editing, Supervision.

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 and code for reproducing numbers and figures shown are available at https://github.com/cs7792b/Effects_of_crop_types

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Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.scitotenv.2023.168825.

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Supplementary Information to Schürings et al.:
Effects of agriculture on river biota differ between crop types and organism groups

Comparison of land use effects on river biota between 2017 and 2018

Table S1: Zero one inflated GLMM with land use groups as fixed effects and the different multimetric indices as response. Shown are the regression coefficients of the different catchment land use groups for the years 2017 and 2018.

	2017	2018
Macroinvertebrates		
Permanent crops	-4.48 [-5.34;-3.61]	-3.99 [-4.84;-3.15]
Vegetables	-1.82 [-2.29;-1.36]	-1.87 [-2.34;-1.41]
Maize	-1.33 [-1.65;-1.01]	-1.65 [-1.97;-1.33]
Intensive Cereals	-1.28 [-1.52;-1.03]	-1.00 [-1.25;-0.75]
Oilseeds	-0.43 [-0.93;0.07]	-1.07 [-1.61;-0.52]
Extensive Cereals	0.49 [-0.07;1.05]	0.29 [-0.27;0.86]
Grasslands	0.65 [0.43;0.87]	0.72 [0.50; 0.94]
Urban	-0.84 [-1.04;-0.64]	-0.86 [-1.06;-0.65]
Forest	0.64 [0.51;0.77]	0.62 [0.48; 0.75]
Macrophytes		
Permanent crops	-2.83 [-4.91;-0.76]	-2.73 [-4.60;-0.86]
Vegetables	-1.71 [-2.59;-0.82]	-1.26 [-2.25;-0.30]
Maize	-0.49 [-1.19;0.21]	-0.44 [-1.12;0.24]
Intensive Cereals	-0.92 [-1.37;-0.47]	-1.35 [-1.82;-0.87]
Oilseeds	-0.63 [-1.49;0.24]	0.50 [-0.49;1.49]
Extensive Cereals	-0.48 [-1.72;0.76]	-0.53 [-1.66;0.60]
Grasslands	0.37 [-0.02;0.76]	0.31 [-0.07;0.70]
Urban	-0.81 [-1.19;-0.43]	-0.79 [-1.17;-0.41]
Forest	0.13 [-0.10; 0.35]	0.15 [-0.07;0.37]
Diatoms		
Permanent crops	-0.46 [-1.34;0.42]	-0.59 [-1.43;0.26]
Vegetables	0.10 [-0.30;0.50]	-0.39 [-0.82;0.05]
Maize	-0.99 [-1.33;-0.64]	-1.01 [-1.35;-0.67]
Intensive Cereals	-0.77 [-0.99;-0.55]	-0.46 [-0.69;-0.23]
Oilseeds	-0.75 [-1.20;-0.31]	-1.16 [-1.64;-0.68]
Extensive Cereals	1.05 [0.52:1.59]	0.65 [0.15;1.14]
Grasslands	-0.38 [-0.55;-0.21]	-0.36 [-0.53;-0.20]
Urban	-0.43 [-0.59;-0.26]	-0.42 [-0.59;-0.25]
Forest	0.12 [0.03;0.22]	0.13 [0.03:0.23]

Allocation of land use types

Table S2: Allocation of land use types to groups. * Data from Griffith et al., 2019

Land use group	**Allocated land use types**
Intensive Cereals	Winter Barley, Spring Barley, Winter Wheat, Other winter Cereals, Other summer Cereals
Extensive Cereals	Oat, Rye
Oilseeds	Winter Rapeseed, Sunflower
Permanent crops	Vineyard, Orchards, Hops
Maize	Forage maize and sweet maize
Vegetables	Sugar beet, Potatoes, Carrots, Asparagus, Onions, Strawberries, Legumes, Other Vegetables

Cropland	Winter Barley, Spring Barley, Winter Wheat, Other winter Cereals, Other summer Cereals, Oat, Rye, Winter Rapeseed, Sunflower, Vineyard, Orchards, Hops, Forage maize and sweet maize, Sugar beet, Potatoes, Carrots, Asparagus, Onions, Strawberries, Legumes, Other Vegetables
Grassland	Grassland
Forest	Small woody features, Forest*
Urban	Built up*

Regression coefficients of simple models

Table S3: Regression coefficients of the simple models for cropland, grasslands, urban land use and forests

Land use type	Regression coefficient	Confidence Interval 95%
Macroinvertebrates		
Intercept	0.08	[-0.04;0.21]
Cropland	-1.31	[-1.45;-1.17]
Grassland	0.75	[0.53;0.97]
Urban	-0.92	[-1.12;-0.73]
Forest	0.56	[0.43;0.70]
Macrophytes		
Intercept	-0.14	[-0.34;0.07]
Cropland	-0.95	[-1.19;-0.71]
Grassland	0.48	[0.11;0.86]
Urban	-0.83	[-1.20;-0.45]
Forest	0.10	[-0.12;0.33]
Diatoms		
Intercept	-1.23	[-1.31;-1.15]
Cropland	-0.63	[-0.73;-0.53]
Grassland	-0.36	[-0.52;-0.20]
Urban	-0.44	[-0.60;-0.28]
Forest	0.13	[0.03;0.22]

Table S4: Pearson correlations between the different land use types and the saprobic index calculated with the PERLODES (https://www.gewaesser-bewertung-berechnung.de/index.php/perlodes-online.html)

Land use types	Pearson ρ
Permanent Crops	0.06
Vegetables	0.25
Maize	0.30
Intensive Cereals	0.35
Oilseeds	0.25
Extensive Cereals	0.23
Grassland	-0.19
Urban	0.09
Forest	-0.32

Supplementary Information to Schürings et al.:
Effects of agriculture on river biota differ between crop types and organism groups

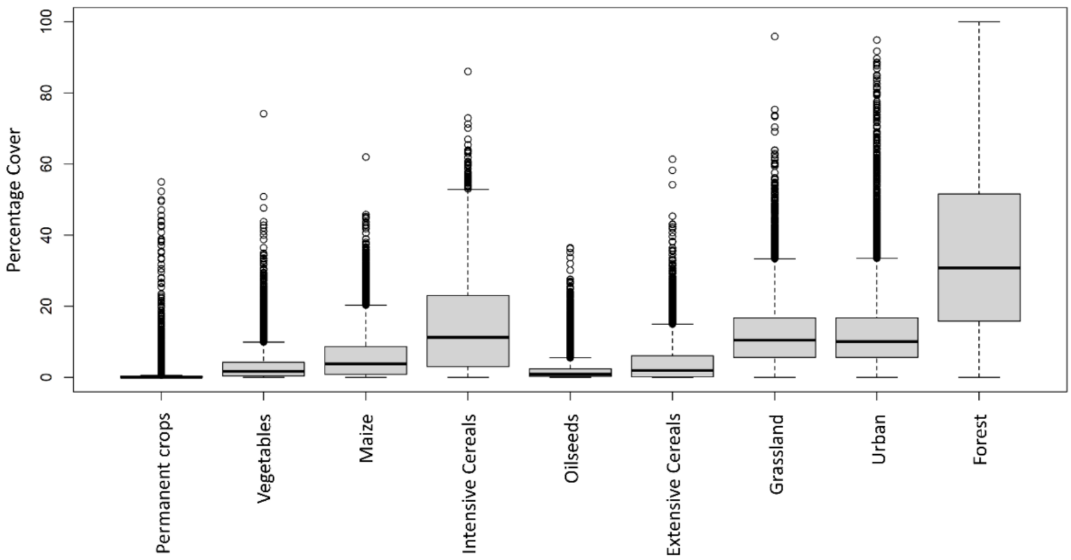

Figure S1: Boxplots with the percentage cover of the different land use types used in this analysis.

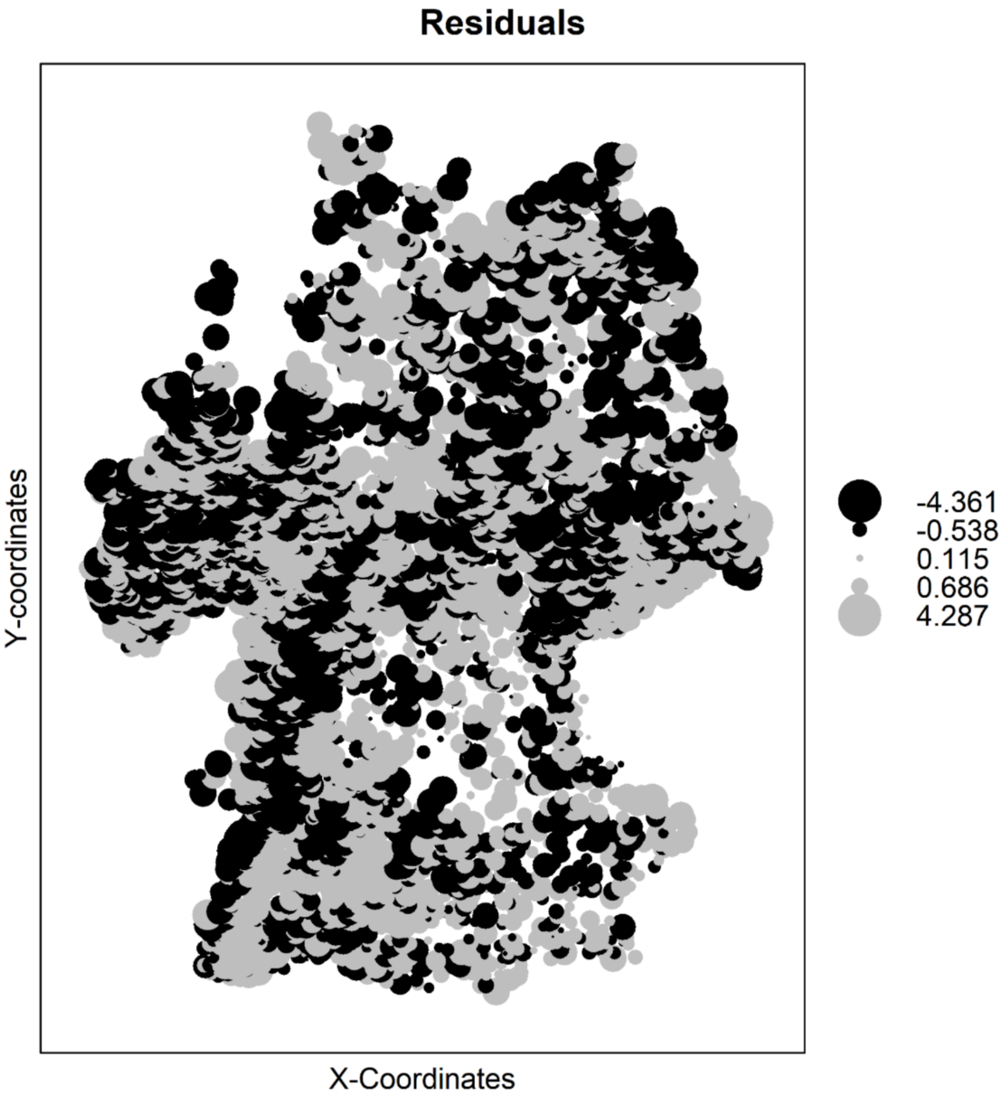

Figure S2: Bubble plot of the residuals for the model with the macroinvertebrates´ multimetric index as response and the different land use types as predictors. Moran I: observed = 0.01, expected = 0.00, Standard deviation = 0.00, p-value = 0.00.

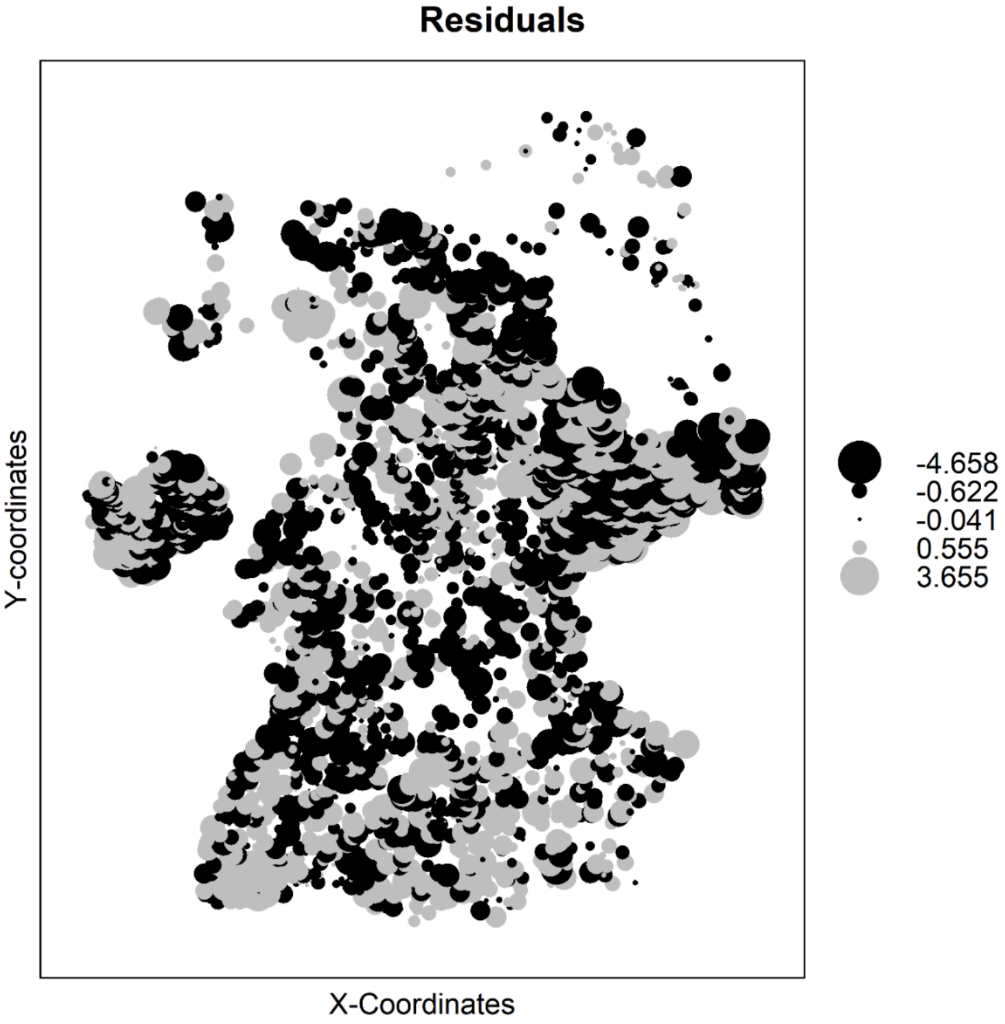

Figure S3: Bubble plot of the residuals for the model with the macrophytes´ multimetric index as response and the different land use types as predictors. Moran I: observed = 0.01, expected = 0.00, Standard deviation = 0.00, p-value = 0.00.

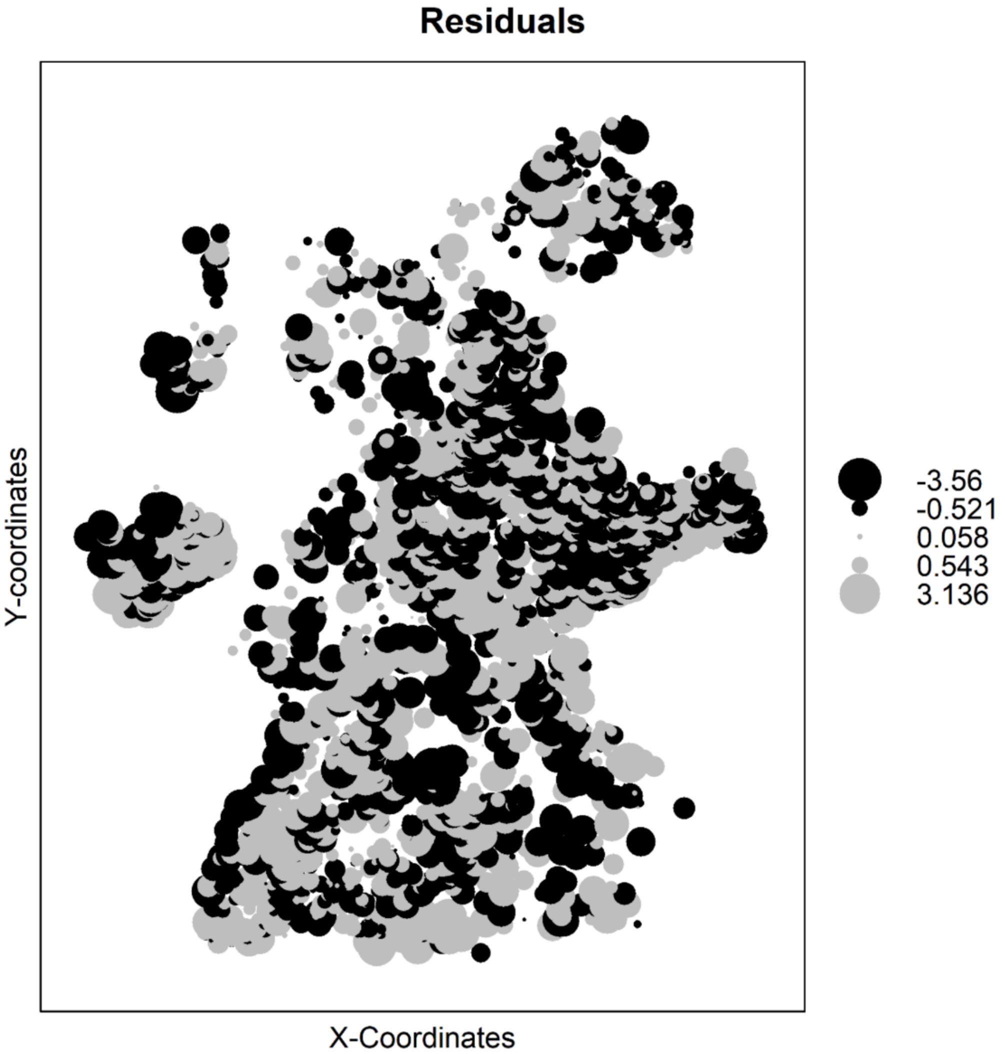

Figure S4: Bubble plot of the residuals for the model with the diatoms´ multimetric index as response and the different land use types as predictors. Moran I: observed = 0.01, expected = 0.00, Standard deviation = 0.00, p-value = 0.00.

Contributions to publications

Cumulative Dissertation of Christian Schürings

Author contributions

Titel: *Effects of agriculture on river biota differ between crop types and organism groups*
Authors: Schürings, C., Kail, J., Kaisjer, W., & Hering, D.

Contributions:

- Conception – 70%
- Conduction of experimental work – not applicable
- Data analysis – 90%
- Species identification – not applicable
- Statistical analysis – 70%
- Writing the manuscript – 100%
- Revision of the manuscript – 60%

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Signature of the Doctoral Candidate Signature of the Doctoral Supervisor

Chapter 4

Water Framework Directive micropollutant monitoring mirrors catchment land use: Agricultural and urban sources revealed

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Water Framework Directive micropollutant monitoring mirrors catchment land use: Agricultural and urban sources revealed

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Keywords: Land use, Agriculture, Urban area, Crop type, Chemical monitoring, Pollution source, Ecological Quality, River basin management

Abstract

River monitoring programs in Germany consistently unveil micropollutant concentrations (pesticide, pharmaceuticals, and industrial chemicals) exceeding regulatory quality targets, which implies deteriorating effects on aquatic communities. However, both the composition and individual concentrations of micropollutants are likely to vary with the catchment land use, in particular regarding urban and agricultural area as the primary sources of micropollutants. In this study, we used a dataset of 109 monitoring sites of micropollutants across the Federal State of North Rhine-Westphalia, Germany, to investigate the relationship between high-resolution catchment land use (distinguishing urban, forested and grassland area as well as 22 different agricultural crop types) and 39 micropollutants using Linear Mixed Models (LMMs). Percent urban area in the catchment was positively related with pharmaceuticals and industrial chemicals (R^2 up to 0.54), whereas percent grassland and forested area generally showed negative relations with micropollutants. Cropland showed weaker relationships with micropollutants (R^2 up to 0.29), which, however, were much higher for individual crop types, for example for vegetables and the herbicide alconifen or for permanent crops and the insecticide thiacloprid (R^2 up to 0.46). The findings suggest crop type-specific pesticide applications, which highlights the need for high-resolution spatial land use to investigate the

magnitude and dynamics of micropollutant exposure and relevant pollution sources, which would remain undetected with highly aggregated land use classifications. Moreover, the findings imply the need for tailored management measures to reduce micropollutant concentrations from different sources and their related ecological effects. Urban point sources could be managed by advanced wastewater treatment. The reduction of diffuse pollution originating from agricultural land uses requires different measures such as, for example, the construction of artificial wetlands and vegetated buffer strips along river courses. Yet, the application of agricultural micropollutants requires further regulation to prevent pesticides from entering the aquatic environment and exceeding regulatory quality targets.

1. Introduction

Globally, societies face three major planetary crises: biodiversity loss, climate change and chemical pollution (UNEP, 2021). The latter is associated with negative effects on biodiversity, ecosystem health (Groh et al., 2022; Sigmund et al., 2022), as well as human health (Fuller et al., 2022) and may impose long-term economic effects for societies (Grandjean & Bellanger, 2017). More than 350,000 chemicals have so far been registered for production and use worldwide (Wang et al., 2020). Hence, a reduction of chemical pollution is essential to remain within the planetary boundaries, which describe the natural limits for human impact to prevent unacceptable environmental change (Rockström et al., 2009; Diamond et al., 2015). Aquatic ecosystems in particular are strongly impaired by a multitude of micropollutants including pharmaceuticals (Fekadu et al., 2019), pesticides (Schäfer et al., 2011b; Liess et al., 2021) and industrial chemicals (Koumaki et al., 2018), which have previously been associated with ecological degradation (Posthuma et al., 2020; Lemm et al., 2021; Schürings et al., 2024b).

International policies (e.g., European Green Deal) and environmental legislation (UNEP, 2017; European Commission, 2019a; European Commission, 2020a) have been developed to promote the sustainable use of chemical substances and achieve a toxic-free environment. Comprehensive programs to monitor chemical pollution already exist (e.g., EU Water Framework Directive (WFD), Directive 2000/60/EC), which, however, cannot adequately address the numerous substances that are applied (Malaj et al., 2014). The risk assessment of micropollutants is typically based on the comparison of its environmental concentrations with substance-specific ecotoxicological assessment values. For several micropollutants (i.e., priority substances and river basin-specific pollutants) environmental quality standards (EQS) and further ecotoxicologically derived assessment values are set by the WFD and related national legislations (e.g., the German surface waters directive, OGewV, 2016).

The sources of micropollutants and the pathway of pollution vary between substances, while two major pathways of pollution can be distinguished. Point sources constitute spatially explicit points of pollution, for example, effluents of industrial or municipal wastewater treatment plants (WWTP) in urban areas (Loos et al., 2013; Finckh et al., 2022). Contrastingly, diffuse sources of pollution cannot be attributed to explicit effluents, but comprise rather broad-scale pathways such as surface and groundwater run-off from agricultural areas into the aquatic environment (Harrison et al., 2019; Wiering et al., 2020). Agricultural practices and pesticide applications vary between crop types (Andert et al., 2015) and relate to particularly high pesticide application rates for permanent crops and vegetables (Dachbrodt-Saaydeh et al., 2021), which in turn translates to enhanced and ecotoxicologically relevant concentrations for riverine biota (Schulz, 2001; Bereswill et al., 2012; Xing et al., 2012). In contrast, forage maize cultivations are often highly fertilized (Britz & Witzke, 2014), but associated with rather small amounts of pesticides, exclusively herbicides, whereas the use of pesticides on grassland is very limited (Dachbrodt-Saaydeh et al., 2021; Riedo et al., 2022). Forested areas in general show low relationships to micropollutant concentrations and often relate positively to river health (Goss et al., 2020).

In this study we investigated the relationships between catchment land use and individual micropollutants in German rivers. More specifically, we aimed to test whether specific crop types reveal crop type-specific pesticide applications that are reported by Andert et al. (2015) and Dachbrodt-Saaydeh et al. (2021). This differentiation between the sources of pollution as well as the source-specific pollutants is deemed of primary importance for water management because the management of diffuse and point sources would require different management strategies. The following research questions were formulated to guide the analytical process:

(1) Which micropollutants do exceed the environmental quality targets that are set by available environmental regulations and ecotoxicological risks assessments?

(2) Do the monitored micropollutant concentrations reflect the percent urban, forested and agricultural area in the catchment of monitoring sites?

(3) Do agricultural pesticide concentrations relate to specific crop types, thus reflecting crop-specific pesticide application rates?

2. Materials and methods

2.1 Study area

In total, 109 micropollutant monitoring sites were included in this study (Figure 1). The sites are located in the Federal State of North Rhine-Westphalia (NRW), Germany and cover lowland (altitude below 200m) and mountainous regions (altitude 200-800m) as well as small (catchment area 10-100km²), mid-sized (catchment area 100-1000km²) and large rivers. Catchment area ranged 5-2834km² (median: 326km^2; see Supplementary Material Table A1 for detailed site characterisation).

2.2 Micropollutant monitoring and ecotoxicological risk assessment

Data on micropollutant concentrations originate from WFD-related chemical monitoring programs of the North Rhine-Westphalian Office of Nature, Environment and Consumer Protection and regional water boards. Sampling was based upon grab samples of surface water (see OGewV (2016) and LAWA (2019) for details on sampling and analysis) and occurred between 2016 and 2019. For each site one sampling year was selected that temporally matched the reference timing of land use data (2016/2017) best (section 2.3). In total, 39 micropollutants (19 pesticides, 14 pharmaceuticals and six industrial chemicals including personal care products and household chemicals; Table 1) were selected for this study because of their ecotoxicological relevance, i.e. they constitute priority substances, river basin-specific pollutants or candidate substances on the watch list listed by the WFD and were identified as ecotoxicologically relevant by previous studies (e.g., Ginebreda et al., 2014; Gustavsson et al., 2017; Markert et al., 2020). In order to quantify the ecotoxicological risk of micropollutants (research question 1), we calculated individual risk quotients (RQ) for each substance, i.e., the quotient of the measured concentration divided by the substance-specific assessment value (Backhaus & Faust, 2012). The estimation of chronic risks during longer exposure periods was based on annual mean concentrations of individual micropollutants (OGewV, 2016). Assessment values were derived from environmental quality standards (EQS) from the WFD, national legislation (OGewV, 2016) and validated ecotoxicological data (e.g., EQS proposals and predicted no effect concentrations) in accordance with the technical guidance for deriving environmental quality standards (European Commission, 2017; Markert et al., 2020). To account for combined risks of micropollutant mixtures the sum of individual RQs (SUM RQ) was calculated for each site (Backhaus & Faust, 2012; Markert et al., 2020). RQ and SUM RQ values above one indicate that individual and combined micropollutant concentrations exceed the ecotoxicological effect levels and thus constitute a potential (mixture) risk.

Since both the number of micropollutants and the composition of substances measured at each site varied among the sites, data gaps occurred for individual micropollutants that ranged between 1 and 32% of the sites (mean across all substances: 12%). Missing values were imputed using an iterative imputation algorithm based on random forests (missForest), which has previously been shown to perform well for data gaps extending up to 30% (or even 50%) of the

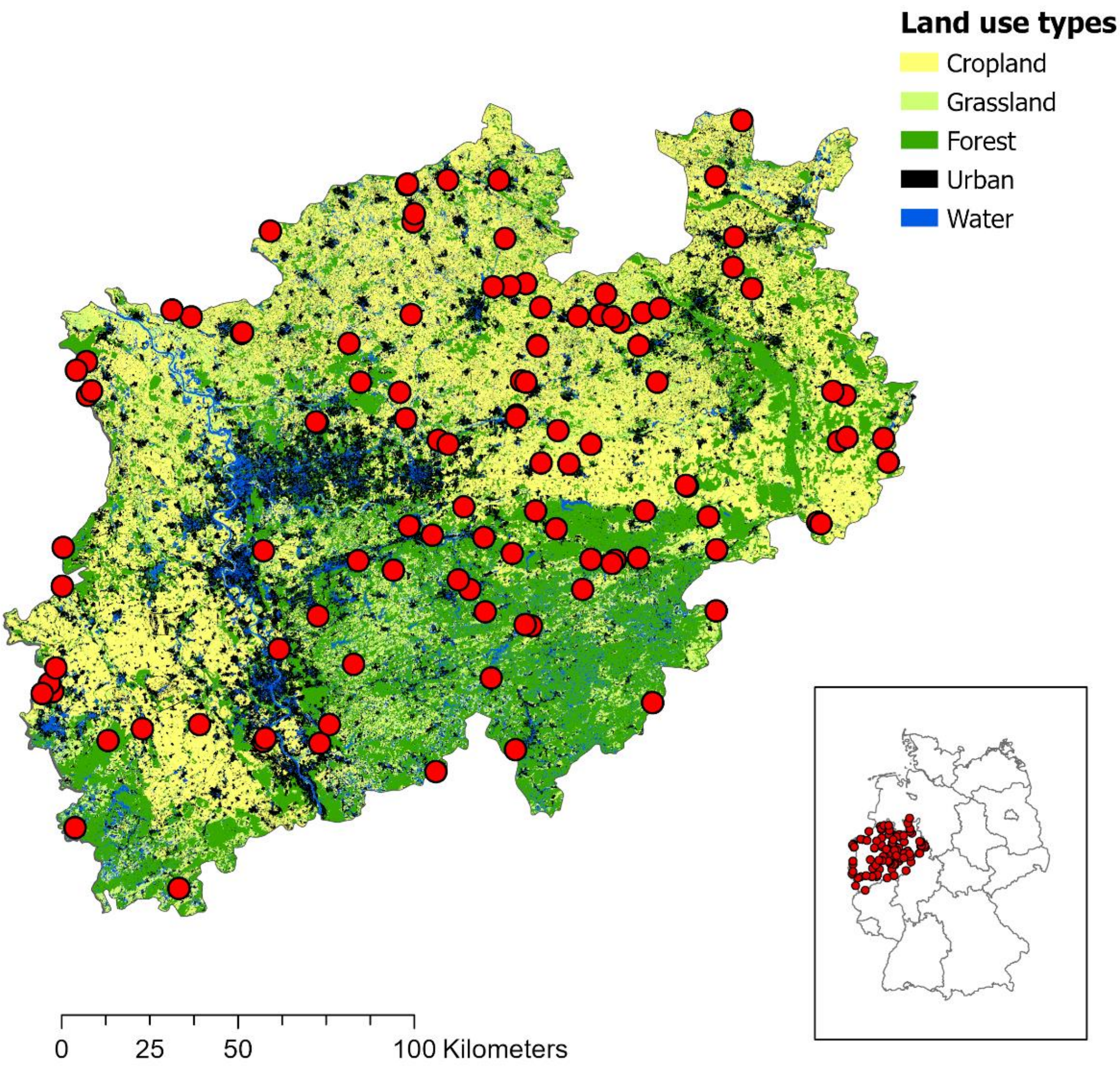


Figure 1: Location of micropollutant monitoring sites in the Federal State of North Rhine-Westphalia (NRW), Germany.

values (Stekhoven & Bühlmann, 2012; Tang & Ishwaran, 2017). Left-censored data, i.e., concentrations below the limit of quantification (LOQ) were replaced by half of the LOQ value for pharmaceuticals and industrial chemicals and by zero for pesticides. This approach was chosen since pharmaceuticals and industrial chemicals are ubiquitously and continuously released into the aquatic environment (Hernando et al., 2006; McEneff et al., 2014), where substitution with zero might lead to a critical underestimation of concentrations. In contrast, pesticides tend to show seasonal concentrations patterns (Vormeier et al., 2023), where substitution with half the LOQ might result in arbitrarily high concentration ranges.

Table 1: Statistical parameters and calculated risk quotients of micropollutants. Scope of application of pesticides ('plant protection products') refers to substance-related approvals in Germany (BVL, 2023a). Three pesticide sub-groups were distinguished: herbicides (H), fungicides (F) and insecticides (I). Risk quotients (RQ) were calculated as quotients of measured concentrations and assessment values in accordance with the technical guidance for deriving environmental quality standards (European Commission, 2017).

			Concentration (µg/L)				Risk quotient (RQ)		
Micropollutant group	Substance	Application (only pesticides)	Min.	Max.	Mean	SD	Min.	Max.	%Sites with RQ > 1
Industrial chemicals	Benzo(a)pyrene	-	0.000	0.010	0.002	0.001	0.29	163.43	100%
Industrial chemicals	Benzotriazole	-	0.025	5.267	1.156	0.991	0.00	0.51	0%
Industrial chemicals	Bisphenol A	-	0.005	0.135	0.040	0.029	0.00	0.17	0%
Industrial chemicals	Fluoranthene	-	0.001	0.028	0.007	0.004	0.08	11.23	44%
Industrial chemicals	Galaxolide (HHCB)	-	0.032	0.243	0.104	0.051	0.00	0.09	0%
Industrial chemicals	Triclosan[2]	-	0.001	0.014	0.006	0.002	0.05	0.88	0%
Pharmaceuticals	Azithromycin	-	0.005	0.079	0.027	0.015	0.26	8.57	60%
Pharmaceuticals	Bezafibrate	-	0.005	0.610	0.040	0.091	0.00	0.43	0%
Pharmaceuticals	Carbamazepine	-	0.005	0.710	0.119	0.124	0.01	1.91	2%
Pharmaceuticals	Ciprofloxacin	-	0.005	0.018	0.013	0.002	0.06	0.39	0%
Pharmaceuticals	Clarithromycin	-	0.006	0.310	0.039	0.052	0.02	3.10	7%
Pharmaceuticals	Clindamycin	-	0.006	0.054	0.021	0.010	0.03	3.32	3%
Pharmaceuticals	Clofibrinicacid	-	0.001	0.062	0.012	0.006	0.00	0.01	0%
Pharmaceuticals	Diclofenac	-	0.005	3.900	0.352	0.586	0.05	78.00	74%

Pharmaceuticals	Erythromycin	-	0.003	0.200	0.020	0.023	0.01	1.00	0%
Pharmaceuticals	Ibuprofen	-	0.004	0.250	0.028	0.039	0.25	46.63	98%
Pharmaceuticals	Naproxen	-	0.002	0.840	0.050	0.108	0.00	0.49	0%
Pharmaceuticals	Paracetamol	-	0.005	0.107	0.025	0.021	0.00	0.01	0%
Pharmaceuticals	Sulfamethoxazol	-	0.005	0.600	0.071	0.086	0.00	1.06	0%
Pharmaceuticals	Venlafaxine	-	0.005	1.000	0.085	0.126	0.00	1.14	1%
Pesticides (H)	Aclonifen	Field crop, Vegetable	0.000	0.006	0.001	0.001	0.00	0.22	0%
Pesticides (F)	Azoxystrobin	Field crop, Vegetable, Fruit, Wine, Biocide, Hop, Ornamental plant	0.000	0.781	0.016	0.094	0.00	3.91	3%
Pesticides (H)	Chlortoluron	Field crop	0.000	0.042	0.003	0.007	0.00	0.29	0%
Pesticides (I)	Clothianidin	Field crop, Vegetable, Ornamental plant, Biocide	0.000	0.016	0.000	0.002	0.00	0.20	0%
Pesticides (H)	2,4-D	Field crop, Fruit, Ornamental plant	0.000	0.031	0.001	0.003	0.00	0.18	0%
Pesticides (H)	Dimethenamid	Field crop, Vegetable, Fruit, Ornamental plant	0.000	0.617	0.010	0.063	0.00	2.37	1%
Pesticides (H)	Diuron[1]	Field crop, Fruit, Wine, Biocide	0.000	0.173	0.006	0.023	0.00	0.87	0%
Pesticides (H)	Ethofumesat	Field crop, Vegetable	0.000	0.035	0.001	0.004	0.00	0.03	0%
Pesticides (H)	Flufenacet	Crop, Vegetable, Fruit, Ornamental plant	0.000	0.097	0.008	0.014	0.00	2.42	3%
Pesticides (I)	Imidacloprid[1]	Field crop, Vegetable, Fruit, Wine, Biocide, Ornamental plant, Hop	0.000	0.159	0.006	0.017	0.00	79.50	43%

Pesticides (H)	Isoproturon[1]	Field crop, Ornamental plant, Biocide	0.000	0.053	0.003	0.009	0.00	0.28	0%
Pesticides (H)	MCPA	Field crop, Hop, Fruit, Ornamental plant	0.000	0.163	0.010	0.022	0.00	0.65	0%
Pesticides (H)	Metazachlor	Field crop, Vegetable, Ornamental plant	0.000	0.195	0.004	0.024	0.00	0.76	0%
Pesticides (H)	Metolachlor	Field crop, Vegetable	0.000	0.117	0.004	0.015	0.00	6.82	0%
Pesticides (H)	Nicosulfuron	Field crop	0.000	0.020	0.002	0.003	0.00	25.56	4%
Pesticides (F)	Tebuconazole	Field crop, Biocide	0.000	0.161	0.004	0.017	0.00	1.11	0%
Pesticides (H)	Terbuthylazine	Field crop, Vegetable	0.000	0.243	0.012	0.034	0.00	1.55	0%
Pesticides (H)	Terbutryn[1]	Biocide	0.000	0.103	0.008	0.016	0.00	3.23	3%
Pesticides (I)	Thiacloprid[1]	Field crop, Vegetable, Fruit, Ornamental plant	0.000	0.014	0.001	0.002	0.00	4.74	2%

[1]The substances diuron, imidacloprid, isoproturon, thiacloprid and terbutryn have been banned for (outdoor) use as ‘plant protection product’ in the EU since 2002 (terbutryn), 2007 (diuron), 2016 (isoproturon), 2018 (imidacloprid) and 2020 (thiacloprid) (BVL, 2023b), but are still approved as biocidal active substances for preservatives, for example, in facade paint or construction material (diuron, terbutryn, isoproturon; ECHA, 2023), for insecticide (imidacloprid, ECHA, 2023), or as veterinary medicinal products (imidacloprid, EMA, 2021).

[2]Triclosan was used as a biocidal active substance for human hygiene, disinfection and preservation but the approval was withdrawn in the EU in 2016 (ECHA, 2023); yet it is used in cosmetic and personal care products (European Commission, 2014).

2.3 Catchment land use

For each sampling site, we quantified the percentage of forested, urban and agricultural terrestrial land use in the catchment area upstream of the site. Catchment delineation was based on a digital elevation model (©dl-zero-de/2.0, Geobasis NRW, 10m resolution) in ArcView 3.3

subsequently checked visually for correctness and clipped with altogether 23 different crop types (including grassland) using ArcGIS Pro 2.9.0 and Spyder (Phyton 3.7). Crop type-specific land uses for 2017 were derived from satellite images (Sentinel-2, Landsat 8 and Sentinel-1, 10 m resolution) (Blickensdörfer et al., 2022). Percent urban and forested area in the catchment for 2016 were derived from Griffiths et al. (2019) and quantified alike crop type-specific land use. To statistically account for the temporal variation of micropollutant data (2016–2019) and land use/cover data (2016–2017), the year of micropollutant sampling was included as a random factor in the models (see below). However, the influence is likely minor, as Schürings et al. (2024a) found no major differences between year when comparing the effect of land use on river biota using the land use data of Blickensdörfer et al. (2022) of the years 2017 and 2018

To quantify and compare catchment land uses, the 23 different crop types (including grassland) and urban and forested area were assigned several categories (Table 2). Except for grassland, all crop types were merged into a category 'cropland' to account for general effects of intensive agricultural land use. In the category, maize and cereals were dominant. Grassland was kept separate because it constitutes a rather extensive form of agricultural land use. To analyse crop type-related effects, the 22 individual crop types were categorized into maize (including silage maize and grain maize), cereals (including wheat, rye, barley, oat and other cereals), oilseeds (including rapeseed and sunflowers), permanent crops (including vineyards, hops and orchards) and vegetables (including potatoes, sugar beets, legumes, strawberries, asparagus, onions, carrots and other vegetables). To further differentiate between different vegetables that are known to be associated with high pesticide application rates (Dachbrodt-Saaydeh et al., 2021) asparagus, strawberries and onions were additionally kept as individual categories (Table 2).

Table 2: Statistical parameters of percent area of land uses in the catchments upstream of the sampling sites and categorization of crop types into sub-groups.

Land use	Min. %	Max. %	Mean %	SD %
Forest	0.04	74.94	31.78	20.07
Urban area	4.05	59.09	20.04	11.51
Grassland	2.89	34.86	12.57	6.54
Cropland	0.11	76.77	29.87	21.66
Maize	0.00	76.91	29.13	17.61
Cereals	5.93	87.39	46.75	15.80
Oilseeds	0.00	25.38	5.56	5.68
Permanent crops	0.00	27.52	2.66	4.83

Vegetables	1.51	48.96	13.72	10.46
Asparagus	0.00	7.20	1.72	1.65
Strawberries	0.00	24.06	2.72	4.23
Onions	0.00	4.53	0.52	0.95

2.4 Statistical analyses

To investigate, whether micropollutant mixture risks are reflected by catchment land uses (research question 2), individual linear regression models of the SUM RQ of industrial chemicals, pharmaceuticals and pesticides (incl. the sub-groups herbicides, fungicides and insecticides) with catchment land uses (cropland, urban area, forest) as predictors were visualized (R package ggplot2 with lm smooth function R Core Team, 2020).

For micropollutant-specific analyses, separate linear mixed models (LMMs) were fitted for each possible combination of four land use categories (urban, forest, grassland, cropland as well as individual crop types) and 39 micropollutants, with the micropollutant concentration as response and the percent area of one land use type as the predictor (i.e. the fixed effect in the model). Ecoregion (lowlands, low mountains) and the year of micropollutant sampling were included in each LMM as random effects. A gaussian distribution was selected for LMMs, as preliminary analyses of the data using Generalized Additive Models (GAMs) suggested that a linear relationship of fixed effects can be assumed. LMMs were run in R with the 'gamlss' package (v5.2-0, Rigby & Stasinopoulos, 2005).

In each of the models, 70 percent of the data were bootstrapped 1,000 times to calculate a mean-peudo-R^2 (from here on referred to as R^2) for the fixed effect including confidence intervals. The R^2 of the fixed effect was calculated as the squared correlation between the fitted response and the predicted response, solely based on the fixed effect. Alongside the individual R^2 (and confidence intervals), an overall R^2 was calculated for each group of micropollutants (pesticide, pharmaceuticals and industrial chemicals), using a random effect model with the metafor package (Viechtbauer, 2010). Effect size (Figure 2) was based on the individual and grouped R^2, however, a sign was added to the plot axis to distinguish positive and negative regression coefficients, i.e. positive or negative effects of land uses on micropollutant concentrations. To analyze whether pesticide concentrations reflect crop-specific pesticide application rates (research question 3), additional models were calculated for the individual crop types, i.e., maize, cereals, oilseeds, permanent crops, vegetables as well as asparagus, strawberries and onions, following the same procedure.

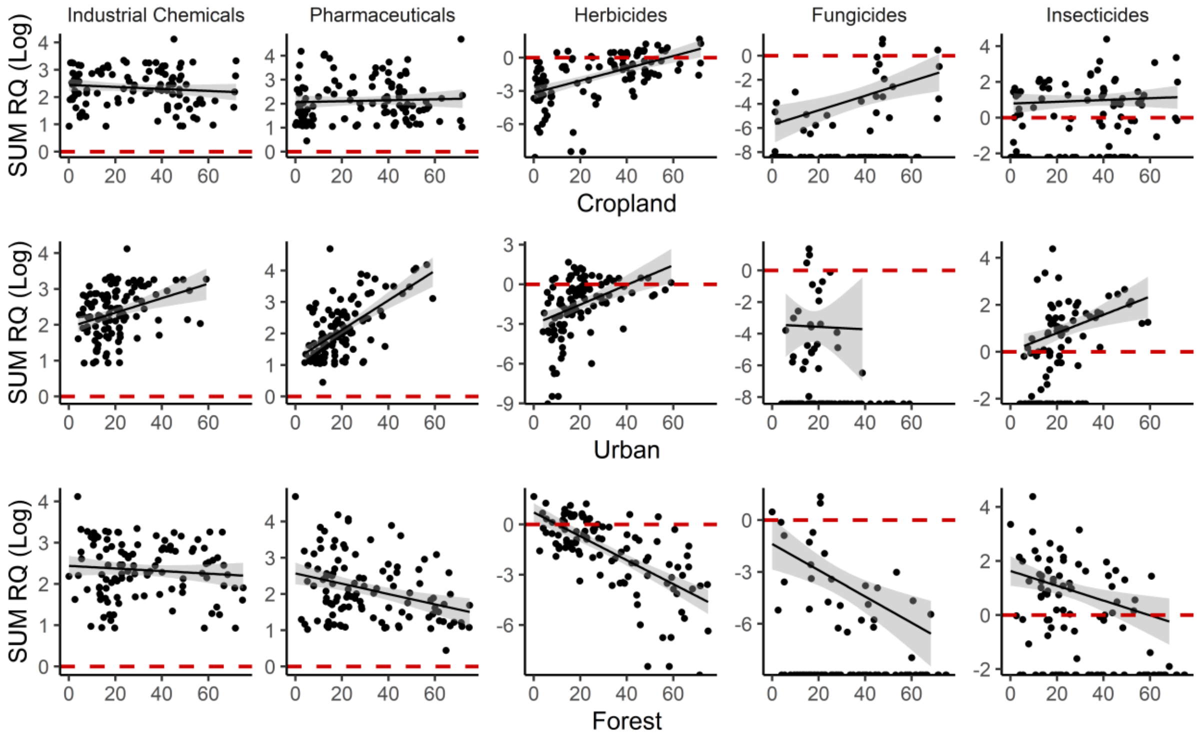
3. Results

3.1 Ecotoxicological risk assessment

Six out of the 39 micropollutants frequently (i.e. at more than 10% of the sampling sites) exceeded regulatory assessment values (RQ > 1) and hence imposed individual ecotoxicological risks: benzo(a)pyrene, ibuprofen, diclofenac, azithromycin, fluoranthene and imidacloprid (Table 1). Furthermore, the pharmaceuticals clarithromycin, clindamycin, carbamazepine and venlafaxine as well as the pesticides thiacloprid, azoxystrobin, nicosulfuron, flufenacet, dimethenamid and terbutryn were found in concentrations exceeding the assessment values, although at less than 10% of the sites. In contrast to individual RQs, mixture toxicity indicated ecotoxicological risks of pharmaceuticals and industrial chemicals at 100% of the sites (SUM RQ > 1, Table A2 Supplementary Material). Mixture toxicity risks by pesticides were evident at 55% of the sites, with insecticides (44% of the sites) dominating the risk assessment over herbicides (27%) and fungicides (3%).

The relationship between mixture toxicity risks and catchment land use varied among micropollutant groups (Figure 2) but showed strong positive relationships of cropland with herbicides ($R^2 = 0.31$) and fungicides ($R^2 = 0.30$) and of urban area with pharmaceuticals ($R^2 = 0.38$). Percent forested area was negatively related with mixture toxicity risks of all micropollutant groups, which was most pronounced for herbicides ($R^2 = 0.45$) and fungicides ($R^2 = 0.33$).

Figure 2: Relationship between percent cropland, urban and forested area in the catchment and mixture risk quotients ($\log_{10}$ SUM RQ) of industrial chemicals, pharmaceuticals, herbicides, fungicides and insecticides. The solid line marks the fit of a linear regression model with 95% confidence interval indicated in gray; dashed red lines mark the threshold of SUM RQ = 1, which translates to 0 along the log-transformed y-axis.



3.2 Link between micropollutant concentrations and land use

Percent urban and forested area, cropland and grassland revealed clear differences in their relationship to individual micropollutants and micropollutant groups (Figures 3 and 4). Urban land use (Figure 3a) was positively related to numerous pollutants, particularly to pharmaceuticals ($R^2 = 0.31$) and industrial chemicals ($R^2 = 0.39$), while its relationship with pesticides ($R^2 = 0.02$) was almost neglectable. Among the pharmaceuticals, antibiotics (azithromycin: $R^2 = 0.54$, clindamycin. $R^2 = 0.45$ and clarithromycin: $R^2 = 0.44$) revealed the strongest relationship to percent urban area. The effect sizes for industrial chemicals were in a similar range and showed particular strong relationships to galaxolide ($R^2 = 0.51$) and triclosan ($R^2 = 0.48$). The strongest individual relationship of a pesticide to urban area was found for terbutryn ($R^2 = 0.27$).

Cropland showed a weak, but positive relationship to pesticides (pooled $R^2 = 0.08$), while its relationship to pharmaceuticals and industrial chemicals was similarly weak, but negative (both $R^2 = 0.02$). The strongest individual relationship between percent cropland and pesticides were found for flufenacet (R^2 = 0.29) and nicosulfuron (R^2 = 0.21), individual relationships to pharmaceuticals and industrial chemicals were negligible.

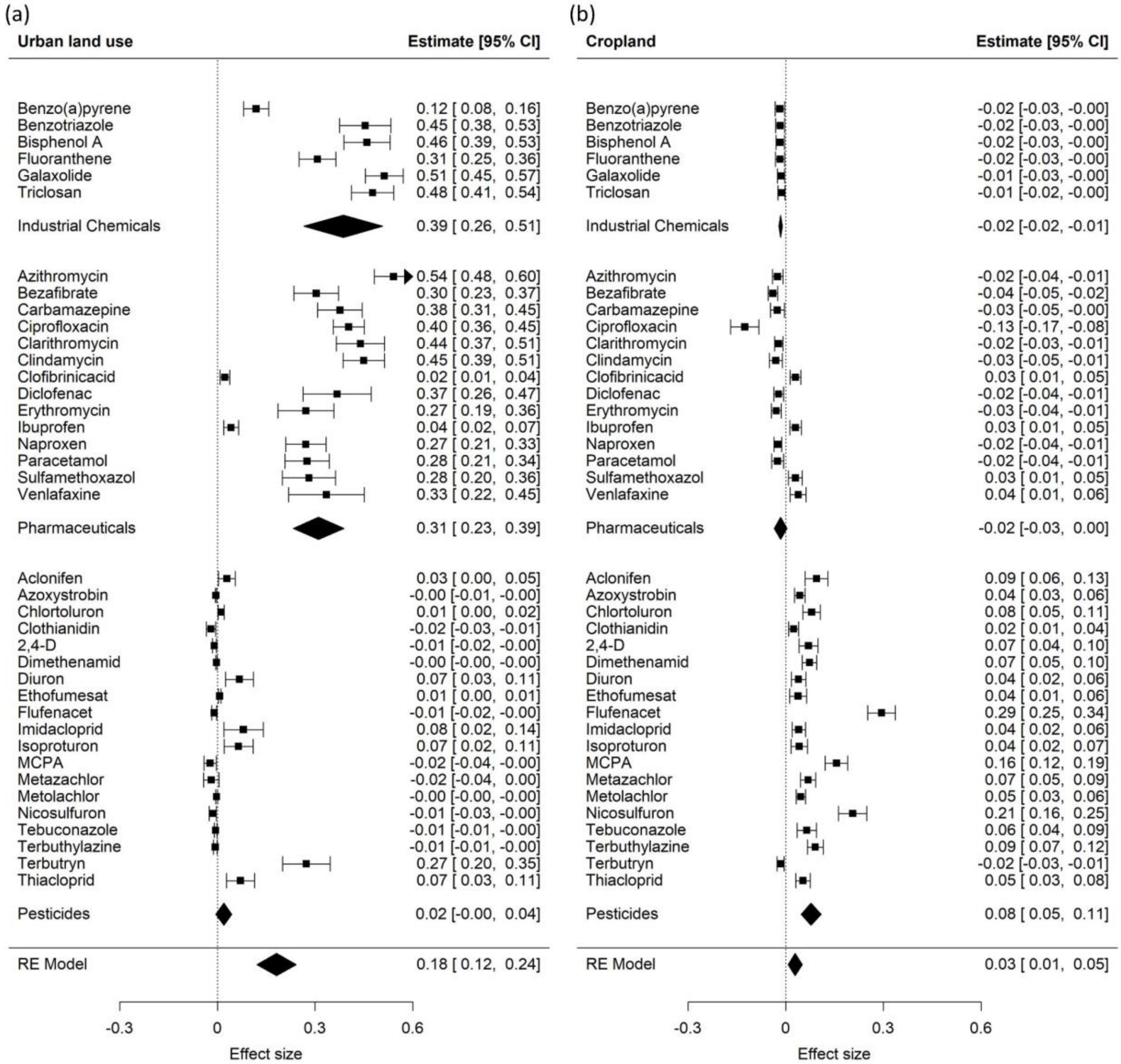


Figure 3: Relationship (effect size) of percent urban area (a) and cropland (b) in the catchment with micropollutant concentrations. Effect sizes represent model fits (pseudo-R^2) derived from bootstrapped (n = 1,000) univariate linear mixed models (LMM) with 95% confidence intervals indicated in brackets. Negative signs were added to account for negative relationships (i.e., negative regression coefficients) – although R^2 values are positive by definition.

Grassland (Figure 4a) showed weak and negative relationships to all micropollutant groups, with pooled effect sizes of R^2 = 0.04, R^2 = 0.07 and R^2 = 0.06 for pesticides, pharmaceuticals and industrial chemicals, respectively. Individual effects of percent grassland were most pronounced and negative for the pharmaceutical ciprofloxacin (R^2 = 0.17) and for the pesticide flufenacet (R^2 = 0.13). Eventually, forest (Figure 4b) showed weak and negative relationships to all micropollutant groups with pooled effect sizes of R^2 = 0.07, R^2 = 0.08 and R^2 = 0.08 for pesticides, pharmaceuticals and industrial chemicals, respectively. Again, the strongest individual relationships to percent forested area were found for ciprofloxacin ($R^2 = 0.33$) and flufenacet ($R^2 = 0.22$).

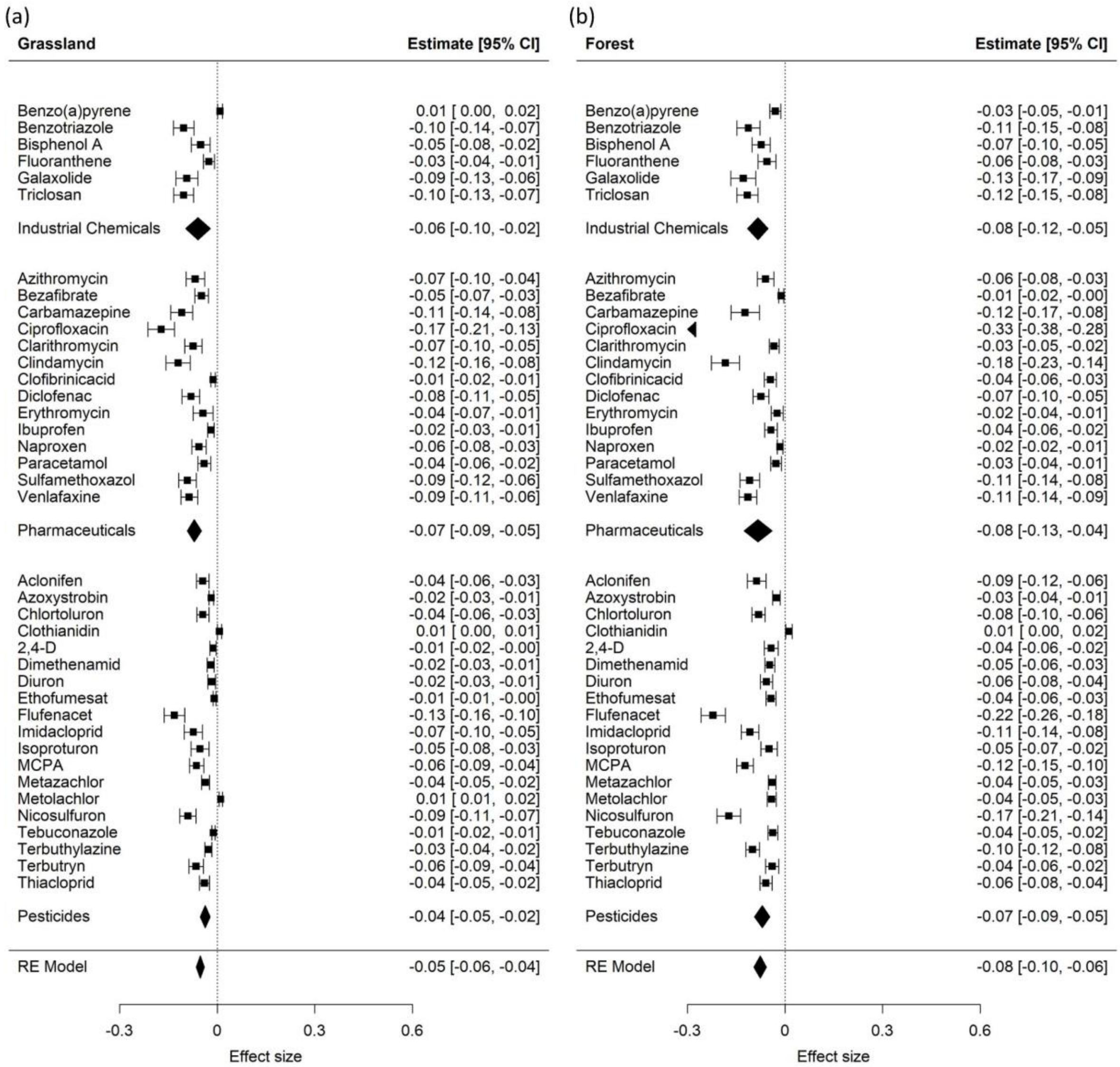


Figure 4: Relationship (effect size) of percent grassland (a) and forested area (b) in the catchment with micropollutant concentrations. Effect sizes represent model fits (pseudo-R^2) derived from bootstrapped (n = 1,000) univariate linear mixed models (LMM) with 95 % confidence intervals indicated in brackets. Negative signs were added to account for negative relationships (i.e., negative regression coefficients) – although R^2 values are positive by definition.

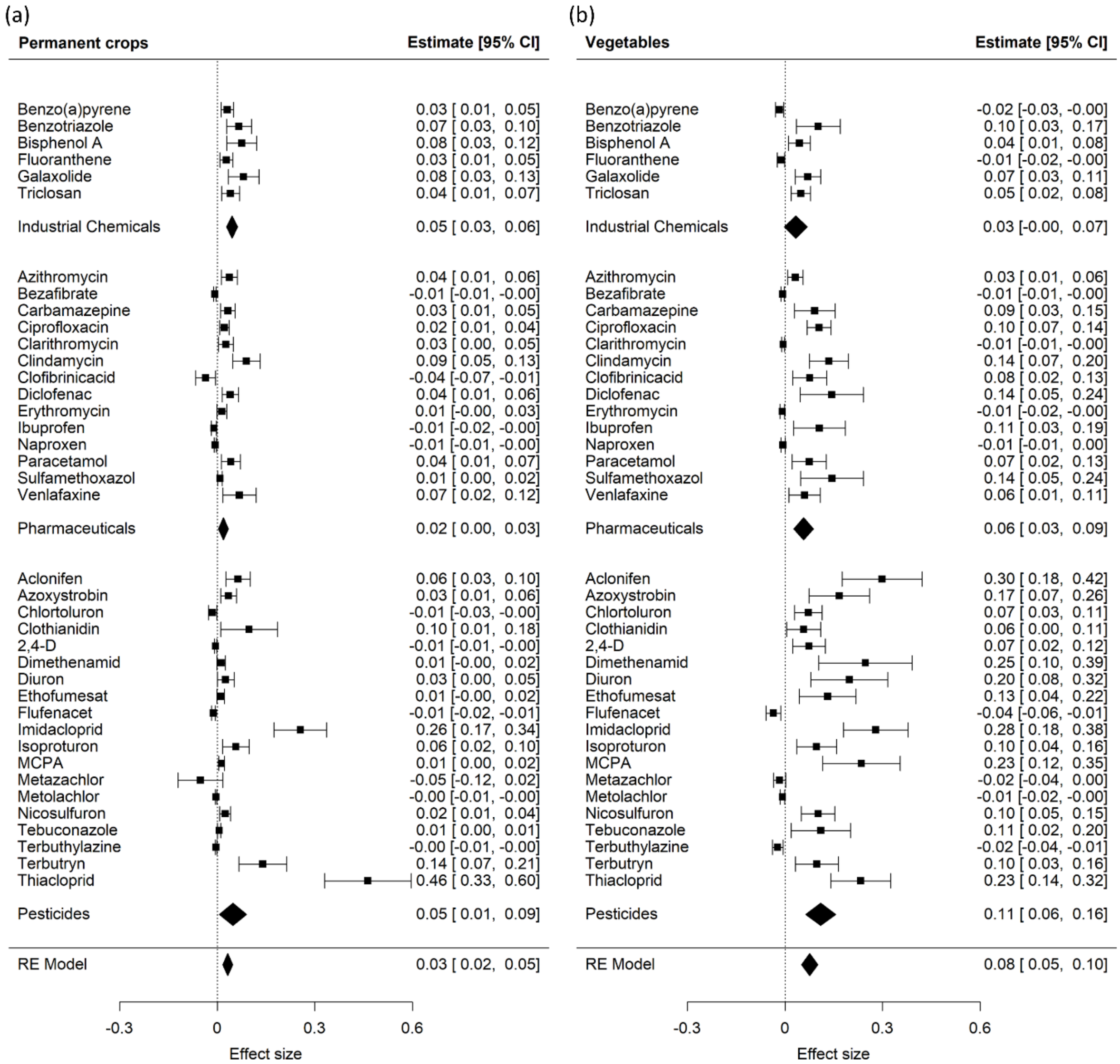


Figure 5: Relationship (effect size) of percent permanent crops (a) and vegetables (b) in the catchment with micropollutant concentrations. Effect sizes represent model fits (pseudo-R^2) derived from bootstrapped (n = 1,000) univariate linear mixed models (LMM) with 95% confidence intervals indicated in brackets. Negative signs were added to account for negative relationships (i.e., negative regression coefficients) – although R^2 values are positive by definition.

3.3 Link between micropollutant concentrations and individual crop types

In contrast to the overall weak effects of percent cropland in the catchment on the majority of micropollutants as described in the previous section, much more pronounced relationships on pesticides were evident for individual crop types. Permanent crops (vineyards, hops and orchards) were strongly related to two insecticides: thiacloprid (R^2 = 0.46) and imidacloprid (R^2 = 0.26.; Figure 5a). Vegetables also showed strong relationships to both insecticides (imidacloprid: $R^2 = 0.28$, thiacloprid: $R^2 = 0.23$) and in addition to the herbicides aclonifen ($R^2 = 0.30$) and dimethenamid ($R^2 = 0.25$; Figure 5b). Imidacloprid and thiacloprid are approved for, amongst others, applications to fruits and hops (both) and viticulture (imidacloprid), while aclonifen and dimethenamid are approved for various field crops and vegetables (Table 1; BVL, 2023a). Cereals and maize constitute the dominating crop types in this dataset of the Federal State of North Rhine-Westphalia and showed the strongest relationships to flufenacet ($R^2 = 0.29$ and $R^2 = 0.27$, respectively) and nicosulfuron ($R^2 = 0.16$ for both crop types; Figure A3.1, Supplementary Material). These herbicides are approved for field crops including maize (both) and cereals such as winter barley, winter rye, winter soft wheat (only flufenacet; Table 1, BVL, 2023a).

Strong relationships were also found between percent strawberry fields and dimethenamid ($R^2 = 0.40$) and diuron ($R^2 = 0.33$), and between percent asparagus fields, and dimethenamid and MCPA (both $R^2 = 0.33$; Figure A3.2, Supplementary Material). Interestingly, MCPA has been approved only for pome and stone fruits (e.g., apple or peach), but not for other fruits or vegetables (BVL, 2023a). Furthermore, percent onion fields was related to dimethenamid ($R^2 = 0.38$), imidacloprid ($R^2 = 0.36$) and aclonifen ($R^2 = 0.33$; Figure A3.3, Supplementary Material), all of which are approved for – and applied to cultivations of onions (BVL, 2023a). Percent oilseeds (e.g., rapeseed, sunflowers) showed comparatively weak relationships with pesticides (max. $R^2 = 0.11$ for 2,4-D; Figure A3.3, Supplementary Material).

4. Discussion

4.1 Micropollutant concentrations exceed regulatory assessment values

Several micropollutants were found to exceed existing regulatory assessment values at multiple sites. Especially, concentrations of pharmaceuticals, the non-steroidal anti-inflammatory drugs diclofenac and ibuprofen, the antibiotic azithromycin as well as concentrations of polycyclic aromatic hydrocarbons, benzo(a)pyrene and fluoranthene, exceeded assessment values, thus indicating a widespread and enhanced ecotoxicological risk for riverine biota (e.g, aus der Beek et al., 2016; Beckers et al., 2018; Markert et al., 2020). For pesticides, ecotoxicological risks were evident for less than 10% of sites and found only for the insecticide imidacloprid, while other pesticides (thiacloprid, azoxystrobin, nicosulfuron, flufenacet, dimethenamid) exceeded assessment values at less than 5% of sites. Pesticide risk assessment, however, substantially changed, when mixture toxicity risks were evaluated, which exceeded the threshold of one (SUM RQ > 1) at 55% of the sites. Thus, while for pharmaceuticals and industrial chemicals ecotoxicological risk were already driven by single substances, pesticide risks originate primarily from joint mixture toxicity risks.

Based on the (mixture) toxicity risk quotients calculated in this study, adverse effects of micropollutant exposure on river biota are very likely. The calculated risks, however, might underestimate actual toxicity risks, because micropollutant monitoring was based on grab sampling. In contrast to high-frequent and event-based monitoring, grab sampling is likely to miss the peak concentrations of micropollutants as they often occur directly after stormwater rain events (Munz et al., 2017; Halbach et al., 2021; Weisner et al., 2022). Moreover, the detection of (mixture) toxicity risks may also be limited by the selection and number of regularly measured micropollutants and their individual detection limits (Malaj et al., 2014). Toxicity risk assessment is often biased by missing or left-censored data (i.e. unknown concentrations between zero and the technical limit of quantification; von der Ohe et al., 2011), which constitutes a main obstacle for multivariate comparisons of effects among sites and in relation to potential sources and biological responses. Despite these sources of uncertainty in the chemical risk assessment, however, our findings confirm those of previous studies (e.g., Markert et al., 2020; Halbach et al., 2021; Finckh et al., 2022): regulatory assessment values for micropollutants are frequently exceeded in the aquatic environment so that freshwater biota are exposed to critical levels of both individual micropollutants and mixtures thereof.

4.2 Micropollutant concentrations relate to catchment land uses

Our results point at clear relationships between particular land use types and individual micropollutants as well as micropollutants groups. Cropland was particularly related to pesticide concentrations and less so to pharmaceuticals and industrial chemicals. This is in line with recent studies describing agriculture as a main determinant for pesticide exposure (Szöcs et al., 2017). Previous studies also suggested urban point sources to substantially contribute to pesticide pollution due to the use of pesticides in urban gardens or as biocidal products, for example in facade paints (Münze et al., 2017; Tauchnitz et al., 2020). We, however, found the major part of the monitored pesticides to relate to percent agricultural area in the catchment, except for terbutryn, which in fact is no longer approved for agricultural use but for biocidal use in facade paint; thus, the herbicide showed a stronger relationship to percent urban area. Percent urban area was found to be strongly associated with individual and mixture toxicity risks of pharmaceuticals and industrial chemicals (Ebele et al., 2017; Bradley et al., 2020). Apparently, percent cropland and urban area in the catchment can explain – and differentiate between – distinct patterns of micropollutant exposure. In contrast, percent forested and grassland area primarily showed a negative relationship to micropollutants, thus indicating that both forms of extensive land use relate to lower pollution (Goss et al., 2020; Dachbrodt-Saaydeh et al., 2021; Riedo et al., 2022).

4.3 Individual pesticide concentrations relate to crop-specific pesticide application

Our findings confirm that individual pesticide concentrations can be linked to individual crop types in the catchment of rivers (Andert et al., 2015; Dachbrodt-Saaydeh et al., 2021; Schürings et al., 2024b). Pesticide concentrations were strongly related to permanent crops and vegetables, in particular to onion fields. These crop types are associated with intensive pesticide application, in particular with insecticides (Dachbrodt-Saaydeh et al., 2021). Further studies reported a deterioration of riverine biota in agricultural catchments with a high areal coverage of permanent crops, vegetables, vineyards or orchards (Schulz, 2001; Bereswill et al., 2012; Xing et al., 2012; Schürings et al., 2024a). Cereals and maize showed weaker relationships to pesticide concentrations (except for the herbicides flufenacet and nicosulferon), which suggests a less intensive pesticide application connected to these crop types, except for herbicides (Andert et al., 2015; Roßberg, 2016). Although the uncertainties in the detection of pesticides in our data (see above) prevent us from drawing final conclusions as to the relationship between pesticides, insecticides in particular, and agricultural land uses (Weisner et al., 2022), our findings support the clear demand to distinguish between crop types. The use of rather general categories like 'cropland' in our study showed that relationships between individual herbicides and insecticides, and individual crop types would have been largely overlooked.

4.4. Implications for micropollutant risk assessment and management

This study shows that both percent urban and agricultural areas in the catchment of rivers are notably related to the micropollutant exposure in the rivers. Agricultural effects on micropollutant concentrations and joint mixture risks are not uniform and strongly vary between individual crop types. The mere differentiation between cropland and grassland does not adequately represent agricultural stress. Notably, the individual pesticides that were found to be strongly associated with individual crop types largely reflected their approved area of application in Germany (BVL, 2023a). Thus, in the absence of site-specific data on pesticide concentrations, percent area of individual crop types cultivated in the catchment (or at finer scales) may provide a good proxy to inform the assessment of potential toxicity risks (Schürings et al., 2024b). The same areal data could also support the identification of specific pollution sources and the assessment of (mixture) risks of micropollutants in the environment. In order to improve the assessment of (mixture) risks of micropollutants, chemical monitoring programs need to further implement high frequent and event-based monitoring or composite sampling (Bundschuh et al., 2014; Carvalho et al., 2019).

Industrial chemicals and pharmaceuticals were mainly related to percent urban area in the catchment, thus indicating a high relevance of urban point sources, especially wastewater treatment plants (aus der Beek et al., 2016). Therefore, advanced wastewater treatment using ozonation or activated carbon (or a mixture of both) require implementation to reduce the concentrations of micropollutants and hence the ecotoxicological risks originating from them (Bundschuh et al., 2011; Triebskorn et al., 2019; Finckh et al., 2022; Kienle et al., 2022; Spilsbury et al., 2024). However, advanced wastewater treatment cannot remove all micropollutants and neither can it remove the secondary (transformation) products that result, for example, from the ozonation of primary pollutants (Bundschuh et al., 2011).

Intensive agriculture constitutes another major source of micropollutants that imposes strong negative effects on riverine biota (Hughes & Vadas Jr, 2021; Schürings et al., 2022). In contrast to waste water treatment plants, the diffuse pollution (and related ecological risks) from agricultural areas cannot be reduced by selective local measures (Rothe et al., 2021). Instead, agricultural approaches minimizing or eliminating pesticide application, such as integrated pest management, organic farming, agroecology or precision agriculture (Gebbers & Adamchuk, 2010; Barzman et al., 2015; Reganold & Wachter, 2016; González-Chang et al., 2020) are required. Additionally, constructed wetlands, vegetated buffer strips and riparian vegetation have been shown to reduce pesticide exposure in surface waters (Stehle et al., 2011; Lerch et al., 2017; Turunen et al., 2019; Vormeier et al., 2023). However, these approaches rely on substantial changes in agricultural management and successful implementation of ambitious regulations (Pe'er et al., 2022).

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Data availability

The authors do not have permission to share data on micropollutant concentrations, but data are available on request from the corresponding agency North Rhine-Westphalia Office of Nature, Environment and Consumer Protection. Land use data were derived from Blickensdörfer et al. (2022).

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Supplementary Materials to

Water Framework Directive micropollutant monitoring mirrors catchment land use: Agricultural and urban sources revealed

Table A1: Statistical parameters of sampling site characteristics

	Minimum	Maximum	Mean	SD
Catchment area [km²]	5	2834	316	503
Altitude [m]	12	465	123	101

Table A2: Statistical parameters of SUM RQ values of micropollutant groups

	Min. SUM RQ	Max. SUM RQ	Mean SUM RQ	SD SUM RQ	% Sites with SUM RQ > 1
Industrial Chemicals	0.0	169.5	12.9	18.3	100
Pharmaceuticals	0.0	108.5	12.9	11.6	100
Pesticides	0.0	80.1	4.1	5.6	55
Herbicides	0.0	26.7	0.7	1.7	27
Insecticides	0.0	80.0	3.3	5.2	44
Fungicides	0.0	3.9	0.1	0.2	3

A3: Effect sizes of land use for micropollutants (LMM)

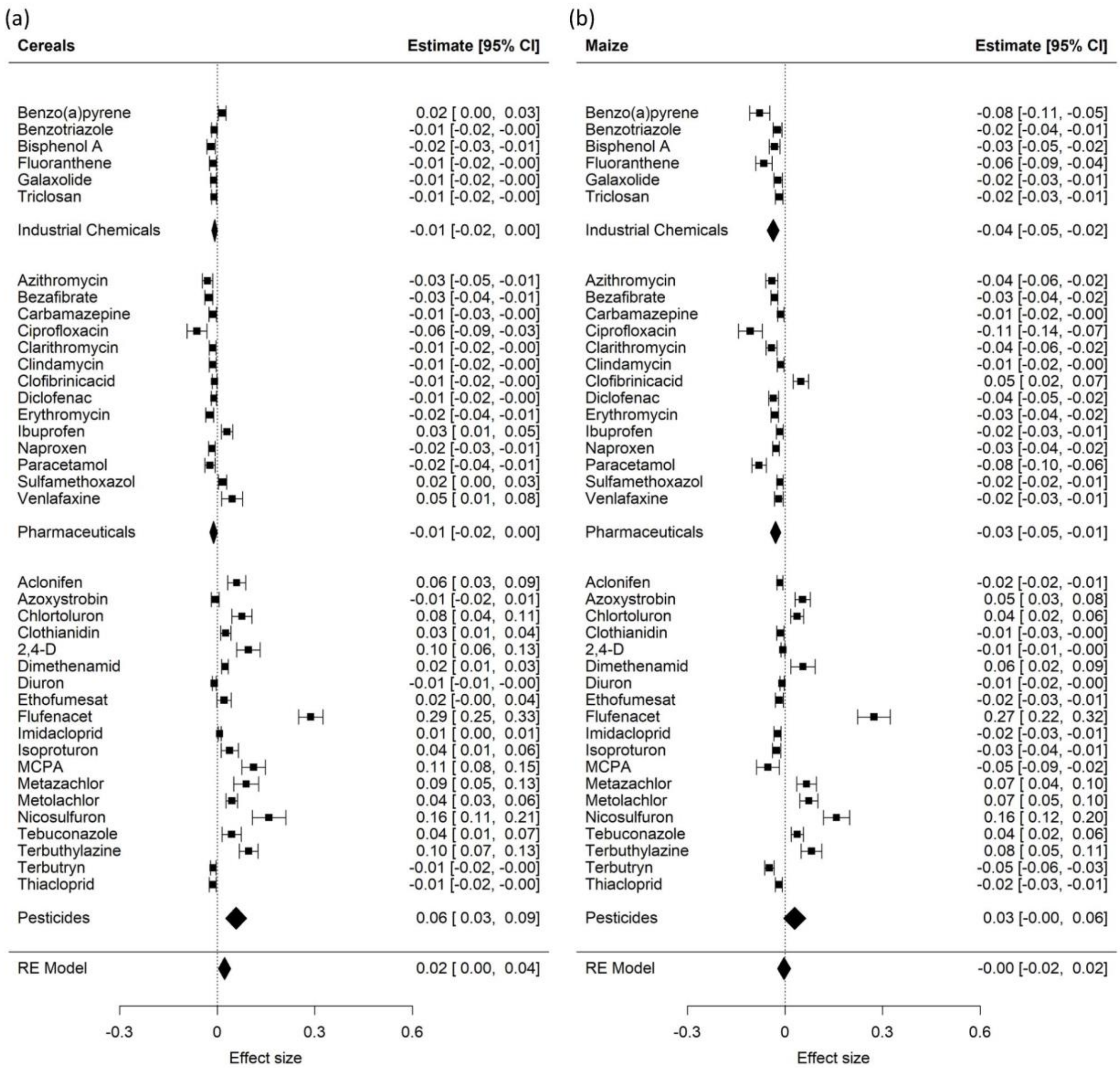

Figure A3.1: Relationship (effect size) of percent cereals (a) and maize (b) in the catchment with micropollutant concentrations. Effect sizes represent model fits (pseudo-R2) derived from bootstrapped (n = 1,000) univariate linear mixed models (LMM) with 95% confidence intervals indicated in brackets. Negative signs were added to account for negative relationships (i.e., negative regression coefficients) – although R^2 values are positive by definition.

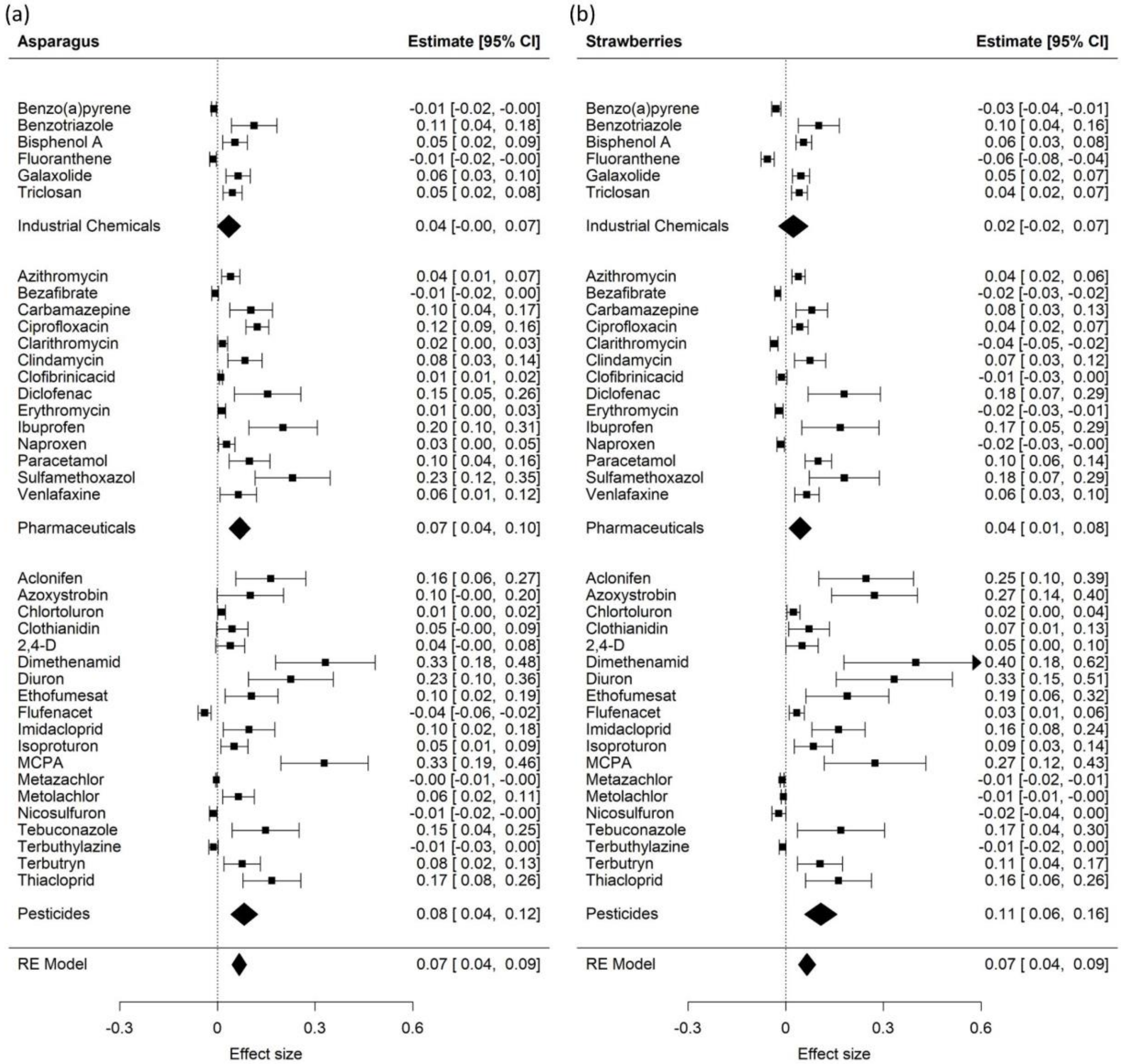

Figure A3.2: Relationship (effect size) of percent asparagus (a) and strawberries (b) in the catchment with micropollutant concentrations. Effect sizes represent model fits (pseudo-R2) derived from bootstrapped (n = 1,000) univariate linear mixed models (LMM) with 95% confidence intervals indicated in brackets. Negative signs were added to account for negative relationships (i.e., negative regression coefficients) – although R^2 values are positive by definition.

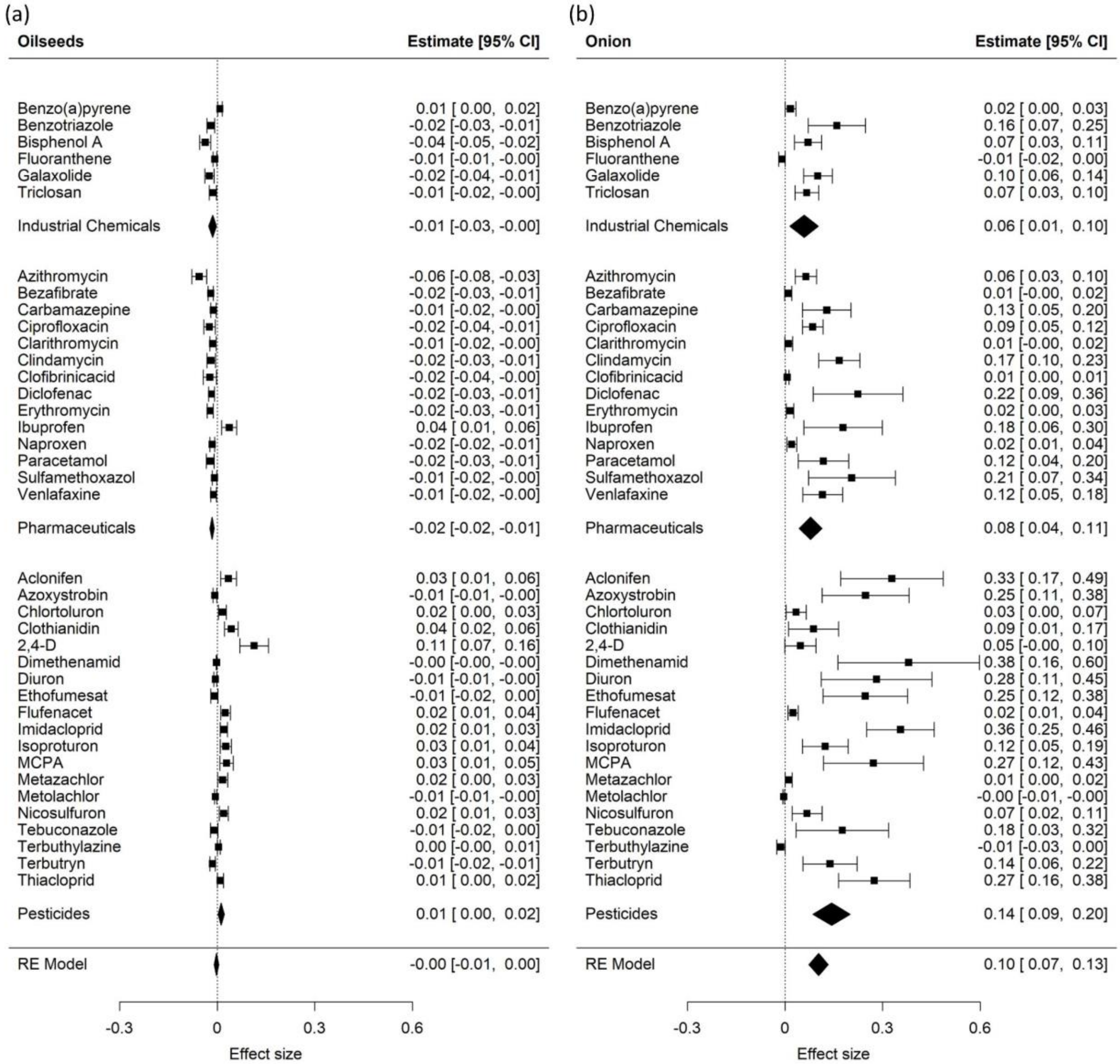

Figure A3.3: Relationship (effect size) of percent oilseeds (a) and onion (b) in the catchment with micropollutant concentrations. Effect sizes represent model fits (pseudo-R2) derived from bootstrapped (n = 1,000) univariate linear mixed models (LMM) with 95% confidence intervals indicated in brackets. Negative signs were added to account for negative relationships (i.e., negative regression coefficients) – although R^2 values are positive by definition.

Author contributions

Titel: *Water Framework Directive micropollutant monitoring mirrors catchment land use: Agricultural and urban sources revealed*

Authors: Markert, N., Schürings, C., & Feld, C. K.

Contributions:

- Conception – 50%
- Conduction of experimental work – not applicable
- Data analysis – 50%
- Species identification – not applicable
- Statistical analysis – 50%
- Writing the manuscript – 50%
- Revision of the manuscript – 40%

Signature of the Doctoral Candidate | Signature of the Doctoral Supervisor

Chapter 5

Assessment of cultivation intensity can improve the correlative strength between agriculture and the ecological status in rivers across Germany

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Assessment of cultivation intensity can improve the correlative strength between agriculture and the ecological status in rivers across Germany

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A R T I C L E I N F O

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Nutrients
Pesticides

A B S T R A C T

Agriculture has been identified as a main cause for more than 90% of Germany's rivers still not meeting good ecological status in 2021. While many large-scale studies observed a negative effect of catchment agricultural land use on river biota, they rarely considered differences in cultivation intensities, although small-scale studies highlight clear differences between the effects of agricultural crops. Here we used Germany-wide and spatially explicit information on crop types to calculate agricultural intensity indices for nutrients and pesticides, weighting different crop types based on average pesticide treatment and nutrient application rates. These indices were then used as explanatory variables for the ecological status of n = 7677 biological sampling sites. Pesticides were more important than nutrient pollution for macroinvertebrates and macrophytes, while diatoms were more sensitive to nutrients. Considering the most relevant intensity index (pesticide or nutrient) slightly increased the correlative strength with ecological status, as compared to the correlation with agricultural land or cropland cover by up to $R^2 = 0.14$ for diatoms. Correlative strength of agricultural intensity indices was substantially larger in small mountain and (pre)-alpine streams compared to lowland streams, with an R^2 up to 0.43 for macroinvertebrates. These results not only confirm previous large-scale studies by demonstrating the detrimental effects of present-day agriculture on river biota, but also shed light on the main pathways involved, particularly highlighting the adverse impacts of agrochemicals. Consequently, to protect river biota, a shift to more sustainable agricultural practices, like reducing pesticide application, is urgently required.

1. Introduction

Agricultural transition as targeted by the Convention on Biological Diversity (CBD, 2022) is urgently required to protect biodiversity, because current agricultural practices strongly impair both terrestrial organisms such as mammals (Janova and Heroldova, 2016), birds (Rigal et al., 2023) and flying insects (Börschig et al., 2013) as well as river biota (Vörösmarty et al., 2010). There is strong empirical evidence for the negative effect of agricultural land use on various riverine organisms (Schürings et al., 2022), including macroinvertebrates, diatoms, macrophytes and fish, notwithstanding large between-study heterogeneity. Although positive effects of agricultural land use are observed in exceptional cases (e.g. Townsend, et al., 2004; Niyogi et al., 2007), most large-scale studies found negative impacts on biota with increasing shares of agriculture adjacent to rivers (e.g. Turunen et al., 2016; Feld, 2013). Agriculture has even been identified as the key factor for freshwater deterioration (Wolfram et al., 2021) and consequently is one of the main reasons for more than 90% of German rivers failing good ecological status in 2021 (UBA, 2022).

These studies indicate that the share of agricultural land use in the upstream catchment is already a good proxy for the agricultural stress exerted on river biota. However, it does not reflect differences in agricultural cultivation intensities between crop types, referred to as agricultural intensity in the following. Depending on agricultural intensity, the effects on aquatic communities can strongly differ, as shown in many small-scale studies (Weijters et al., 2009). In Germany, average maize fields are fertilized with approximately twice the amount of nitrogen and phosphorus compared to cereal fields, three times more than vegetables, and at least ten times more than permanent crops such as orchards and vineyards (Britz and Witzke, 2014). In contrast, permanent crops such as orchards receive the highest pesticide treatment, which is on average three times higher compared to vegetables, more than five times higher compared to cereals, and even 15-fold higher compared to maize (Andert et al., 2015; Dachbrodt-Saaydeh et al., 2021).

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These differences in agricultural intensity are reflected in the stress exerted on rivers. Permanent crops such as orchards and vineyards have been linked with high pesticide concentrations in rivers and river sediments (Schulz, 2001; Bermúdez-Couso et al., 2007). Corn farming adjacent to rivers can cause massive fine sediment influx combined with strong phosphorus loads (Secchi et al., 2011), while intensive livestock farming in the river catchment is prone to nutrient influx following heavy rainfall events (Mouri and Aisaki, 2015). These stressors are known to impact river biota through many pathways:

Macroinvertebrates have been found to be strongly sensitive to agrochemicals (Berger et al., 2016) and fine sediments eroded from agricultural land, clogging interstitial spaces on the stream's bottom and covering lentic zones (Gieswein et al., 2019; Davis et al., 2022). For sensitive macroinvertebrates, pesticides were even identified as the most important stressor (Liess et al., 2021). While for both macrophytes and diatoms negative effects of pesticides have also been observed (Debenest et al., 2010; Ribeiro et al., 2019), macrophytes have been found mainly depending on river morphology (Kaijser et al., 2022) and to some extent also on nutrients (van Zuidam and Peeters, 2013, O'Hare et al., 2018), whereas diatoms most strongly responded to eutrophication (Giorgio et al., 2016; O'Hare et al., 2018). Therefore, organism groups should be distinguished when investigating the effect of agricultural intensity resulting from different crop types on river biota.

Furthermore, the relationships between agricultural intensity and river biota are likely to differ between ecoregions because stronger effects in mountain compared to lowland streams were already observed in studies using the sheer share of agriculture to quantify agricultural stress (Feld, 2013, Li et al., 2018). Possible reasons may be truncated gradients in lowlands yielding smaller effect sizes (Mack et al., 2022), but also differences in land-use legacies. Since the mid-19th century, European lowland rivers have been strongly degraded by adjacent agriculture, in particular pesticides, nutrients, and morphological alterations (Feld, 2013), which likely still mirrored by the degree of hydromorphological alterations (Greenwood et al., 2012). Moreover, Leps et al. (2015) found that the effect of agricultural stress on river biota depends on river size, with smaller associated effects for large rivers compared to small streams. Hence, analysing stream types separately is advisable.

These differences in agricultural intensity and biological effects resulting from different crop types are likely responsible for a substantial part of the between-study heterogeneity of large-scale studies, which used the sheer share of catchment agricultural land use to quantify agricultural stress (e.g. Turunen et al., 2016; Davis et al., 2022). Such crop-specific differences have not yet been considered in large-scale empirical studies, because respective data on crop types to assess agricultural intensity are usually not available at large spatial scales. Only recently, spatially explicit high-resolution maps on crop types were published, such as the German-wide maps of Blickensdörfer et al. (2022). These maps allow for large-scale crop-specific analyses of agricultural effects on river biota, to complement and upscale the existing small-scale case studies on the effects of nutrients and agrochemicals, which found differences in the organism groups' responses to agricultural stress.

Against this background, this large-scale study aimed at analysing the impact of agricultural land use on river biota, considering agricultural intensity by calculating pesticide and nutrient intensity indices based on typical application rates for different crop-types and recently published crop-type maps for Germany. More specifically, we hypothesized that: (1) Organism groups differ in their sensitivity to agricultural stress, with macroinvertebrates being more sensitive to pesticides, while macrophytes and diatoms are more sensitive to nutrients. (2) Considering agricultural intensity increases the correlative strength of the relationships between freshwater biota and agriculture compared to the sheer share of agriculture or cropland cover. (3) The effect of agricultural intensity clearly differs between ecoregions, with the strongest effects in mountain and (pre)-alpine streams.

2. Methods

2.1. Biological data

Biological data on n= 7677 sampling sites were acquired from all German federal states, except the Saarland, which are partly autonomous regions within the federal republic of Germany, with samples taken between 2010 and 2019. The sampling sites were grouped into the following five groups of river types (Pottgiesser and Sommerhäuser, 2008) that were investigated separately (from here on referred to as stream type groups). This was done to reflect the main differences in river size (streams vs. rivers) and between the three main ecoregions, resulting in differences in environmental factors such as soil types, flow velocity, or stream morphology, potentially influencing the effect of agricultural intensity on river biota. The following groups were considered: Small (n= 3065) and large (n= 924) mountain streams, small (n= 2648) and large (n = 630) lowland streams and (pre-)alpine streams (n= 410) (grouping of stream types see Table S1).

The biological samples were taken by the German federal agencies using standardized methods for ecological status assessment according to the EU Water Framework Directive in Germany. Macroinvertebrates were sampled according to a multi-habitat sampling method (Haase et al., 2004) and the species-level taxa lists were then processed by the online tool PERLODES (https://www.gewaesser-bewertung-berechnung.de/index.php/perlodes-online.html) to calculate a river-type specific multimetric macroinvertebrate index (n= 7677). In a similar way, macrophytes and diatoms were sampled according to a multi-habitat method (Schaumburg et al., 2012) and the species-level taxa lists were then processed by the online tool PHYLIB (https://www.gewaesser-bewertung-berechnung.de/index.php/phylib-online.html) to calculate both, a multimetric macrophyte index (n=2792 sampling sites) and diatom index (n=3307). The three biological metrics (macroinvertebrate index, macrophyte index, and diatom index) assess the ecological status and reflect the response of the three organism groups to various stressors, i.e. pollution or hydromorphological alterations (Böhmer et al., 2004). For sampling sites that had been sampled multiple times, the sample closest to 2018 was chosen to describe the biological conditions the year after 2017 from which the land use data originate.

2.2. Catchment delineation

For each sampling site, terrestrial land use in the entire upstream catchment was quantified. First, the drainage basins upstream of the sampling sites were derived from a digital elevation model (DEM) with 10 m resolution using ArcView 3.3 and subsequently visually checked. Second, the percentage cover of 23 different crop types was quantified using ESRI ArcGIS Pro 2.9.0 and Spyder (Phyton 3.7). This was based on land use data for 2017 derived through random forest classification of Sentinel-2, Landsat 8 and Sentinel-1 data with a 10 m resolution by Blickensdörfer et al. (2022).

2.3. Agricultural land use

To quantify agricultural land use in the catchments, the 23 different crop types distinguished by Blickensdörfer et al. (2022) were grouped in three different ways. First, all crop types were merged to calculate the area in the catchments covered by any type of agricultural land use, referred to as agriculture in the following, with an average catchment coverage of 44.46% (SD = 22.38). Second, the area covered by all crop types except for grassland was calculated (referred to as cropland in the following), given that grassland is the by far least intensive agricultural land use type, and this distinction has already been made by other large-scale studies with an average catchment coverage of 31.92% (SD = 23.36). Third, the area in the catchments covered by the different crop types (including grassland) was weighted based on pesticide treatment

(Dachbrodt-Saaydeh et al., 2021) and nutrient application rates, in particular nitrogen and phosphorus (Britz and Witzke, 2014), referred to as the pesticide-, nitrogen-, and phosphorus indices in the following. The weights for the pesticide intensity index were derived by allocating the 23 crop types of Blickensdörfer et al. (2022) to the 15 crop types distinguished by Dachbrodt-Saaydeh et al. (2021), who assessed pesticide treatment based on data from 90 farms distributed across Germany for the year 2017, supplemented with the information on pesticide treatment for legumes reported in Andert et al. (2015). For grassland, no pesticide treatment was assumed. Subsequently, the percentage area covered by the 23 crop types in the catchments was weighted based on the average pesticide treatment (Table S2), ranging between 0 and 1, resulting in values for the pesticide index. Similarly, the 23 crop types of Blickensdörfer et al. (2022) were allocated to the 16 crop types (including grassland), for which the mean nutrient application rates were derived from the CAPRI model (Britz and Witzke, 2014) for the years 2011–2013, separately for nitrogen and phosphorous (used as baseline for the CAPRI model). The crop-specific nutrient applications used in the CAPRI model draw from European statistics, including the Land Use / Cover Area Frame Statistical Survey (LUCAS), EUROSTAT, and FAOSTAT. They are estimated based on the overall nutrient input per region and area, crop-specific nutrient and phosphorous consumption and nutrient loss from harvest, biological fixation, and nutrient transformation processes. Again, the percentage area covered by the 23 crop types in the catchments (including grassland) was weighted accordingly (Table S2), resulting in values for the nitrogen index and the phosphorus index. Forest land use was disregarded for all grouping, as pesticide and nutrient application rates are generally not available (Halbach et al., 2021) and observed concentrations are low, wherefore forests are often regarded as refuges (Schneeweiss et al., 2022).

2.4. Statistical analysis

To test hypothesis 1, that organism groups differ in their sensitivity to agricultural stressors (pesticides, nitrogen, and phosphorus), we used random forest models as implemented in the R-package randomForest (v4.7–1.1). This method was chosen because no prior decision on distribution is required and potential non-linear relations can be assessed. We ran the models with ntree = 1000 and default settings separately for each combination of the three organism groups and five stream groups, resulting in a total of 15 models. The ecological status of the respective organism group was used as response and the following variables were used as predictors: the three agricultural intensity indices, river type (Pottgiesser and Sommerhäuser, 2008) to account for i.a. altitude and size of rivers, year of biological sampling to account for differences in agricultural land use between years, federal states to account for differences in environmental conditions and possible differences in sampling, and the water body category, i.e. whether a sampling site was identified as natural or heavily modified according to Article 4 of the EU Water Framework Directive. Then we calculated the relative variable importance of the three intensity indices to identify the intensity index capturing the most variance relative to the other two indices for each of the 15 models. Relative variable importance is a measure of how well the intensity indices are predicting changes in the response and can therefore be considered a proxy for the strength of the correlation. For each of the 15 combinations of organism groups and stream groups (15 models), the relative importance of the three intensity indices sums up to 100% and the index with the highest relative variable importance was selected for further analyses.

To investigate hypothesis 2 of higher correlative strength between agriculture and freshwater biota, when agricultural intensity is considered, we fitted Generalized Linear Mixed Models (GLMMs) with the gamlss package in r (v5.2–0). A logit link was used in the GLMMs as well as a beta distribution (BEINF) given that the ecological status used as response can take values from zero to one. More precisely, zero-one inflated beta models were used to allow the ecological status to take the exact value of zero and one. We built separate models for each combination of the three organism groups as response and three different ways to group the 23 crop types as predictors (percentage cover of agriculture, cropland, and the intensity index with the highest relative variable importance), called fixed effects in GLMMs, resulting in nine different models. Only one fixed effect was used in each model to adequately compare the different ways to group the crop types and avoid co-correlation issues between the different ways of grouping the crop types, wherefore we refrained from also using the intensity indices with lower relative importance.

These nine models were further subdivided into models for each of the five groups of sampling sites (small and large mountain and lowland streams, as well as (pre-)alpine streams), resulting in a total of $9 \times 5 = 45$ models. In addition to the single fixed effect per model, the following random effects were included to account for general differences in the ecological status resulting from these random variables: river types, year of biological sampling, federal state, category (natural vs. heavily modified water body). The fixed variables (percentage cover of agriculture and cropland and the intensity indices) were square root transformed to improve the models. For each of the 45 models, 70% of the data were bootstrapped, and cross-validation was applied to the remaining 30% for 1000 iterations, each to calculate a mean-pseudo-R^2 (from here on referred to as R^2) for the fixed effect, including confidence intervals. The R^2 of the fixed effect was calculated as the squared correlation between the fitted response and the predicted response solely based on the fixed effects (the R^2 of the full models are shown in Table S3. The models were checked visually for residual distribution against predicted values and each variable, yielding centered averages and symmetrical distributions.

To investigate hypothesis 3 of clear differences in agricultural effects among ecoregions, the same models already built for testing hypothesis 2 were used, focusing on the difference between the stream type groups.

3. Results

3.1. Relative importance of agricultural intensity indices (pesticide, nitrogen and phosphorus) differed between organism groups

Our first hypothesis on differences in the sensitivity of the three organism groups to the three agricultural stressors (pesticides, nitrogen, and phosphorus) was largely supported by the results of the random forest models. Macroinvertebrates were indeed most sensitive to pesticides (Table 1). In the five random forest models for macroinvertebrates in the different stream groups, the relative importance of the pesticide index ranged between 39% and 49% and was always larger compared to the relative importance of the nitrogen (29–34%) and phosphorus index (22–30%). For both macrophytes and diatoms, the differences in relative importance of the three agricultural intensity indices in each of the five models were smaller compared to the models for macroinvertebrates. Other than expected, macrophytes were generally not most sensitive to nutrients but to pesticides (Table 1). In four out of the five models, relative importance values for the pesticide index were larger (35–41%) compared to the other two intensity indices, and the phosphorus index was only largest in the model on the (pre)-alpine stream group (37%). For diatoms, the relative importance of the phosphorus index (34–39%) exceeded the importance of the nitrogen index (30–34%) and pesticide index (30–33%) in all five random forest models for the different stream groups, in accordance with our expectations (Table 1). For the subsequent analyses to test hypotheses 2 and 3, the pesticide index was used as a proxy for agricultural intensity for macroinvertebrates and macrophytes, except for macrophytes in the (pre)-alpine stream group, while the phosphorus index was used for macropyhtes in the (pre-)alpine stream group and for diatoms in all five stream groups.

Table 1
For each of the three organism groups, relative importance of the three different agricultural intensity indices (pesticide, nitrogen, phosphorus) are given, separately for each stream group. The largest relative importance value for each model is given in bold. Relative importance was calculated based on random forest models with the three intensity indices as fixed effects and four additional random factors (river type, year of biological sampling, federal state, category).

	Mountain small	Mountain large	Lowland small	Lowland large	(Pre)-alpine
		Macroinvertebrates			
Pesticide index	**49%**	**41%**	**39%**	**37%**	**41%**
Nitrogen index	29%	31%	33%	33%	29%
Phosphorus index	22%	28%	28%	30%	30%
		Macrophytes			
Pesticide index	**41%**	**38%**	**36%**	**35%**	33%
Nitrogen index	32%	31%	34%	**35%**	30%
Phosphorus index	27%	31%	30%	30%	**37%**
		Diatoms			
Pesticide index	32%	32%	33%	33%	30%
Nitrogen index	29%	30%	33%	33%	34%
Phosphorus index	**39%**	**38%**	**34%**	**34%**	**36%**

3.2. Considering agricultural intensity slightly increased correlative strength

As expected in hypothesis 2, the correlative strength of the relationships between the ecological status and agriculture tended to be higher when considering agricultural intensity compared to using the sheer share of cropland or agricultural land, but differences were small in most of the 45 models. In general, the agricultural intensity indices and the percentage cover of cropland and agriculture were strongly correlated (Table S4). This was especially true for the agricultural intensity indices and the percentage cover of cropland, with Spearman ρ ranging between 0.77 and 0.98. Hence, the correlative strength only slightly differed between the models, with the intensity indices and percentage of cropland cover as fixed effects, and most confidence intervals overlapped (Figs. 1–3).

The highest overall correlative strength and differences between the agricultural intensity index and percentage cover of cropland and agriculture was found for macroinvertebrates' ecological status. The correlative strength R^2 of the pesticide intensity index as fixed effect was up to $R^2 = 0.43$, capturing 0.06 more than cropland cover ($R^2 = 0.37$) and 0.17 more compared to the percentage cover of agriculture ($R^2 = 0.26$) as fixed effect (Fig. 1). While the correlative strength of percentage agricultural cover (including grassland) was clearly lower in some of the five stream groups, confidence intervals of the percentage cover of cropland and pesticide intensity index overlapped in all five stream groups.

For macrophytes, the correlative strength of the relationships between the ecological status and agriculture was generally lower, and R^2 was only up to 0.20 (Fig. 2). Again, the correlative strength tended to be higher for the models with the pesticide or phosphorus intensity index as fixed effect (up to $R^2 = 0.20$) compared to the models ́percentage cover of cropland (up to $R^2 = 0.16$) or agriculture (up to $R^2 = 0.12$) as predictor, but confidence intervals strongly overlapped in all five stream groups.

For diatoms, the correlative strength was similarly low compared to the macrophytes' models, except for the (pre)-alpine stream group with R^2 up to 0.40 (Fig. 3). However, in all models, the correlative strength of the models with the phosphorus intensity index as fixed effect tended to be higher compared to the models with the sheer share of cropland and agriculture as predictors, even though most confidence intervals overlapped. At the same time, the correlative strength of the phosphorus index of $R^2 = 0.14$ was more than four times higher compared to the models with percentage cover of cropland ($R^2 = 0.03$) and agriculture ($R^2 = 0.03$) in one model. Other than for macroinvertebrates and macrophytes, the correlative strength of the diatom models did not tend to be higher for the percentage cropland compared to the percentage agriculture.

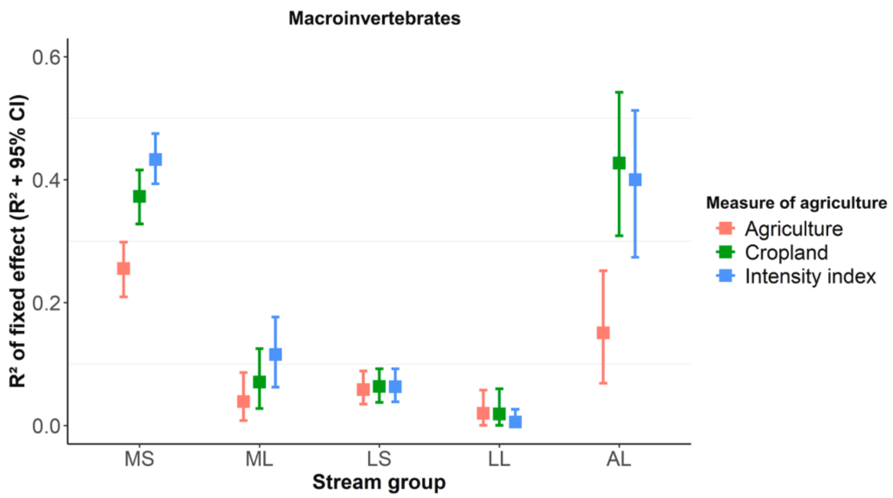

Fig. 1. Zero one inflated GLMMs with the macroinvertebrate's multimetric index (ecological status) as response. For each stream group (MS = Mountain Small, n=3065; ML = Mountain Large, n=924; LS = Lowland Small, n=2648; LL = Lowland Large, n=630; AL = (Pre-)alpine, n=410) the correlative strength (R^2) of the three ways of grouping crop types in the catchment as fixed effect (percentage cover of agriculture and cropland, pesticide intensity index) is shown with 95% confidence intervals (based on bootstrapping with 1000 replications).

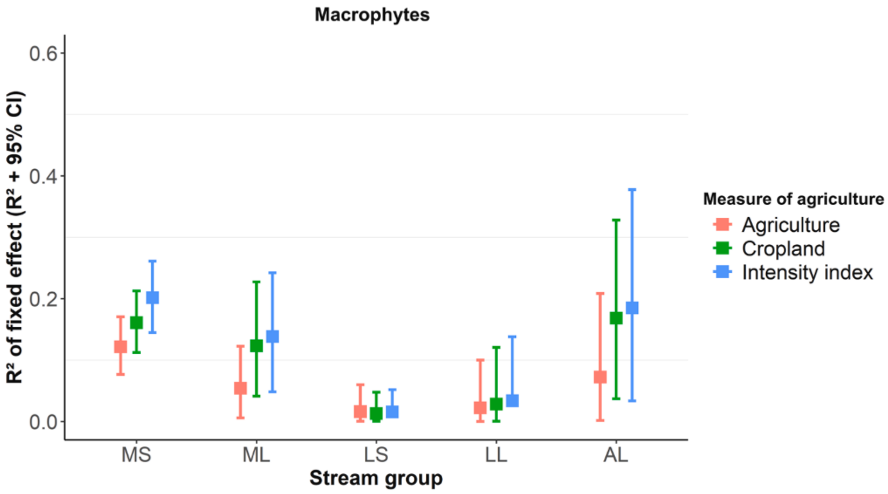

Fig. 2. Zero one inflated GLMMs with the macrophytes multimetric index (ecological status) as response. For each stream group (MS = Mountain Small, n=449; ML = Mountain Large, n=924; LS = Lowland Small, n=677; LL = Lowland Large, n=186; AL = (Pre-)alpine, n=232) the correlative strength (R^2) of the three ways of grouping crop types in the catchment as fixed effect (percentage cover of agriculture and cropland, pesticide Index for mountain- and lowland streams and Phosphor Index for (pre)-alpine streams) is shown with 95% confidence intervals (based on bootstrapping with 1000 replications).

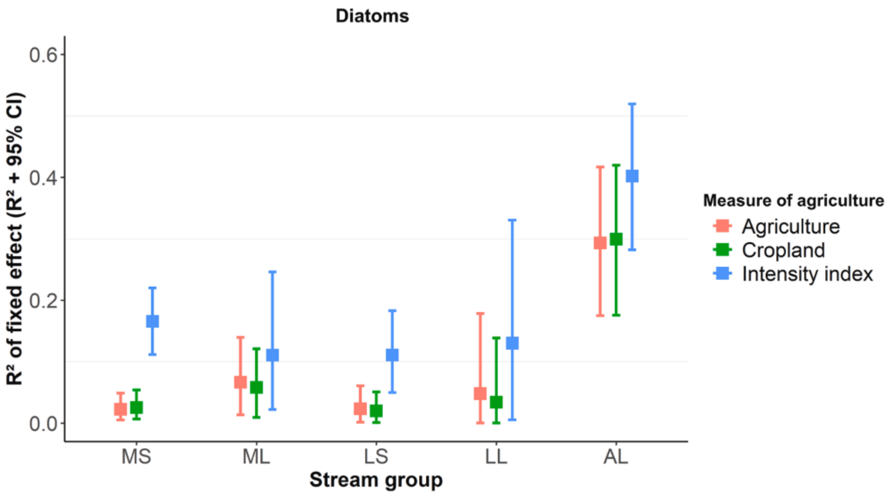

Fig. 3. Zero one inflated GLMMs with the diatom's multimetric index (ecological status) as response. For each stream group (MS = Mountain Small, n=1556; ML = Mountain Large, n=506; LS = Lowland Small, n=747; LL = Lowland Large, n=171; AL = (Pre-)alpine, n=327) the correlative strength (R^2) of the three ways of grouping crop types in the catchment as fixed effect (percentage cover of agriculture and cropland, phosphorus intensity index) is shown with 95% confidence intervals (based on bootstrapping with 1000 replications).

3.3. *The effect of agricultural intensity clearly differed between ecoregions and stream types*

As expected in hypothesis 3, the effect of agricultural intensity on the ecological status clearly differed between ecoregions and stream types, as reflected in the five stream type groups. For all three organism groups, agricultural intensity had the strongest effects in (pre-)alpine and small mountain streams. The variation in correlative strength among stream type groups was most pronounced for macroinvertebrates, with a correlative strength that was more than three times higher in small mountain streams (R^2 = 0.43) and (pre-)alpine streams (R^2 = 0.40), compared to large mountain streams with R^2 = 0.12 (Fig. 1). In small- and large lowland streams (R^2 = 0.06 and 0.01), the correlative strength was almost negligible. Regarding macrophytes, the correlative strength was smaller in general, and the differences were less distinct between small mountain streams (R^2 = 0.20) and (pre)-alpine streams (R^2 = 0.18), compared to large mountain streams with R^2 = 0.14 (Fig. 2). Similar to macroinvertebrates, small- and large lowland streams showed only minimal correlative strength (R^2 = 0.02 and 0.03). For diatoms, unlike macroinvertebrates and macrophytes, the correlative strength in the lowlands was not negligible and much higher for the intensity index compared to percentage cover of cropland and agriculture (Fig. 3). The differences between mountain and lowland streams were relatively small, ranging between R^2 = 0.11 in small lowland streams and R^2 = 0.17 in small mountain streams. However, large differences were observed when compared to (pre)-alpine streams (R^2 = 0.40).

4. Discussion

This large-scale study aimed at analysing the effect of agricultural intensity (as reflected by pesticides, nitrogen, and phosphorus) on three riverine organism groups (macroinvertebrates, macrophytes, diatoms) in five different types of streams (stream type groups). In contrast to many previous large-scale studies using the sheer share of cropland or agriculture (including grassland), agricultural intensity was assessed by calculating pesticide and nutrient intensity indices. The area of the upstream catchment of biological sampling sites covered by different crop types in maps recently published for Germany (Blickensdörfer et al., 2022) was weighted based on typical pesticide and nutrient application rates (Table S2) for the crop types (Britz and Witzke, 2014; Dachbrodt-Saaydeh et al., 2021). Even though the application data are affected by uncertainties caused by needed assumptions and upscaling, yielding the large spatial coverage of this study, they still offer crucial insight into crop-specific differences in farming practices.

The results might potentially be affected by a temporal mismatch between the crop data for 2017 and the year of the biological sampling (2010–2019) as well as years of fertilizer (2011–2013) and pesticide inputs (2017). However, crop-specific fertilizer and pesticides inputs only slightly varied in the last decade (Britz and Witzke, 2014; Dachbrodt-Saaydeh et al., 2021). Although the biological sample closest to 2018 was chosen to describe the biological conditions the year after 2017, the mean deviation of the sampling year from 2018 was nearly three years. However, even though the crops grown on specific fields change between years, the overall share of the individual crops at larger spatial scales, i.e. within the catchments, tended to change only slightly in the years 2017, 2018, and 2019 (Blickensdörfer et al., 2022), and analysis with data from 2018 gave similar results (Table S5). Still, the year of biological sampling was used as a random factor in the GLMMs to account for differences in the crops grown between years.

4.1. Relative importance of agricultural intensity indices on pesticide, nitrogen and phosphorus differed between organism groups

Our analysis highlights clear differences in the responses of the different organism groups to the agricultural stressors: nutrients and pesticides (Hypothesis 1). The expectation of higher pesticide sensitivity of macroinvertebrates was largely supported. The higher relative importance of pesticides for macroinvertebrates compared to the two nutrient intensity indices is consistent with recent findings in literature. While nutrients affect macroinvertebrates rather indirectly by increasing primary production and temporarily reducing oxygen concentrations (Dodds, 2006), which favors competitive species and eliminates sensitive species (Weijters et al., 2009), direct pesticide effects on macroinvertebrates have been shown in detail in many mesocosm studies (Roessink et al., 2013; Morrissey et al., 2015). Also in the field, effects of pesticide groups such as pyrethroid (Wurzel et al., 2020) have been observed with negative effects, particularly on sensitive aquatic insect larvae (e.g. Ephemeroptera and Trichoptera), which are particularly relevant for the assessment of the ecological status. Findings of Wernecke et al. (2019) indicate that pesticide mixtures, as present in real-life situations such as assessed in this study, are likely to have even larger effects. On that note, Liess et al. (2021) identified pesticides as the most relevant stressors for sensitive macroinvertebrates. Similarly, Wolfram et al. (2021) reasoned that agriculture is the most important stressor for macroinvertebrates, mainly based on pesticide pressure.

While we expected macrophytes to be most sensitive to nutrients following van Zuidam and Peeters (2013) and O'Hare et al., (2018), this was only true for the (pre-)alpine streams. The relative importance of the pesticide intensity index was higher in the other stream groups, suggesting higher pesticide sensitivity, likely caused by herbicides (e.g. Mohr et al., 2007). However, macrophytes seem to recover rather quickly after typical short-term pesticide peaks (King et al., 2016; Wieczorek et al., 2017), and the differences in relative importance between the intensity indices in the models on macrophytes were small. Several recent studies rather suggest that hydrological and morphological factors as well as river management may be more important for macrophyte occurrence than pesticides and nutrients (Baczyk et al., 2018; Kaijser et al., 2022), which also explains the relatively low explanatory power of our macrophyte models.

The results for diatoms, on the other hand, met our expectation of a stronger response to nutrients compared to pesticides, as suggested by the highest relative importance for the phosphorus index in all stream types. This is in line with the observations of Hilton et al., (2006) and O'Hare et al., (2018). Similar to macrophytes, diatoms also quickly recover after field typical pesticide peaks (Bighiu et al., 2020). However, they appear to strongly react to nutrients, and tolerant taxa may benefit, while sensitive taxa are eliminated (Kelly et al., 2009; Giorgio et al., 2016). The higher relative importance of the phosphorus index compared to the nitrogen index could be associated with the phosphorus limitation of diatoms (Bothwell and Kilroy, 2011). The relative importance particularly differs in the mountain streams, which could be related to different pathways of entry. Other than nitrogen, phosphorus mainly enters rivers bound to particles (Dorioz et al., 2006) and the steeper slopes in mountainous streams may result in fewer particles settling before reaching the rivers (Parkyn, 2004). Consequently, the phosphorus index may also indicate the impact of fine sediment on diatoms (Jones et al., 2017).

4.2. Considering agricultural intensity slightly increased correlative strength

Our expectation of an increase in the correlative strength of agriculture for the ecological status when cultivation intensity is considered (Hypothesis 2) could be only partly supported by the results, as the differences were relatively small, particularly between percentage cropland and the intensity indices. Only for macroinvertebrates and, to some extent, for macrophytes, the correlative strength strongly differed between percentage share of cropland and agriculture, which is likely because no pesticides are applied on grasslands (Riedo et al., 2022). As diatoms appear more sensitive to nutrients compared to pesticides (see above), the observed similar correlative strength between percentage share of cropland and agriculture for diatoms is likely caused by nutrient runoff into rivers caused by both fertilization of grasslands and nutrients from livestock farming (Mouri and Aisaki, 2015). Our observation that the correlative strength of the intensity indices were only slightly higher in comparison to percentage share of cropland can be explained by strong correlations between both indices (Table S4). This can be explained by the differences in area cover between the individual crops. Wheat, maize, rape seed, and barley account for more than 70% of the cropland area in Germany (Blickensdörfer et al., 2022), which are all known for relatively high nutrient (Britz and Witzke, 2014) but only medium pesticide application rates (Andert et al., 2015). Pesticide intensive crops such as vineyards, orchards, hops, and vegetables, on the other hand (Dachbrodt-Saaydeh et al., 2021), only cover less than ten percent of the cropland. Consequently, those crop types less frequently sum up to a percentage resulting in more severe stress than the ubiquitous crop types already impose. Aside from the strong correlation with cropland, the relatively low performance of the intensity indices may also be linked to the underlying data, particularly the underlying crop maps of Blickensdörfer et al. (2022), which do not allow discrimination between regional differences of agricultural practices such as organic farming. While nutrient application rates do not tend to differ strongly between conventional- and organic farming (Oelofse et al., 2010), no pesticides are applied in the latter resulting in lower pesticide residues in soils from organic farming (Geissen et al., 2021). Additionally, the soil conditions (Dobbie and Smith, 2003), slope (Ekholm et al., 2000; Cambien et al., 2020) and riparian vegetation (Palt et al., 2023) have been shown to strongly influence agricultural effects on river biota.

4.3. The effect of agricultural intensity clearly differed between ecoregions and stream types

As expected in hypothesis 3, the correlative strength between agricultural land use and the biota response differed between ecoregions and was higher in the small mountain streams and the (pre-)alpine region than in large streams, particularly in the lowlands. This coincides with the findings of Li et al. (2018), who showed that catchment land use is less important for large streams compared to low-order streams. This can be partly explained by the river continuum concept (Vannote et al., 1980), suggesting that species composition changes downstream from the sources and becomes more homogeneous. Moreover, small streams generally have smaller catchments, which are more likely to be dominantly covered by individual crop types, resulting in larger land use gradients, while in large catchments, differences in land use are more likely to average out. The difference in gradients may partly explain the difference in correlative strength between small and large streams (Mack et al., 2022), showing strong relations between gradients and effect sizes. While the correlative strength in the (pre)-alpine streams increased for diatoms using the phosphorus index, no increase (even a small decrease) was observed for the (pre-)alpine streams for the pesticide index compared to cropland for macroinvertebrates. A potential explanation for this may be that agricultural crop types with similar pesticide application rates are grown in the (pre)-alpine regions, so no differences are found. In addition, there is a strong correlation between the pesticide index and cropland of Spearman $\rho = 0.97$ in the (pre)-alpine region. Another explanation may be that (pre-)alpine regions are home to very pesticide sensitive species, which strongly react to small amounts of pesticides, so that the crop compositions do not matter too much. This is also supported by preliminary analysis (results not shown) using the SPEAR Index as a response measure, which was developed to identify species at risk of being affected by pesticides (Liess et al., 2008). The SPEAR Index showed the highest correlative strength of agriculture for macroinvertebrates in (pre-)alpine streams, compared to other ecoregions, where the ecological status was stronger correlated to the land use intensity indices. The overall small correlative strength of agriculture with the macroinvertebrates´ and macrophytes´ ecological status in the lowlands and, to some extend, also in large mountain streams is likely resulting from truncated gradients with fewer sampling sites in good ecological status, compared to (pre-)alpine and small mountain streams (Figures S1-S3) and can be explained by findings on the relation between gradient and effect size of Mack et al. (2022). Hence, subsequent analysis by dividing the agricultural gradient using recursive partitioning (Zeileis et al., 2008), a method recently used by Palt et al. (2023) to successfully disentangle the effects of riparian vegetation in agricultural catchments, showed much larger effects in the first part of the gradient with low agricultural intensity (Figures S4-S6). Reasons for the truncated gradients in the lowland rivers may be the land use past in the European lowlands, which degraded the rivers now lacking sensitive species (Feld, 2013). Those river ecosystems may already be stressed to such a degree that agricultural land use cannot deteriorate the ecological state much further.

5. Conclusion

The present study clearly shows that the effect of agriculture on river biota differs with agricultural intensity, environmental conditions, and organisms. Its results advise against oversimplified conclusions drawn based on single organism groups. On the contrary, the different organism groups strongly differ in their sensitivity to agricultural stressors. While macroinvertebrates respond most strongly to pesticides, diatoms appear to be more sensitive to nutrients. Macrophyte response is less clear and likely depends on hydromorphology. Weighting agricultural land use based on typical pesticide and nutrient application rates can slightly improve the correlative strength, even though information on regional practices such as organic farming is lacking on a large scale. The correlative strength of agriculture is highest in the mountainous ecosystems and (pre-)alpine regions and the effects are best captured at the catchment scale, explaining up to 43% of the ecological status. Consequently, the overall clear effect of agriculture on the ecological status and the strong relation to pesticides for macroinvertebrates and to nutrients for diatoms shows that a transition to more sustainable agricultural practices is urgently required to protect river biodiversity.

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.

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Appendix A. Supporting information

Supplementary data associated with this article can be found in the online version at doi:10.1016/j.agee.2023.108818.

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Assessment of cultivation intensity can improve the correlative strength between agriculture and the ecological status in rivers across Germany

Assessment of cultivation intensity can improve the correlative strength between agriculture and the ecological status in rivers across Germany

Supplementary material

Table S1: Allocation of river types defined by Pottgiesser & Sommerhäuser, 2008 to 5 stream groups.

Stream groups	Allocated River types
Small mountain streams	River type 05; River type 05.1; River type 06; River type 06_small; River type 7
Large mountain streams	River type 09; River type 09.1; River type 09.1_K; River type 09.2
Small lowland streams	River type 11; River type 14; River type 16; River type 18; River type 19
Large lowland streams	River type 12; River type 15; River type 15_large; River type 17
Alpine streams	River type 01.1; River type 01.2; River type 02.1; River type 02.2; River type 03.1; River type 03.2; River type 04

Table S2: Allocated standardized Nitrogen, Phosphorus and Pesticide application rates based on nutrient application data from the CAPRI mode (Britz & Witzke, 2014) and Pesticide Intensity data of Dachbrodt-Saaydeh et al. (2021). *Legume pesticide intensity was supplemented by Andert et al., 2015.

Crop type	Nitrogen application [0-1]	Phosphorus application [0-1]	Pesticide application [0-1]
Winter wheat	0.56	0.43	0.19
Winter barley	0.40	0.35	0.14
Winter rye	0.44	0.32	0.13
Other winter cereals	0.46	0.37	0.15
Spring barley	0.40	0.35	0.14
Spring oat	0.44	0.32	0.13
Other spring cereals	0.42	0.34	0.13
Winter rape	0.55	0.54	0.21
Sunflower	0.31	0.29	0.21
Maize	1	1	0.06
Other leafy vegetables	0.30	0.28	0.22
Potato	0.60	0.41	0.46
Sugar beet	0.97	0.93	0.15
Legume*	0.14	0.25	0.13

Carrot	0.35	0.28	0.19
Asparagus	0.35	0.29	0.24
Onions	0.35	0.29	0.33
Strawberry	0.35	0.29	0.24
Orchard	0.09	0.08	1
Vineyard	0.09	0.08	0.53
Hops	0.09	0.08	0.32
Other	0.43	0.38	0.23
Grassland	0.24	0.14	0

Table S3: Zero one inflated GLMMs including bootstrapping with macroinvertebrate´s, macrophyte´s and diatom´s multimetric index (ecological status) as response. For each stream group (MS = Mountain Small, ML = Mountain Large, LS = Lowland Small, LL = Lowland Large, AL = (Pre-)Alpine), the correlative strength (R^2) and confidence intervals of the full models with the three ways of grouping crop types in the catchments as fixed effect (percentage cover of agriculture, cropland and the intensity indices) and the random factors (river types, year of biological sampling, federal state, category (natural vs. heavily modified water body)) are shown for the years 2017 and 2018.

Macroinvertebrates						
	Agriculture	**95% CI**	**Cropland**	**95% CI**	**Intensity Index**	**95% CI**
MS	0.42	0.37 – 0.46	0.47	0.43-0.51	0.52	0.48-0.56
ML	0.24	0.17-0.32	0.29	0.21-0.37	0.34	0.26-0.42
LS	0.17	0.13-0.21	0.16	0.13-0.20	0.17	0.13-0.21
LL	0.13	0.06-0.21	0.13	0.06-0.21	0.13	0.07-0.21
AL	0.37	0.25-0.49	0.52	0.40-0.63	0.54	0.42-0.65
Macrophytes						
	Agriculture	**95% CI**	**Cropland**	**95% CI**	**Intensity Index**	**95% CI**
MS	0.17	0.12-0.23	0.17	0.12-0.23	017	0.15-0.20
ML	0.11	0.04-0.20	0.15	0.06-0.26	0.16	0.11-0.20
LS	0.01	0.00-0.04	0.01	0.00-0.04	0.03	0.01-0.06
LL	0.07	0.00-0.17	0.08	0.00-0.20	0.11	0.03-0.20
AL	0.12	0.02-0.27	0.16	0.04-0.32	0.18	0.03-0.37
Diatoms						
	Agriculture	**95% CI**	**Cropland**	**95% CI**	**Intensity Index**	**95% CI**
MS	0.54	0.49-0.60	0.54	0.48-0.59	0.53	0.51-0.55
ML	0.37	0.25-0.48	0.37	0.26-0.48	0.40	0.34-0.45
LS	0.44	0.36-0.52	0.44	0.35-0.52	0.45	0.41-0.49
LL	0.27	0.07-0.48	0.25	0.06-0.44	0.27	0.09-0.48
AL	0.34	0.22-0.46	0.37	0.23-0.50	0.41	0.30-0.53

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Table S4: Correlation of cropland with the different intensity indices across the different stream groups. Shown are the spearman rank correlation coefficients.

Correlation between cropland and intensity indices					
	Mountain small	Mountain large	Lowland small	Lowland large	Alpine
Pesticide Index	0.98	0.97	0.91	0.87	0.97
Phosphorus Index	0.77	0.85	0.77	0.77	0.94
Correlation between agriculture and intensity indices					
	Mountain small	Mountain large	Lowland small	Lowland large	Alpine
Pesticide Index	0.85	0.82	0.83	0.77	0.71
Phosphorus Index	0.77	0.85	0.78	0.79	0.85
Correlation between cropland and agriculture					
	0.90	0.88	0.94	0.94	0.75

Table S5: Zero one inflated GLMMs with macroinvertebrate´s, macrophyte´s and diatom´s multimetric index (ecological status) as response. For each stream group (MS = Mountain Small, ML = Mountain Large, LS = Lowland Small, LL = Lowland Large, AL = (Pre-)Alpine), the correlative strength (R^2) of the fixed effect for the three ways of grouping crop types in the catchment as fixed effect (percentage cover of agriculture, cropland and the intensity indices) are shown for the years 2017 and 2018.

Macroinvertebrates						
	2017			**2018**		
	Intensity Index	**Cropland**	**Agriculture**	**Intensity Index**	**Cropland**	**Agriculture**
MS	0.43	0.37	0.26	0.42	0.37	0.25
ML	0.12	0.07	0.04	0.11	0.07	0.04
LS	0.06	0.06	0.06	0.06	0.06	0.06
LL	0.01	0.02	0.02	0.01	0.02	0.02
AL	0.40	0.42	0.15	0.39	0.42	0.14
Macrophytes						
	2017			**2018**		
	Intensity Index	Cropland	Agriculture	Intensity Index	Cropland	Agriculture
MS	0.20	0.16	0.12	0.19	0.16	0.12
ML	0.14	0.12	0.05	0.13	0.12	0.05
LS	0.02	0.02	0.01	0.02	0.01	0.01
LL	0.03	0.03	0.02	0.03	0.01	0.01
AL	0.18	0.16	0.07	0.18	0.15	0.06
Diatoms						
	2017			**2018**		
	Intensity Index	Cropland	Agriculture	Intensity Index	Cropland	Agriculture
MS	0.17	0.03	0.03	0.13	0.02	0.02
ML	0.11	0.06	0.07	0.09	0.05	0.06
LS	0.11	0.02	0.02	0.09	0.02	0.02

LL	0.13	0.03	0.05	0.11	0.02	0.03
AL	0.40	0.30	0.29	0.39	0.29	0.29

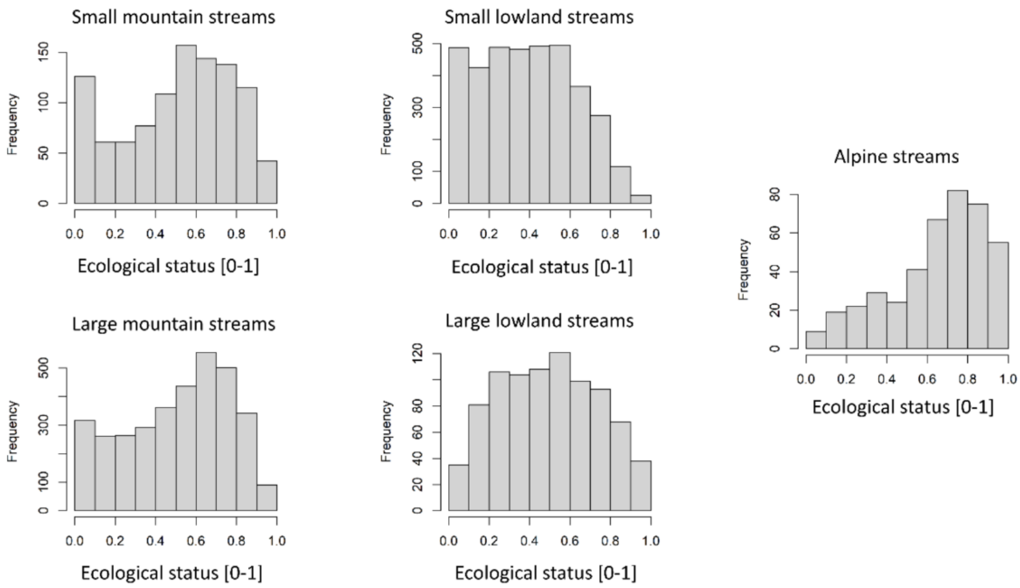

Figure S1: Histogram of the Macroinvertebrates´ ecological status across the different stream types.

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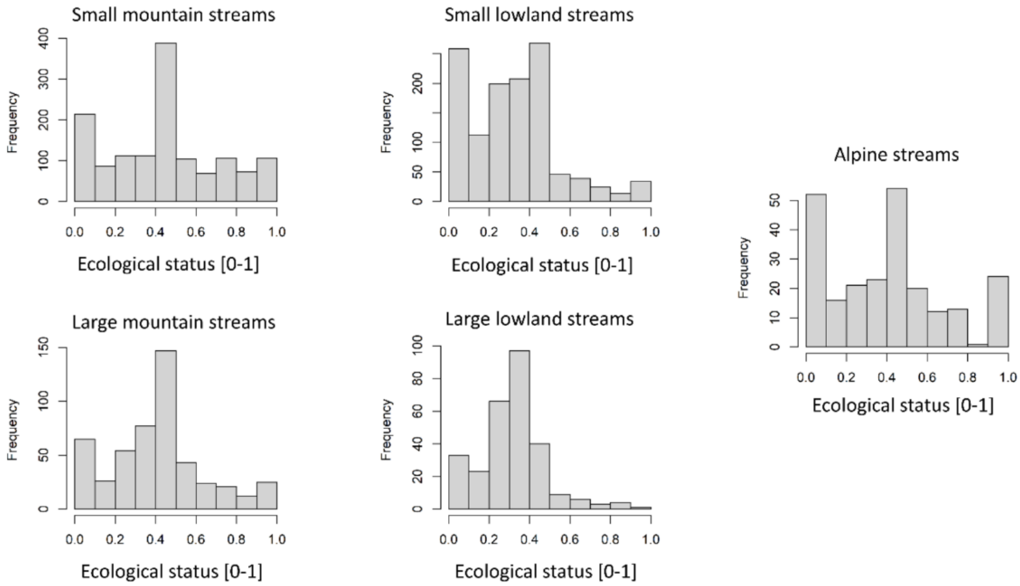

Figure S2: Histogram of the Macrophytes´ ecological status across the different stream types.

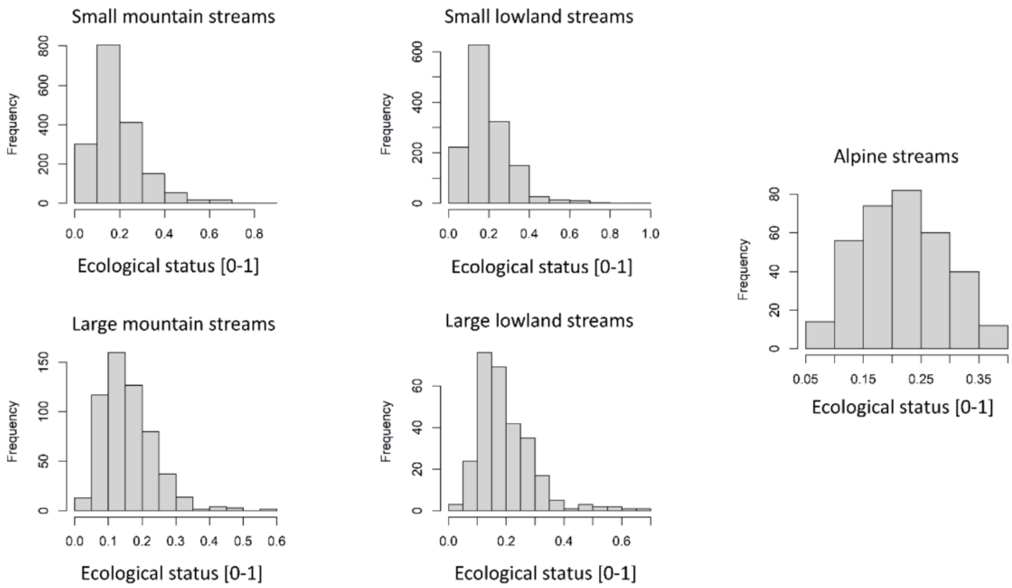

Figure S3: Histogram of the Diatoms´ ecological status across the different stream types.

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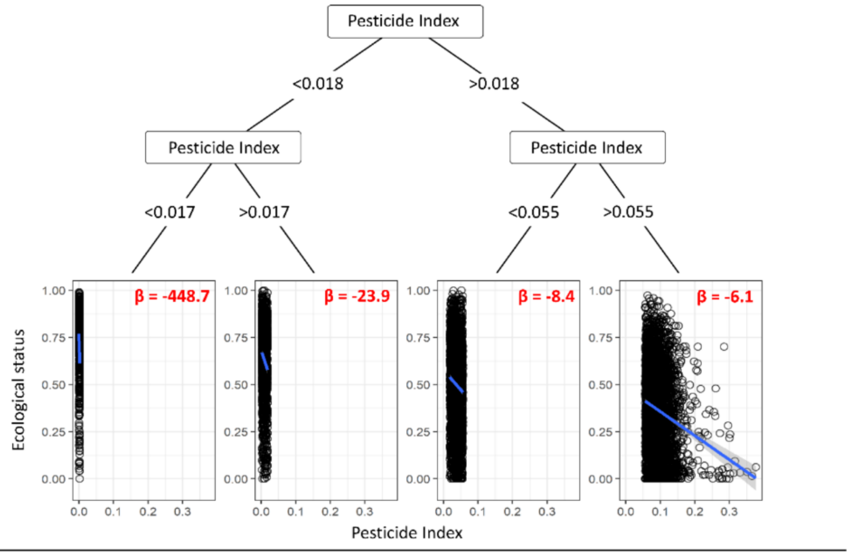

Figure S4: Partitioning tree of the for the pesticide index as fixed effect and the macroinvertebrates´ ecological status as response. Relationship between the ecological status and the pesticide index for each sub-dataset of the agricultural gradient with regression coefficients in red.

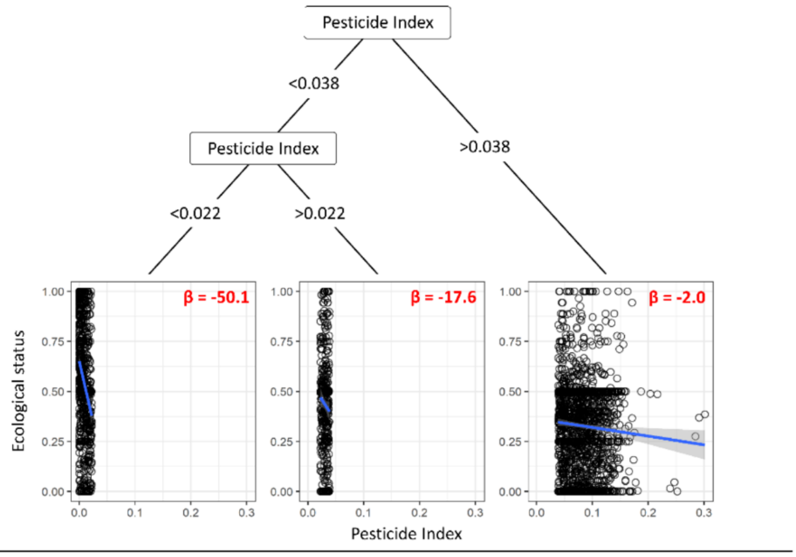

Figure S5: Partitioning tree of the for the pesticide index as fixed effect and the macrophytes' ecological status as response. Relationship between the ecological status and the pesticide index for each sub-dataset of the agricultural gradient with regression coefficients in red.

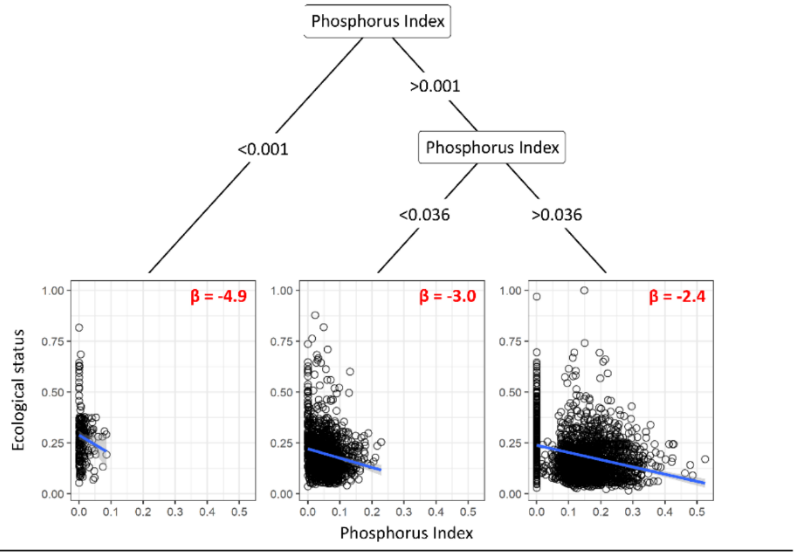

Figure S6: Partitioning tree of the for the phosphorus index as fixed effect and the diatoms´ ecological status as response. Relationship between the ecological status and the phosphorus index for each sub-dataset of the agricultural gradient with regression coefficients in red.

Author contributions

Titel: *Assessment of cultivation intensity can improve the correlative strengths between agriculture and the ecological status in rivers across Germany*

Authors: Schürings, C., Hering, D., Kaijser W., & Kail, J.

Contributions:

- Conception – 80%
- Conduction of experimental work – not applicable
- Data analysis – 80%
- Species identification – not applicable
- Statistical analysis – 70%
- Writing the manuscript – 100%
- Revision of the manuscript – 60%

Signature of the Doctoral Candidate

Signature of the Doctoral Supervisor

Chapter 6

Securing success for EU Nature Restoration Law

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POLICY FORUM

BIODIVERSITY

Securing success for the EU Nature Restoration

The law would complement many others, but challenges loom

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In an attempt to halt and reverse biodiversity losses, the European Commission has proposed a new regulation, the Nature Restoration Law (NRL). It could become a cornerstone of Europe's ambitions to restore biodiversity and ecosystem services for decades to come (*1*) and demonstrate global leadership in addressing ongoing environmental crises. The draft of the law, which is a first globally, has been under political pressure from various sides, and scientists have contributed intensively to the discussion (*2*). After trilogue negotiations among the European Parliament, the Council of Europe, and the European Commission, the final text of the NRL has been agreed on (see the figure). However, it will still be subject to final votes within the Council and Parliament. Here, we assess the potential for the NRL to overcome problems associated with implementation of related European Union (EU) legislation, strategies, and policies and what can be learned for implementation of the NRL.

The NRL acknowledges that existing EU legislation and policies have so far failed to halt biodiversity losses (*1*) and consequently, without new instruments, cannot meet the targets of international agreements, such as the Kunming-Montreal Global Biodiversity Framework. Although some of the NRL's aims and approaches overlap with other EU directives, strategies, and policies, in particular with the EU Biodiversity Strategy for 2030, the NRL is distinct in terms of its coverage targeting the majority of European ecosystem types, its strong focus on restoration, and its provision of binding targets and clear timelines. This potential for regulatory power may largely explain the contested nature of its passage into legislation.

The prospect of the NRL achieving its aims will be strongly determined by other European legislation and policies that address the environment as well as land and water uses (see fig. S2). Policy coherence requires complementary objectives and instrument mixes within environmental domains (*3*) while mainstreaming environmental objectives into other policy domains (*4*). These may enhance options for, or pose restrictions on, the implementation of the NRL. Key directives, some of which came into force decades ago, include the Habitats Directive (HD), Birds Directive (BD), Water Framework Directive (WFD), and Marine Strategy Framework Directive (MSFD). They share aims in safeguarding Europe's biodiversity but have not halted its decline. The Biodiversity Strategy for 2030 (BS) targets halting biodiversity loss, while the Forest Strategy (FS) and Common Fisheries Policy (CFP) address major land and sea uses. Last, the Common Agricultural Policy (CAP) has the largest budget and affects nearly 40% of the EU's terrestrial area, yet agriculture remains the lead driver of biodiversity loss (*5*). Together, these directives and policies cover a broad range of targets, sectors, and approaches and are representative for other instruments that will also interact with the NRL implementation (see the supplementary materials for details on our analysis of existing legislation, strategies, and policies).

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LESSONS LEARNED

In developing the NRL, the EU has learned from past experiences with European environmental legislation and policies and avoids several obstacles that have obstructed their implementation.

As a regulation, the NRL will come into force soon after it has been passed by the EU Parliament. This is an advantage in comparison with the HD/BD, WFD, and MSFD, which needed to be transposed into national law—a process that takes several years. Although the NRL will also need national implementation—for example, through National Restoration Plans—these could be passed by authorities without legislative procedures. This is a major advantage because speed is vital for tackling the biodiversity crisis and fulfilling the EU's international commitments (*6*).

The NRL sets ambitious quantitative targets in terms of both the areas to restore and the timeframe, with targets for 2030, 2040, and 2050 (see the figure). Experiences with previous legislation support this approach. The WFD and the MSFD defined deadlines for meeting the good status of all water bodies and seas (although in case of the WFD, allowing for an extension), but these firm deadlines made continuous restoration activities with intermediate targets more difficult. Timing,

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however, is also an issue for the NRL. The NRL's success hinges on prompt action and the provision of effective tools for achieving targets within short timeframes, recognizing the necessary time for nature to recover.

The NRL defines measurable and applicable indicators for restoration success. These include the area of restored habitats, which is easy to document and to control. Other more generic indicators, such as the Grassland Butterfly Index, are well established, thus facilitating implementation. A third group of indicators will require some standardization, such as indicators of forest restoration. Although no specific indicators are defined for marine ecosystems, criteria from the MSFD could be applied. Hence, the NRL can to a large degree capitalize on existing indicators, in sharp contrast to the WFD, MSFD, and HD/BD, all of which ignited extensive indicator development processes that delayed implementation.

Another advantage is the use of National Restoration Plans (NRPs), which has the potential to provide an appropriately tailored national framework for NRL implementation. Although all the above-listed directives are implemented at the national level, there have been particularly good experiences when actions take account of local contexts and needs, as seen with the River Basin Management Plans under the WFD. It is therefore imperative to ensure that the NRPs will be backed by robust implementation tools that adopt an adaptive cycle, whereby the commission can request member states to increase their ambition.

Key features of the EU Nature Restoration Law

Chapter I: General Provisions
- Defines the overall targets (continuous recovery of nature, fulfilment of climate change objectives and international regulations)
- Defines key terms: Favorable reference area (minimum area to ensure the long-term viability of a habitat type), good condition (characteristics that ensure favorable conservation status according to the HD or good environmental status according to the MSFD), sufficient quality and quantity of habitat (conditions required by a species for maintaining itself on a long-term basis)

Chapter II: Restoration Targets and Obligations
- For Natura2000 sites (Article 4): Good condition (30% by 2030, 60% by 2040, 90% by 2050); and favorable reference area (30% of the area needed to reach the goal for each habitat type by 2030, 60% by 2040, 100% by 2050); improved connectivity
- For habitats of species listed in Annexes II, IV, and V of Habitats Directive and of Birds Directive (Art. 4): Reach sufficient quality and quantity of habitats (no time frame given)
- Marine Ecosystems (Art. 5): Reaching good condition (30% by 2030, 60% by 2040, 90% by 2050) and favorable reference area (at least 30 % by 2030, 60 % by 2040 and 100 % by 2050)
- Urban Ecosystems (Art. 6): No loss in total national area of urban green spaces, increase in urban green spaces (3% of area of cities by 2040, 5% by 2050)
- Rivers, floodplains (Art. 7): Removal of barriers to longitudinal and lateral connectivity to achieve restoration targets and 25,000 km of free-flowing rivers; maintain and improve natural functions of floodplains
- Pollinator populations (Art. 8): Improve pollinator diversity, reverse decline of pollinator populations by 2030; achieve thereafter an increasing trend of pollinator populations
- Agricultural ecosystems (Art. 9): Increasing trend at national level in two of the three indicators: 'Grassland Butterfly Index', 'stock of organic carbon in cropland mineral soils', 'share of agricultural land with high-diversity landscape features'; targets for 'Common Farmland Bird Index': increase by 10% (2030), 20% (2040), and 30% (2050) for Member States with depleted farmland bird populations, and by 5% (2030), 10% (2040), and 15% (2050) for Member States with less depleted populations; restoration of organic soils in agricultural use constituting drained peatlands: 30% (by 2030), 40% (by 2040), 50% (by 2050)
- Forest ecosystems (Art. 10): Increasing trend at national level of the 'Common Forest Bird Index' and in 6 out of 7 additional indicators such as standing deadwood or forest connectivity

Chapter III: National Restoration Plans
- Obliges Member States to prepare restoration plans to implement the measures required for targets of Chapter II, and to quantify the area to be restored
- Member States have full flexibility to use or to discard funds from Common Agricultural Policy and Common Fisheries Policy for NRL implementation

Chapter IV: Monitoring
- Obliges Member States to monitor indicators for restoration targets; progress reports by the Commission

Chapter VI: Final provisions
- Application of the NRL will be evaluated by 2033, including possibly legislative proposals for amendments
- 'Emergency brake' allows Member States to halt NRL implementation in farmland, if agricultural production is at risk

ADVANCING IMPLEMENTATION

The NRL's aims reach well beyond the targets of existing legislation and policies (see the figure). In addition, the NRL offers great potential to boost the implementation of other European directives and policies. Whereas the WFD and MSFD focus on individual ecosystem types (surface water and marine ecosystems), the HD/BD take a broader approach, including a wide range of habitats, and the BS is even more comprehensive (*7*) because it addresses species, habitats, ecosystems, ecological processes, and public engagement. The NRL is broad but targets specific ecosystem types with tailor-made approaches (see the figure and fig. S2). It may therefore have impacts beyond the targeted ecosystems: For example, restoring agricultural ecosystems and forests has the potential to benefit rivers and lakes, and restoring peatlands can positively affect the landscape's water budget (*8*). Consequently, implementation of the NRL can substantially benefit the implementation of the HD/BD, WFD, and MSFD. This is most obvious for the HD/BD, which addresses a greatly overlapping list of habitats and species. The WFD and the MSFD can benefit from reduced pollution from agriculture and from the additional approaches the NRL provides. For example, the WFD does not explicitly address floodplains, although floodplains play an important role in the healthy functioning of rivers and their ecological quality (*9*). Also, the implementation of the BS will benefit from the restoration measures initiated by the NRL.

At first glance, the NRL may seem to be "conservative." It focuses mainly on the protection and restoration of habitats per se and of habitats for individual species. This is reminiscent of an approach from the 1980s, seemingly ignoring calls for more systemic, adaptive, and integrated approaches to managing nature. Article 8, with its focus on pollinators, is an exception to this. Ecosystem-based approaches, nature-based solutions, and cobenefits of restoration for other environmental and societal objectives are mentioned, but the text does not elaborate on their implementation. Despite this, the NRL holds considerable potential to operate at ecosystem levels, providing widespread societal benefits, particularly through the increased supply of ecosystem services (*10*). Enhancing landscape structure and rewetting peatlands can increase the resilience of agricultural ecosystems to droughts and pests, and restoring pollinator populations can have direct positive impacts on agricultural production. Similarly, reconnecting rivers with their floodplains can mitigate flood risks (*11*); increasing urban green spaces can benefit urban climate and people's health; increasing forest diversity can enhance resilience to extreme events; and restoring marine ecosystems can benefit recreation (*12*).

AVOIDING PITFALLS

A recurring problem with the implementation of European environmental legislation and policies is the gap between targets and effective implementation options. HD, BD, WFD, and MSFD have so far not achieved their aims, and neither has the BS (Annex 3). Reasons are manifold. Besides shortcomings in aims and approaches (see table S3, a to g), a common denominator is the lack of resources needed to implement them successfully, including funding, human resources, appropriate planning procedures, and administrative capacities for implementation. The passing of legislation and policies have not always been followed by the provision of appropriate resources and capacity-building for implementation and monitoring. The NRL encounters similar challenges because it is even more ambitious. Implementation at the national level must therefore assure a stringent procedure and a resilient funding structure, as suggested by the original Commission proposal. Although the targets are legally binding, the measures to achieve them will be voluntary actions by land and water owners and managers, who would need to accept co-responsibility and possess the capacity to respond. This requires not only financial investments but also supportive institutions for cooperation, peer-to-peer learning, business models supporting support land-use change, and societal acceptance to work with nature.

The required resources are not exclusively of public origin. After the NRL's approval, the EU and member states are tasked with mobilizing private financing of restoration, endorsing suitable business models that incorporate cost recovery (*13*). These may involve refined carbon credit trading, collaboration with insurance companies to mitigate flood or drought risks, or customized options for investing in nature. The European Investment Bank, and its enhanced capacity to offer advisory services alongside conventional financing, could assume a more prominent role in this regard.

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It will be of equal importance to acquire public funds for restoration of nature from other components of the EU budget—in particular, regional development and agriculture. So far, despite the instalment of relevant instruments, the CAP has not succeeded in achieving the aims of HD, BD, and WFD. The CAP is unlikely to contribute sufficiently to the NRL implementation if its support schemes are not modified to strengthen the ambition of measures, strictly enforce cross-compliance, and increase funding for focused measures. A specific clause is granting member states full flexibility in using or foregoing CAP or CFP funds for NRL implementation. Using these funds could potentially offer unprecedented, cost-efficient opportunities for both the NRL and the CAP and CPF. The CAP's agriculture-environment-climate measures, along with the somewhat less ambitious "Eco-schemes," could support habitat restoration and the recovery of pollinator populations. Implementing the NRL in farmlands is also vital for achieving various goals, including river-floodplain connectivity, river to coast-marine connectivity (through controlled floods), peatland targets (through alternative agricultural schemes such as paludiculture), and even urban restoration (by maintaining urban and peri-urban green and blue spaces). Simultaneously, addressing climate change in agriculture necessitates restoration measures such as landscape water storage, reduced livestock densities, and diminished nitrogen inputs.

The trilogue negotiations have introduced two further elements that substantially weaken the NRL. First, member states may permanently deprioritize restoration actions in areas used for other targets such as renewable energy infrastructure and military facilities. Second, the inclusion of an "emergency brake" enables member states to temporarily suspend NRL implementation in farmland, over their entire area, under exceptional circumstances that affect land availability for agricultural production. However, an evaluation of the NRL planned for 2033 could result in legislative proposals for amendments, including a better coherence with other legislation or policies.

Translating ambitions into actions still requires a close alignment with both existing and emerging European legislation and policies. Stability in the legislative developments is crucial, considering that nature restoration requires long-term perspectives. Provision of funding schemes will determine whether the NRL will address current pressures and drive much-needed transitions. Given the urgency of global crises, Europe cannot afford to delay; the opportunity to install and implement an ambitious law, and the opportunity to show global leadership, should not be missed.

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SUPPLEMENTARY MATERIALS

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Securing success for the EU Nature Restoration Law

The law would complement many others but challenges are still ahead

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The file includes:

Materials and Methods

For each of the following legislation, strategies and policies, we extracted key targets, strengths and weaknesses related to conservation and restoration of habitats and biodiversity, using the published text of the legislation, strategies and policies, and (if available) recent review papers:

- Habitats Directive (HD)
- Birds Directive (BD)
- Water-Framework Directive (WFD)
- Marine Strategy Framework Directive (MSFD)
- Biodiversity Strategy for 2030 (BS)
- Forest Strategy (FS)
- Common Agricultural Policy (CAP)
- Common Fisheries Policy (CFP)

We particularly looked into identifying co-benefits to be expected from the implementation of the Nature Restoration Law (NRL) legislation, in particular socio-economic benefits, and a reflection on lessons learnt that might help with the implementation of the NRL. We then rated the degree to which the targets have been met, using the most recent official reports on the implementation of the legislation, strategy or policy.

For each legislation, strategy or policy, we compiled implementation challenges (from literature and expert opinion) that have been responsible for the failure to achieve their targets, e.g., administrative obstacles, responsibilities, lack of funding, and contradicting targets. For each of these challenges, a qualitative / narrative check was performed on (i) if (and how) the challenge is addressed by the NRL and (ii) if similar challenges are to be expected for the implementation of the NRL (narratively and along a scoring of 0 to 2 with 0 = not relevant; 1 = partly relevant; and 2 = relevant). We checked if these obstacles are also applicable for the NRL implementation and formulated recommendations on how the NRL could overcome the particular challenges.

For FS, CFP, CAP, we performed a literature analysis on which of their targets and implementation steps have obstructed the achievement of biodiversity goals. Similar to the analysis of implementation obstacles, we performed a qualitative / narrative check on which of these obstructions is addressed at all by the NRL and if similar challenges are to be expected for the implementation of the NRL.

We then compiled the results of the previous four steps separately for each legislation, strategy or policy (tables S2e to S2g) and summarised in table S1.

Table S1. Overview of main references used to compile the Supplementary Text and tables S2 and S3.

Legislation, strategy, policy	References
Habitats Directive (*16*)	*1, 19, 20, 21, 23, 24, 25, 26, 27, 28, 29*
Birds Directive (*17*)	*23, 25, 27, 30, 1, 29*
Water Framework Directive (*33*)	*32, 33, 34, 35, 36, 37, 38*
Marine Strategy Framework Directive (*42*)	*41, 42, 43, 44, 45, 46, 47, 48, 49, 50, 51, 52, 53, 54, 55, 56, 57, 58*
Biodiversity Strategy for 2030 (*61*)	*61, 62, 63 64*
Forests Strategy (*66*)	*66, 67*
Common Fisheries Policy (*69*)	*69, 70, 71, 72, 73, 74*
Common Agriculture Policy (*76*)	*76, 77, 78, 18, 79, 80, 81, 82, 83*

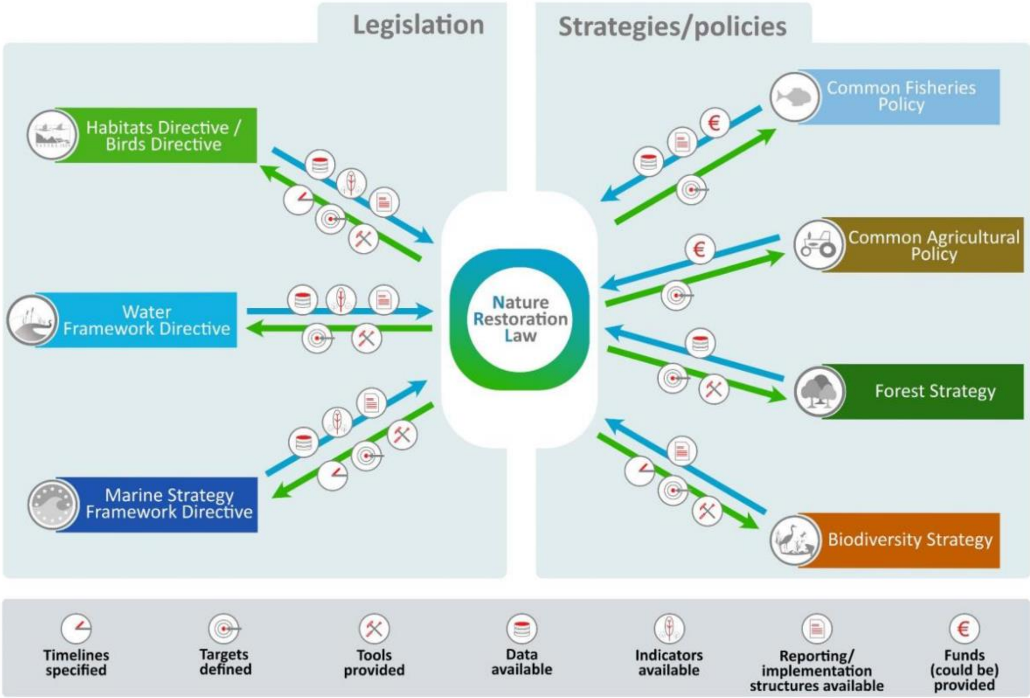

Fig. 2. Possible relations between the Nature Restoration Law (NRL) and other European environmental legislation, strategies and policies.

Supplementary Text

Habitats Directive (HD) and Birds Directive (BD)

Targets, strengths and weaknesses: The aim of both HD (*14*) and BD (*15*) was to install and mainstream protection measures of selected species and habitats across the EU. The target is to maintain or reach favourable conservation status for all wild bird species in the EU and their habitats (BD), non-bird animal and plant species as well as habitats (HD) that are 'of community interest', i.e., endangered, vulnerable or rare. For this purpose, Annexes list valuable and threatened species and habitats, and Member States established protected areas, forming the Natura2000 network. Especially the deterioration prohibition stated in Article 6(2) fuelled an increase in conservation activities and made it possible to judicially halt environmentally damaging practices (e.g. CJEU Cases C-117/00 (*16*), C143/02 (*17*)). The main strength of the directives is the creation of an EU wide network of protected areas through the Natura2000 network. In theory, this could ensure mainstreamed conservation efforts and wildlife that can flourish independent of national borders. In practice, however, even though enforcement of the laws by the European Court Justice is considered to be strict (*18*), the implementation of the Directives is slow and effects remain patchy. Merely 27% of species and 15% of habitats have reached favourable conservation status (*1*), while 81% of habitats and 63% of species are still in unfavourable status. Moreever, compared to 9% of habitats show improving trends, 36% show deterioration. Similarly, species assessments show an improving trend for 6% of species, but a deteriorating trend for 35% (*1*). Given the soft requirements for connectivity restoration in Natura 2000 sites, it could be argued that the failure to implement a proper, EU-wide network of protected sites is by design (19). Similarly, the focus on conservation instead of specific restoration with clear restoration norms, the inflexible structure of the Annexes with no adaptation or updating process in place, and the lack of a deadline to reach the targets are by-design weaknesses of the Directives (*20, 21)*. A lack of funding and lack of personnel qualified to create management plans, perform monitoring and work with stakeholders and national authorities are furthermore commonly identified as major challenges to the implementation and have led to vast differences in the implementation between member states (*25, 29*).

How the NRL can support the implementation: Large parts of the NRL are building upon the Habitats Directive, potentially improving restoration and conservation efforts through setting targets to establish effective and area-based restoration measures to cover all ecosystems in need of restoration and at least 20 % of the European Union's land and sea areas. The Birds and Habitats Directives have no specific quantitative, area-based or time-bound targets for Member States to carry out restoration measures, just to report on the measures taken, their purpose, location and expected time-frame when a habitat or species response would be expected. In this respect, the NRL can be viewed as a strong enabler by requiring more quantitative targets for restoration actions to achieve the HD and BD goals. The NRL also addresses the need to identify funding mechanisms to implement NRL-related restoration actions, granting Member States full flexibility in utilizing or foregoing CAP or CFP funds for NRL implementation.

Lessons learned that help with the NRL implementation: Designation of Natura 2000 sites and the implementation of restoration measures had been criticised as top-down and non-inclusive especially in the starting phase of the legislation's implementation (*29*). The preparation of management plans for Natura 2000 areas is not mandatory and was consequently not performed for all sites. Where management plans have been created, standards differ widely between Member States (*29*). This has led to unclear responsibilities, a lack of funding designated towards nature restoration and subsequently, a lack of qualified personnel to perform this task (*25*). Strategies to aid implementation of restoration measures were developed by the European Commission, such as the development of guidance for economic sectors or the formation of an expert group on the management of Natura2000 sites (*23, 83*). Still, the Directives' targets are far from being met and obstacles are still related to stakeholder engagement and organisation of the management plans. What is needed to improve the implementation of both BD and HD is not only more ambitious and quantified targets like the NRL proposes; but much more a functioning support system around the Directives, concerning funding, organisational support from the EU where local authorities can learn from experts, and training programmes in how to engage with different stakeholder groups. The NRL requires Member States to provide appropriate funding, yet it remains in the Member States' responsibility whether CAP or CFP funds are used for NRL implementation. While these provisions are still vague, they are an improvement compared to the current practice of HD/BD implementation.

Importantly, the NRL requires Member States to set up National Restoration Plans, which need to be approved by the EU commission. These plans will offer opportunities to learn from best practices, provide specific guidance to those Member States that need it and orchestrate cooperation and synergies between Member States' activities. While the Annexes of the Habitats and Birds Directives are rigid without much room for adaptation, the NRL proposes to update annexes after five years.

Even though interpretation and enforcement of the Directives by the European Court Justice is considered strict (*18),* critics claim the European Commission abandoned their post as 'Guardians of the Treaties' in recent years by drastically reducing the number of infringement procedures even where they would have been warranted (*22*). A support system for the implementation of the NRL can only help so far if failure to comply is not strictly prosecuted.

<u>Water Framework Directive (WFD)</u>

<u>Targets, strengths and weaknesses</u>: The WFD (*31*) aims at achieving the 'good ecological status' for all surface waters, which is defined by biodiversity-related criteria, e.g., abundance and diversity of aquatic organisms. The WFD was a step in widening the focus from point-source pollution to understanding aquatic ecosystem functioning and the wide range of ecosystem services these ecosystems provide (*34*). The WFD can serve as a prime example for establishing a Europe-wide monitoring system and has initialised a multitude of restoration actions, while its targets have only been achieved to a minor degree: by 2015, only 40% of the surface waters were in 'good status', i.e., their biodiversity remained depleted. Nevertheless, the WFD is the prime driver of aquatic ecosystem restoration in Europe and through the mandatory requirement for public participation, gave impetus to integrated basin planning and cross-sectoral dialogue (*38*). A large number of individual restoration measures have been performed under the WFD, which have led to local ecosystem status enhancement and widespread socioeconomic benefits (*84*).

<u>How the NRL can support the implementation</u>: Direct linkages of WFD and NRL are minor. The NRL targets rivers (Article 7), but it is focussed on selected impacts, in particular connectivity. While measures to enhance connectivity can support the probability of achieving good status, other stressors have a stronger impact on ecological status, in particular water pollution and habitat degradation (*85, 86*), the effects of which are becoming more acute due to the high water temperature peaks associated with global warming. However, the targets for urban systems

(Article 6), agricultural ecosystems (Article 9) and rewetting peatlands (Article 9) could be very beneficial also for water bodies, which are strongly affected by catchment land use. Therefore, a strong indirect effect on water bodies can be expected if the NRL targets are met.
If the NRL demands higher ecological status due to Natura 2000 commitments, WFD objectives are supported as well. However, should these habitats be deemed very common and widespread, the deadline becomes 2050 and the potential derogations proposed under Article 17 may apply.

Lessons learned that help with the NRL implementation: In terms of monitoring, the WFD is a prime example on how to implement a Europe-wide monitoring system. Lessons learned for the NRL include the importance of intercalibration of national assessment systems and the importance to centrally collect original monitoring data. In terms of measure implementation, the WFD experience underlines that restoration is not successful if the responsible authorities have no options to mitigate stressors originating from adjacent land. It also underlines that ambitious and binding targets help with initialising restoration measures, but are not necessarily met if they are not accompanied by sufficient restoration tools, funding and political will. Article 11 leaves open, if CAP-funds, including the European Agricultural Fund for Rural Development (EAFRD), are used to support freshwater restoration (*39*).
The most recent reviews of River Basin Management Plans show that it takes time to embed the integrated basin approach but there are signs that more challenging issues are now being addressed (e.g., e-flows). The WFD has a holistic view on aquatic ecosystems and thus requires that all relevant stressors / pressures are addressed, which is often not realistic. Against this background, the more limited and targeted NRL goals seem to be more feasible and achievable; whilst the adaptive management approach of the NRL plans allows for learning and improvement. Recital 49 offers an important acknowledgement of the synergistic relationship between agricultural land use and floodplain or riparian restoration when such restoration can 'benefit of the long-term functioning and productivity of the agricultural ecosystems'. To see that such synergy is achieved, it will be important to set targets on the level of Member States and provide conflict resolution mechanisms where there are competing claims on land and water uses. The introduction of an "emergency brake" (article 22a), allowing member states to temporarily halt the implementation of Article 9 under extreme (but poorly defined) conditions, could undermine progress to restoration (and therefore to WFD objectives).

Marine Strategy Framework Directive (MSFD)

Targets, strengths and weaknesses: The MSFD (*40*) aims at achieving the 'good environmental status' for all regional seas (i.e., Baltic Sea, North-East Atlantic Ocean, Mediterranean Sea and Black Sea), which is defined by eleven descriptors, including biodiversity, non-indigenous species, commercial species, food webs, eutrophication, seafloor integrity, hydrography, contaminants in the environment, contaminants in seafood, litter and energy/noise. The MSFD can serve as a prime example for establishing a European ecosystem-based management (EBM) approach, in which humans and their activities are considered part of the marine system, and they should be undertaken in a sustainable way, to achieve good environmental status. If this status is not achieved by 2020 (or 2026), a program of measures, including restoration actions, must be implemented by each Member State. A certain number of individual restoration measures have been performed under the MSFD, as well as other measures of protection and conservation of the seas (*44, 48*).

How the NRL can support the implementation: Linkages of MSFD and NRL include some marine habitats, such as seagrasses, sediment bottoms, rocky habitats and dunes, but also habitats of iconic species, such as marine mammals, sharks or seabirds. The NRL targets coastal (Article 4) and marine ecosystems (Article 5), focussing on selected impacts, and related to Marine Protected Areas and Habitats Directive. Since some of the targets are included in the MSFD programmes of measures set up by member states, the NRL can reinforce the achievement of MSFD goals.

Lessons learned that help with the NRL implementation: In terms of monitoring, the MSFD is an example on how to collaborate within European Regional Seas to jointly monitor them (*45, 56*) However, some weaknesses and threats still remain (*50*). Lessons learned for the NRL include: (i) harmonize the target setting, at least regionally, improving the evaluation and comparability of the updated monitoring programmes; (ii) ensure consistency in the monitoring programmes among countries, for indicators and criteria; (iii) assign a detailed list of species and habitats to each monitoring programme; (iv) consider natural and seasonal variability for monitoring and restoration (*55*). In terms of measure implementation, the MSFD experience underlines that restoration is not successful if the competent authorities have no options to mitigate stressors originating from adjacent countries or Areas Beyond National Jurisdiction. The MSFD has a holistic view on marine ecosystems (under the EBM approach) and thus requires that all relevant human activities and pressures, as well as the ecosystem components, are addressed. Against this background, the more limited and targeted NRL goals seem to be more feasible and achievable, especially those related to the protected areas, if these targets are enforced.

Biodiversity Strategy for 2030 (BS)

Targets, strengths and weaknesses: The Biodiversity Strategy for 2030 (BS) of the EU (*59*), a non-binding strategy, aims at halting biodiversity loss and preserving ecosystem functioning by protecting at least 30% of the EU´s land, while restoring ecosystems . For this, it targets creating green and blue infrastructure, i.e., wildlife corridors and ecological networks, improving connectivity between protected areas. It also aims at reducing anthropogenic stress through more sustainable agriculture, fisheries, forest management, and urbanisation and halting the establishment of invasive species. The BS offers a comprehensive approach addressing multiple dimensions of biodiversity protection, including the species level, habitat protection, entire ecosystems and ecological processes (*60*). It also acknowledges the importance of public engagement and awareness and promises regular monitoring and reporting with a long-term vision (*63*). This makes the BS a key example for an interdisciplinary approach towards biodiversity protection. However, meeting the ambitious targets is obstructed by several challenges, including that the targets are not legally binding, so that only 15% of habitats and 27% of species listed in the HD and 47% listed in the BD are not under risk of extinction (*1*), despite aiming to improve the status of 100% of the habitats and 50% of the species. Reasons are limited funding and resources and varying degrees of implementation among EU member states and conflicts with short term economic objectives. Also, policy coherence with other legislation is challenging, and unharmonized monitoring methods across the EU further halt rapid progress. Lastly, slow adaptation to global and climate change may further retard successful protection of biodiversity (*61, 62, 64*). Nonetheless, the BS offers a foundation for pan-European ecosystem management and joint efforts for biodiversity protection.

How the NRL can support the implementation: The NRL partly builds on the BS, both targeting ecosystem restoration at different spatial scales of a multitude of ecosystems (NRL Articles 4-10). The NRL can support the implementation of the BS through by improving enforceability of joint targets by providing binding targets, approaches and timelines, quantifying these, and further supporting public involvement and introducing novel funding opportunities. Hence, harmonising different environmental legislation, strategies and policies will be crucial to successfully reach the environmental targets.

Lessons learned that help with the NRL implementation: The BS is a clear commitment of the European Union to stop biodiversity loss while enabling sustainable development. It is a pioneer interdisciplinary strategy with clear ambitious targets regarding several environmental legislation and policies (*60*) and is in line with international agreements such as the UN Sustainable Development goals (*88*) and the Convention on Biological Diversity (*89*). However, its success greatly relies on the Member State's willingness to implement the strategy and to provide sufficient funding, planning, governance and enforcement, and to mitigate competing interests (*64*). These topics will also be crucial for the NRL implementation, but as a regulation, the NRL will provide a much stronger lever. Particularly the lack of enforcement of several environmental policies, such as the CFP and the CAP (*90, 82*) highlights the need for binding targets that are now provided by the NRL. These must be supported by sufficient funding, planning and a long-term economic perspective.

Forest Strategy to 2030 (FS)

Targets, strengths and weaknesses: As an additional initiative of the European Green Deal that is building on the EU Biodiversity Strategy for 2030, the Commission adopted a new EU Forest Strategy to 2030 (*65*). The FS outlines a vision and concrete actions for increasing the quantity and quality of forests in the EU and strengthening their protection, restoration and resilience. It aims to facilitate adapting Europe's forests to new conditions, weather extremes and high uncertainty brought about by climate change (*65*). The main objectives of the FS are effective afforestation, forest preservation and restoration in Europe, to help to increase the absorption of CO_2, reduce the incidence and extent of forest fires, and promote the sustainability of forest-based bioeconomy, while accounting for biodiversity. It also aims to strictly and effectively protect all primary and old-growth forests in the EU. Most importantly, the FS demands that clear cutting practices in the EU countries should be used only in duly justified cases (*65*). It also requires adequate distribution of funding for landowners dedicated to restoring their forests through carbon credits compensations and payments for ecosystem services. Implementing the NRL alongside the EU Forest Strategy for 2030 presents opportunities for synergistic ecosystem restoration, enhanced forest governance, climate change mitigation, and green job creation. However, challenges may arise due to conflicting objectives, competing land use priorities, stakeholder engagement, and monitoring requirements.

In practical terms, the FS defines 'thresholds and ranges' that establish the boundaries of sustainability. The FS includes a pan-European indicator set to monitor progress towards the Strategy's objectives. Two thirds of the objectives and commitments identified in the FS can be monitored at least partially - and in some cases weakly - by those indicators, whereas new indicators need to be developed for the remaining third (*66*).

How the NRL can support the implementation: The Nature Restoration Law establishes a legal framework for ecosystem restoration, including forests, aligning with the FS goals. It can prioritise biodiversity conservation, supporting diverse forest ecosystems and sustainable forest management. The law can foster coordination among different groups of stakeholders, as well as enhance monitoring and evaluation of restoration projects, by allocating resources and incentives. Public awareness and engagement can be further promoted, while research and innovation can contribute to effective forest restoration and management. Additionally, the NRL can address climate change challenges, emphasising climate-resilient approaches to protect forests from climate impacts.
Lessons learned that help with the NRL implementation: Implementing the NRL can benefit from setting well-defined goals and guidelines for ecosystem restoration, ensuring clarity and consistency in actions, by adopting robust monitoring and evaluation processes to measure the impact of restoration activities and make informed decisions. It can learn from employing an adaptive management approach, enabling adjustments in response to emerging challenges and new information. Furthermore, the FS underlines that the NRL should ensure sufficient resources and incentives to motivate stakeholders to participate actively in restoration activities as well as to focus on long-term objectives for ecosystem restoration, recognizing that positive outcomes may take time to materialise (*67*). There is also a need for long-lasting support and political will to achieve restoration goals.

Common Fisheries Policy (CFP)

Targets, strengths and weaknesses: The CFP (*68*) aims at having all fish stocks within safe limits in Europe, defined by the Maximum Sustainable Yield, which is the largest catch that can be taken from a species' stock without reducing the size of the population (*72*). The CFP recognises that fishing activities affect marine ecosystems through seabed disturbance, bycatch of key species and effects on marine food webs (*70*). However, at the same time, fisheries are affected by climate change, river nutrient discharges and other human pressures (e.g., agriculture, industry, shipping, etc.), introducing diverse pollutants at sea, such as litter, traditional and emerging contaminants (*69, 73*). The CFP implementation needs to be strengthened, especially regarding: (i) the landing obligation; (ii) the contribution to the implementation of environmental legislation and the related governance system; (iii) the improvement of the knowledge base and the strengthening of the ecosystem-based approach, keeping in mind both socio-economic and environmental objectives; (iv) the allocation of quotas at national level and the transparency of the process; (v) the sector's energy transition; and (vi) the development of biophysical and socioeconomic indicators to be used in fisheries management and conservation measures (*72*).
How the NRL can support the implementation: Linkages of CFP and NRL include marine habitats, such as seagrasses, sediment bottoms, rocky habitats and dunes, but also habitats of iconic species, such as marine mammals, sharks or seabirds (some of them affected by bycatch). The NRL targets coastal (Article 4) and marine ecosystems (Article 5), focussing on selected impacts, and related to Marine Protected Areas and Habitats Directive. Since some of the targets are related to the CFP, the NRL can reinforce the achievement of CFP objectives. Vice versa, Member States can use CPF funds to fund restoration activities under the NRL.
Lessons learned that help with the NRL implementation: In terms of monitoring, the CFP is an example of how long-term and detailed data recording for biophysical variables that characterise

many stocks, can be obtained by collaborating within large organisations (e.g., Copernicus, ICES, CIESM). Lessons learned for the NRL include: (i) the harmonisation of methods across stocks and seas will improve the evaluation and comparability for the recovery of restored habitats; (ii) consistency in the use of indicators needs to be ensures; (iii) a detailed list of species and habitats in the NRL is useful for implementation; (iv) the interaction among species needs to be considered (*72, 74*).

Common Agricultural Policy (CAP)

Targets, strengths and weaknesses: Following Article 39 of the Treaty on the Functioning of the EU (*75*), the CAP aims to support productivity, farm incomes and stable markets. Recently, the CAP also included operational targets of environmental sustainability by promoting sustainable farming practices, as reflected in the CAP objectives for the funding period 2023-2027 (Article 6). The environmental objectives specifically seek to promote sustainable land management practices, enhance biodiversity protection on agricultural landscapes, improve water resource management, and introduce climate change mitigation measures. This should be achieved by three key elements in the updated 'green architecture' of the CAP, particularly 'enhanced conditionality' (formerly Cross Compliance and Greening) defining basic standards to which farmers have to adhere, newly established Eco-Schemes as voluntary measures in Pillar 1, and Agri-Environmental and Climate Measures (AECM) in the CAP's rural development plan (Pillar 2) – with the aim to incentivize farmers to improve agricultural practices (*91, 92*). While the CAP's targets of (short term) food security and financial support have been met (*92*), targets of environmental sustainability have been only achieved to a minor degree (*79, 81*). The environmental targets set too low requirements for environmental protection; and the introduced eco-schemes, in many Member States, may not be ambitious enough to generate change in management and restoration. It has been shown that the AECMs have the potential to support the specific implementation of the BD and HD, depending on the regional design and funding of targeted measures (*94*). However, in total, the AECM did not receive sufficient funding budgets; the investments are not balanced between uniform payments; and many of the measures chosen by member states are not effective enough to promote environmental protection and biodiversity-friendly farming (*77, 83*). The CAP could still potentially serve as a key instrument to achieve the future NRL-targets in terms of biodiversity protection and habitat restoration, and has some instruments to promote pan-European improvements for land use management challenges. However, the high share of CAP subsidies that are independent of business models and farm practices (especially Direct Payments) sets a challenge to transform agriculture toward a more environmentally friendly orientation (*59, 78, 86*). Furthermore, most farmers are exempt from basic good practices ('Good Agricultural Ecological Condition', GAEC) as required by enhanced conditionality. Recent derogations from some of these standards, cancelling the Ecological Focus Areas (EFA) in the year 2022, and cancelling Good agricultural and environmental conditions (GAECs) 7 and 8 in 2023 in response to the Russian war against Ukraine, demonstrate high sensitivity of the CAP to deregulation pressures, and highlighting the importance of an external legislative framework.

How the NRL can support the implementation: Both CAP and the NRL target sustainable, resilient and biodiverse agriculture (NRL Articles 8 and 9), thus addressing the largest share of land use in Europe (*93*). The NRL can support the implementation of the CAP's environmental targets by setting clear, binding, and more ambitious targets for 2030, 2040 and 2050, by e.g., promoting diverse landscapes, securing (and increasing) the stock of organic carbon in soils and restoring pollinator populations. These are practically achievable through existing CAP instruments, as they merely require an update of Eco-Scheme lists, improving the alignment of some existing instruments (e.g. payments for Areas of Natural Constraints) and possibly conducting some changes of funding priorities in the CAP. Other agricultural measures target extensification (or in rare cases abandonment) of farm practices to allow habitat restoration and rewetting of peatlands. In cases where land is taken out of production, they go beyond the CAP and possibly complementing it. However, lacking a clear financial plan, a substantial change based on the NRL alone is unlikely and will need more ambitious targets in the next update of the CAP. Many approaches of the current CAP contradict the NRL targets, in particular the main focus on direct payments. If the NRL, in particular Articles 8 and 9, should be successful, the next CAP needs to have a much stronger focus on results-based agri-environmental measures that particularly support the NRL's aims, for example funding schemes for rewetting peatlands. It is of crucial importance that measures to implement the NRL can also capitalise on CAP funds, but this option was unfortunately delegated to Member States' responsibilities in the NRL. Given the demonstrated low ambition of most Member States with regards to farmland biodiversity, this is a risk. For land that is taken out of agricultural production, additional funding beyond the CAP needs to be more clearly defined.

<u>Lessons learned that help with the NRL implementation</u>: The CAP is an example for EU-wide policy acting over decades with a long-term perspective (*92*). It fosters promising approaches of incentivizing agricultural transition to more sustainable land management with approaches such as the Eco-Schemes, rural development plans or the associated Farm to Fork Strategy. In terms of environmental objectives, however, the implementation is not very successful, partly due to conflicts of interest with short term economic goals and the funding of business-as-usual farming models, diluting ambitious environmental protection measures (*81*). Also, the funding of environmental transition is not sufficient (*77*), highlighting the importance of securing funding for NRL implementation. The more ambitious and binding targets for environmentally friendly agricultural management, as proposed by the NRL (particularly Articles 8 and 9), and the introduction of new and scientifically well-established indicators (butterflies, carbon stocks, landscape features) can yield more promising results and are hence important to retain and even expand. Also, by highlighting the economic sense of investing in environmental protection with larger returns than the initial costs, the NRL may help delivering promising results also for farmlands.

Table S2. Biodiversity-related targets and degree of achievement obtained by the individual legislation, strategies and policies.

Legislation	Ecosystem types addressed	Organism groups addressed	Biodiversity-related targets	Degree of target achievement
Habitats Directive	Terrestrial, freshwater and marine	Habitats (EUNIS)	Favourable conservation status of defined habitats and species	15% of habitats, 27% of species in good conservation status
Birds Directive	Terrestrial, freshwater and marine	Birds	Favourable conservation status of all bird species and of habitats of certain bird species	47% of bird species in good conservation status
Water Framework Directive	Rivers, lakes, transitional and coastal waters	Phytoplankton, phytobenthos, macrophytes, invertebrates, fish	Good ecological status / potential of all surface waters by 2027	By 2015, 40% of surface waters met targets, 60% failed
Marine Strategy Framework Directive	Marine waters from coast to EEZ boundary	All marine, from plankton to mammals	Good environmental status in all seas by 2026	By 2020, loss of marine biodiversity has not been halted; marine ecosystem condition is generally not 'good'; signs of recovery for some species and areas
Biodiversity Strategy for 2030	Terrestrial (soil, agriculture, urban, forests), freshwater, marine	All from Habitats from plants to invertebrates and vertebrates	Coherent trans-European nature network; Protection of at least 30% of land and sea by 2030	By 2020: 15% of habitats and 27% of species listed in Habitats Directive and 47% of species listed in Birds directive not under risk of extinction

Legislation	Ecosystem types addressed	Organism groups addressed	Biodiversity-related targets	Degree of target achievement
Forests Strategy	Forests	All components of forest biodiversity	Biodiversity friendly afforestation and reforestation; Closer-to-nature-forestry practices; forest reproductive material	Recently adopted strategy, not yet quantified
Common Fisheries Policy	Marine (and freshwater)	Fish, shellfish	All fish stocks within safe limits	In 2018, from 188 stocks 22% in good status (using 2 criteria), and further 34% (using one criterion)
Common Agricultural Policy	Agricultural ecosystems, soil, indirectly other ecosystems (e.g., aquatic ecosystems) by reduction of diffuse pollution	Farmland birds, pollinating insects, soil organisms, aquatic organisms, High-Nature Value and Natura2000 habitats and species	Maintain and enhance biodiversity to support agricultural production - particularly by Ecological Focus Areas; Climate Change mitigation by Agri-Environment-Climate-Measures	Few positive outcomes - local success depending on farmers engagement; CAP has mainly objectives, rather than quantifiable targets (except for share of Ecological Focus areas or share of organic farming, which slightly increased)

Table S3a. Implementation challenges for meeting the biodiversity-related targets of the **Habitats Directive (HD) and the Birds Directive (BD)** and relation to NRL text. Column 'Is a similar challenge expected…': 0 = not relevant; 1 = partly relevant; and 2 = relevant.

Challenges of implementation	(How) does the NRL address this challenge?		Is a similar challenge expected for the implementation of the NRL?	Recommendations for NRL implementation to overcome the challenge
Policy				
Lacking or weak political commitment and support for biodiversity policy	When writing National Restoration Plans, synergies with other sectors/measures can be identified	1	Less relevant, as the NRL targets are binding, likely leading to stronger policy support	
Finance, economy and capacity				
Funding • Insufficient funding designated for implementing the Directives • Unsuitable financing mechanisms resulting in a lack of management plans for implementation	• Financial incentives for member states • Integration of NRL-objectives into EU-Funding programmes	2	Appropriate and flexible funding will also be decisive for the implementation of National Restoration Plans under the NRL. There is an investment gap of over 40 billion euros for NRL (*94*).	Establish appropriate funding mechanisms, also using private funding and funding provided by the CAP
Institutional capacity • Insufficient personnel capacities • Lack of knowledge and/or skills of management staff	Not addressed	2	Well-trained personnel will be even more relevant for the NRL implementation, as the targets are more ambitious and to be implemented in a shorter timespan.	Enhance and train personnel in the relevant authorities.

Challenges of implementation	(How) does the NRL address this challenge?		Is a similar challenge expected for the implementation of the NRL?	Recommendations for NRL implementation to overcome the challenge
Implementation barriers				
Communication barriers between authorities and stakeholders	Open and inclusive preparation of National Restoration Plans	2	Communication will be even more relevant for the NRL implementation, as the targets are more ambitious and to be implemented in a shorter timespan	Training programmes in how to engage with different stakeholder groups
Inconsistencies / intransparency • between Member States in designation and management of protected sites (Natura 2000) • within Member States, if implementation is delegated to federal states or regions • management plans not consistent in scope and content, as these are just optional instruments • High complexity of the Directives and corresponding guidelines result in a lack of transparency and understanding by implementing authorities	Obligatory National Restoration Plans	2	The proposed content for the Restoration Plans is fairly concrete, however, inconsistencies between Member States will remain, as it is up to the Member States how the targets will be achieved	Clear guidelines for National Restoration Plans.

Challenges of implementation	(How) does the NRL address this challenge?		Is a similar challenge expected for the implementation of the NRL?	Recommendations for NRL implementation to overcome the challenge
Conflicts in targets with other sectors or measures, such as Climate Change mitigation	• NRL targets more closely linked to Climate Change mitigation • Integrated planning towards areas where both can be achieved simultaneously	2	• Conflict with land users will inevitably occur, but conflicts with Climate Change mitigation measures will be less relevant • Despite integrated planning approaches, it is still likely that the space will not suffice the needs for both biodiversity protection and the constantly growing economy	• Develop guidance for economic sectors; establish expert groups • Transition of economy needed to live within 'planetary boundaries'
Data and monitoring				
Lack of data for the assessment of sites	Obligatory monitoring programmes under the NRL	1	NRL indicators are simpler as compared to the BD/HD indicators; but monitoring and reporting requirements are set too sparsely in most cases (every 6 years)	Harmonise indicators between countries and regions to enable comparability of results; ensure yearly monitoring
Rigid and outdated species lists in the Annexes with no adaptation practice	Update of the Annex lists after five years	2	All restoration targets are based on the condition of Annex species/habitats, so careful and adaptive selection is crucial	Ensure ecologically sound selection and adaptation of targeted habitats
Soft requirements for connectivity in Natura2000 sites	No quantification for improved connectivity is included, but measures outside Natura2000 sites can improve connectivity	1	Connectivity will remain as a challenge, but NRL has more options to improve connectivity	Formulate clear targets and indicators of connectivity

Table S3b. Implementation challenges for meeting the biodiversity-related targets of the **Water Framework Directive (WFD)** and relation to NRL text. Column 'Is a similar challenge expected…': 0 = not relevant; 1 = partly relevant; and 2 = relevant.

Challenges of implementation	(How) does the NRL address this challenge?		Is a similar challenge expected for the implementation of the NRL?	Recommendations for NRL implementation to overcome the challenge
Policy				
Ecological and political timescales • Implementation and success of restoration measures requires long time periods • Insufficient knowledge on how fast biota will respond to restoration • Long time needed to implement measures that require land use change • Time lags due to internal nutrient loading and low recolonisation potential expected	Not addressed	2	Long time periods for the achievement of the goals, independently from implemented restoration actions, are expected (e.g., due to required recolonisation)	Success should not only be measured by the achievement of status, but also by the degree of implemented restoration actions
Finance, economy and capacity				
Financing • The use of WFD economic instruments is partial and not well implemented in many Member States	Not addressed	2	There is an investment gap of over 40 billion euros for NRL (*78*)	Use a combination of financial strategies to fund NRL including public incentives, cost-recovery measures and private investments

Challenges of implementation	(How) does the NRL address this challenge?		Is a similar challenge expected for the implementation of the NRL?	Recommendations for NRL implementation to overcome the challenge
Lack of governance tools to address the main stressors affecting ecological status • No direct influence of implementing organisation on riparian and catchment land use • lack of capacity to direct voluntary measures	Indirectly addressed by the demand for the removal of lateral barriers and floodplain restoration	2	• Impacts of surrounding land use on status of Natura2000 sites is likely, independently from implemented restoration actions • Difficulty in spatially targeting measures associated with other legislation, strategies and policies	Strong policy coherence mechanisms in developing but crucially implementing and reviewing National Restoration Plans
Implementation barriers				
Slow implementation of measures due to conflict with over land and water uses (e.g., hydropower) and lack of skills/capacity in implementing Natural Water Retention Measures or paludiculture	Not addressed	2	Lack of political will, lack of capacity building for advisors, resistance from land managers can be expected	Better policy coherence with CAP measures
Slow and insufficient progress in ecological status improvement	Indirectly addressed by the demands for additional restoration actions (removal of longitudinal and lateral barriers)	2	As targets are ambitious and not only influenced by the implementation of restoration measures, but also by large-scale stressors and recolonisation obstacles, slow progress in habitat condition improvement is expected, too	Success should not only be measured by the achievement of status, but also by the degree of implemented restoration actions

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Challenges of implementation	(How) does the NRL address this challenge?		Is a similar challenge expected for the implementation of the NRL?	Recommendations for NRL implementation to overcome the challenge
Derogations (Article 7) allows less stringent objectives or timescales	Not addressed	0	There is no mechanism for derogation but the objectives are targeted, so Member States can choose where to target their efforts. Conflicts between private and public good delivery is likely to impede implementation.	Illustrate where and how working with nature can protect private enterprises and society from risks or increase productivity. Conflict resolution processes and tools should be developed.
Bridging ecology and management in River Basin Management Plans • Deriving management decisions from ecological data are difficult in case of complex multi-stressor situations • Some assessment metrics are not related to specific pressures (general degradation metrics) and are difficult to apply to plan restoration measures	Not addressed	1	It is only vaguely outlined how management / restoration decisions are to be based on monitoring results	Monitoring results, amended by data on implemented restoration actions, should be centrally collected and stored to allow for an assessment of restoration efficiency

Challenges of implementation	(How) does the NRL address this challenge?		Is a similar challenge expected for the implementation of the NRL?	Recommendations for NRL implementation to overcome the challenge
Emerging stressors • Assessment metrics often focussed on 'traditional stressors' (organic pollution, eutrophication) • Lack of attention for emerging stressors (climate change, water scarcity, alien species) included	Not addressed	2	Effects of emerging stressors are not addressed explicitly, only as a reason for the non-fulfilment of targets according to Article 4. Overall, the NRL is targeting both, biodiversity decline and climate change.	Consider ecosystem development ('prestoration') rather than restoration targets
Ecological status response to restoration • Response of biota to restoration measures in complex multi-stressor situations poorly predictable • Lack of data and experience on spatial and temporal scales required for restoration	Not addressed	1	• NRL allows for considerable degree of freedom on how the targets will be achieved • No central collation of data and experiences on restoration activities and their successes planned	Monitoring results, amended by data on implemented restoration actions, should be centrally collected and stored to allow for an assessment of restoration efficiency
Data and monitoring				
Intercalibration • Differences in national assessment systems, due to biomonitoring traditions • Effort and time required for intercalibration has been more than expected	Not addressed	1	• Most monitoring criteria according to Art. 17 (e.g., habitat area) do not need to be intercalibrated • Condition of habitat types is assessed differently by member states	Consider intercalibration of habitat condition between member states to avoid differently stringent targets

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Challenges of implementation	(How) does the NRL address this challenge?		Is a similar challenge expected for the implementation of the NRL?	Recommendations for NRL implementation to overcome the challenge
Monitoring data • Comparability of original data between countries is limited due to different sampling methods, taxonomic resolution and density of sampling sites • Original data are not centrally stored	Not addressed	1	There are no plans for central collection of original monitoring data	Europe-wide collection of original monitoring data should be planned from the very beginning of the implementation
Surveillance monitoring and operational monitoring • Very few surveillance monitoring sites in many member states, which limits European State-of-Environment overviews, as well as the detection of emerging stressors and long-term trends • No Europe-wide data base on surveillance monitoring	Not addressed	1	Monitoring is focussed on the improvement of conditions, not on long-term changes caused by other drivers, e.g., climate change	A fraction of the monitoring sites should be placed in habitats that already achieve good condition, to allow for unbiased analysis of long-term trends

Challenges of implementation	(How) does the NRL address this challenge?		Is a similar challenge expected for the implementation of the NRL?	Recommendations for NRL implementation to overcome the challenge
Monitoring requirements of WFD and other European legislation • Definitions of objectives and requirements of WFD and other directives are not always consistent • Potential synergies of monitoring systems resulting from different directives not fully exploited	Not addressed	1	The NRL is capitalising on the condition assessment of the HD and on the ecosystem condition assessment in the accounting framework, but ignores assessment results of other directives	In case of freshwater habitats, good condition (according to Article 3) could be defined as 'good ecological status'

Table S3c. Implementation challenges for meeting the biodiversity-related targets of the **Marine Strategy Framework Directive (MSFD)** and relation to NRL text. Column 'Is a similar challenge expected…': 0 = not relevant; 1 = partly relevant; and 2 = relevant.

Challenges of implementation	(How) does the NRL address this challenge?		Is a similar challenge expected for the implementation of the NRL?	Recommendations for NRL implementation to overcome the challenge
Policy				
Ecological and political timescales • Implementation and success of restoration measures require long time periods • Insufficient knowledge on how fast species, habitats and ecosystem services will respond to restoration • Long time needed to implement measures that require human activities change	Not addressed	2	Long time periods for the achievement of the goals, independently from implemented restoration actions, are expected (e.g., due to required recolonisation)	Success should not only be measured by the achievement of status, but also by the degree of implemented restoration actions
Implementation barriers				
Lack of governance tools to address the main pressures affecting environmental status No direct influence of organisation responsible for sea uses	Indirectly addressed by involving various ecosystem types that influence each other	2	Impacts of surrounding uses on status in protected areas is likely, independently from implemented restoration actions	Strong cooperation with land and sea users and the relevant legislation, strategies and policies, in particular with the CAP

Challenges of implementation	(How) does the NRL address this challenge?		Is a similar challenge expected for the implementation of the NRL?	Recommendations for NRL implementation to overcome the challenge
Slow and insufficient progress in environmental status improvement	Indirectly addressed by the demands for additional restoration actions	2	As targets are ambitious and not only influenced by the implementation of restoration measures, but also by large-scale pressures and recolonisation obstacles, slow progress in habitat condition improvement is expected also for NRL-related measures	Success should not only be measured by the achievement of status, but also by the degree of implemented restoration actions
Bridging ecology and management • Deriving management decisions from ecological data are difficult in case of complex multi-pressures situations • Some indicators and criteria are not related to specific pressures (general degradation) and are difficult to apply to plan restoration measures	Not addressed	1	It is only vaguely outlined how management / restoration decisions are to be based on monitoring results	Monitoring results, amended by data on implemented restoration actions, should be centrally collected and stored to allow for an assessment of restoration efficiency

Challenges of implementation	(How) does the NRL address this challenge?		Is a similar challenge expected for the implementation of the NRL?	Recommendations for NRL implementation to overcome the challenge
Emerging pressures • Assessment criteria often focussed on 'traditional pressures' (eutrophication) • No (or still little tested) criteria for the effects of emerging pressures (climate change, non-indigenous species, noise, litter) included	Not addressed	2	Effects of emerging stressors are not addressed explicitly, only as a reason for the non-fulfilment of targets according to Article 4 Overall, the NRL is targeting both, biodiversity decline and climate change	Consider ecosystem development ('prestoration') rather than restoration targets
Environmental status response to restoration • Response of species, habitats and ecosystem services to restoration measures in complex multi-pressure situations poorly predictable • Lack of data and experience on spatial and temporal scales required for restoration	Not addressed	1	• NRL allows for considerable degree of freedom on how the targets will be achieved • No central collation of data and experiences on restoration activities and their successes planned	Monitoring results, amended by data on implemented restoration actions, should be centrally collected and stored to allow for an assessment of restoration efficiency
Data and Monitoring				

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Challenges of implementation	(How) does the NRL address this challenge?		Is a similar challenge expected for the implementation of the NRL?	Recommendations for NRL implementation to overcome the challenge
Monitoring data • Comparability of original data between countries is limited due to different sampling methods, taxonomic resolution, density of sampling sites, and spatial and temporal cover • Original data are not centrally stored (raw data used in the assessments are not available)	Not addressed	2	There are no plans for central collection of original monitoring data	Europe-wide collection of original monitoring data should be planned from the very beginning of the implementation
Coordinated monitoring among Member States sharing the same regional sea • Very few coordinated monitoring surveys, to ensure results comparability and reduce costs of monitoring • No Europe-wide database on monitoring	Not addressed	2	Monitoring is organised in already existing networks, but needing adaptation to include more species and descriptors, as well as ensuring full spatial cover and long-term monitoring, to understand changes caused by climate change	Ensure that adequate monitoring networks cover spatio-temporal changes, to allow for unbiased analysis of long-term trends

Challenges of implementation	(How) does the NRL address this challenge?		Is a similar challenge expected for the implementation of the NRL?	Recommendations for NRL implementation to overcome the challenge
Monitoring requirements of MSFD and other European legislation • Definitions of objectives and requirements of MSFD and other directives are not always consistent (e.g. Habitats and Birds Directives) • Potential synergies of monitoring systems resulting from different directives not fully exploited	Not addressed	2	The NRL is capitalising on the condition assessment of the HD and on the ecosystem condition assessment in the accounting framework, but ignores assessment results of other directives.	In case of marine systems, good condition (according to Article 3) could be defined as 'good environmental status'.

Table S3d. Implementation challenges for meeting the biodiversity-related targets of the **Biodiversity Strategy for 2030 (BS)** and relation to NRL text. Column 'Is a similar challenge expected…': 0 = not relevant; 1 = partly relevant; and 2 = relevant.

Challenges of implementation	(How) does the NRL address this challenge?		Is a similar challenge expected for the implementation of the NRL?	Recommendations for NRL implementation to overcome the challenge
Policy				
Political collaboration • Success relies on willingness of Member States, as the targets are not binding and coordination across and within Member States (e.g. designation of protected areas or Natura 2000 sites) • Management plans often not in accordance with EU standards • Conflict with short term economic objectives • Most protected area designations by country-based assessments, yielding inconsistencies, management gaps and lacking connectivity	• EU-wide platform for coordination and cooperation and provision of binding targets, approaches and timelines • EU-wide monitoring and reporting framework • Financial incentives for member states • 'EU Green Network' of natural and semi natural areas	1	Despite financial incentives and EU-wide coordination and cooperation planned, the success still relies on the EU-Member States. Question remains, if financial incentives/legal requirements will be sufficient to target short term economic objectives.	Internalisation of externalities, so that short term objectives are no longer economically sound, if not sustainable for the long term
Finance, economy and capacity				

Challenges of implementation	(How) does the NRL address this challenge?		Is a similar challenge expected for the implementation of the NRL?	Recommendations for NRL implementation to overcome the challenge
Economy • Competing interest in land/sea use may result in designation of protected areas in remote, isolated areas • Incompatibilities with other policies such as CAP (direct payments potentially increasing further intensification of agriculture) • Blue growth in marine areas can potentially compromise biodiversity protection	• EU Green Network (ecological corridors) • Financial incentives	1	Question remains, if the financial incentives proposed will be sufficient. The EU Green Network is a good start to connect restored habitats	Internalisation of long term effects of economic practice

Challenges of implementation	(How) does the NRL address this challenge?		Is a similar challenge expected for the implementation of the NRL?	Recommendations for NRL implementation to overcome the challenge
Funding/Monitoring • Financial resources are not sufficient to implement all desired measures • Payment gap for management of current protected areas • Lack of funding for monitoring past implementation projects • Funding from private sector may have other motives • Potential conflicts with stakeholders, lacking involvement or misinformation despite efforts	• Integration of NRL-objectives into EU-Funding programmes • Economic incentives for conservation and restoration	1	Appropriate and flexible funding will also be decisive for the implementation of National Restoration Plans under the NRL. There is an investment gap of over 40 billion euros for NRL (*79*)	Rather implement measures now than postponing them. Ensure long term funding dedicated for biodiversity (conservation covenants); use EU Taxonomy to ensure eligibility of private investments.
Institutional capacity • The success of the BS relies on capacities of national authorities, NGOs and research and public engagement - joint work is a challenge	Not addressed	2	The same problem is likely to occur	Provision of enough funding needed and education
Implementation barriers				

Challenges of implementation	(How) does the NRL address this challenge?		Is a similar challenge expected for the implementation of the NRL?	Recommendations for NRL implementation to overcome the challenge
Public awareness • Some of the currently protected areas are used for recreational activities with potential negative impacts	• Promotion of sustainable tourism practices • Education	2	Similar challenges are expected, because promotion and education alone will not be sufficient	Enforcement and potentially barriers needed to protect especially sensitive species from stress
External pressures • Current protected areas affected by global change such as climate change, invasive species • Ecosystem management sometimes rather harm than protect ecosystems (e.g. by planting exotic trees) • Simple planting of trees does not benefit biodiversity, but rather forest renewal	• Linking existing protected areas across EU Member States - more connected networks are more resilient • New tools and approaches of management and conservation	1	Although the linking of existing and new protected areas is likely to increase the resilience, external pressures is likely to increase over time	Continuous adaptation to global changes
Conflicts in targets • Potential conflicts between climate and biodiversity protection	Integrated planning approach to find areas where both can be achieved simultaneously	1	Despite integrated planning approaches, it is still likely that the space will not suffice the needs for both biodiversity protection and the constantly growing economy	Transition of economy needed to live within 'planetary boundaries'

Challenges of implementation	(How) does the NRL address this challenge?		Is a similar challenge expected for the implementation of the NRL?	Recommendations for NRL implementation to overcome the challenge
International cooperation • The Strategy does not specify how 'externalisation' of biodiversity burden should be prevented (e.g., rain forests)	Promotion of sustainable and responsible consumption support of international efforts of biodiversity protection and EU-trade agreements	1	Global crises and trade agreements cannot be predicted and influenced easily. Therefore, externalisation cannot be precluded easily.	Aiming for global cooperation and coordination e.g. by investment funds
Data and monitoring				
Data • Soil biodiversity only partly addressed, rather as a side effect from other measures	Increasing share of organic farming (Article 9, currently removed) requires national strategies for soil health improvement	1	Protection of soil biodiversity still appears to be considered a 'by product' but is at least highlighted	Clear focus on transition to more sustainable land uses

Table S3e. Implementation challenges for meeting the biodiversity-related targets of the **Forest Strategy (FS)** and relation to NRL text. Column 'Is a similar challenge expected…': 0 = not relevant; 1 = partly relevant; and 2 = relevant.

Challenges of implementation	(How) does the NRL address this challenge?		Is a similar challenge expected for the implementation of the NRL?	Recommendations for NRL implementation to overcome the challenge
Policy				
Conflicting Objectives: Forest restoration activities may involve the removal of certain tree species or modification of forest landscapes, which could potentially conflict with biodiversity conservation (e.g. of grasslands) and sustainable forest management	Indirectly addressed by the requirement of national restoration plans that may address these conflicts	2	Conflicts between restoration targets and approaches for different ecosystems may occur, but due to the specificity of the targets defined for individual ecosystem types these will be limited	Identifying priority areas for the restoration of individual ecosystems in the national restoration plans
Finance, economy and capacity				
Stakeholder engagement and participation: Ensuring effective involvement of diverse stakeholders, including forest owners, local communities, NGOs, and indigenous groups. It requires transparent decision-making processes, capacity building, and addressing conflicting interests and power dynamics to avoid resistance from stakeholders.	National restoration plans offer additional options to liaise different stakeholders groups including governments, private sector actors, local communities, and NGOs	1	Liaising stakeholder interests and gaining support from various societal groups for restoration activities is a challenge for the NRL as well	Aligning the NRL with the FS, there is an opportunity to strengthen forest governance, promote stakeholder participation, and ensure the effective implementation of restoration measures in forested areas
Implementation barriers				

Challenges of implementation	(How) does the NRL address this challenge?		Is a similar challenge expected for the implementation of the NRL?	Recommendations for NRL implementation to overcome the challenge
Climate change impacts: Significant impacts on forests and the forest sector expected, which will require adaptation measures and sustainable forest management	The focus of the NRL is much more on restoration rather than on sustainable forest management	2	The NRL's targets are similarly affected by climate change. At the same time, the NRL can contribute to climate change mitigation efforts by emphasising reforestation and afforestation activities.	Implementation should include adaptive management approaches
Competing land use Priorities: The availability of land for restoration projects and sustainable forest management may be limited due to competing demands for agriculture, urbanisation, or infrastructure development	Indirectly addressed by the requirement of national restoration plans that may address these conflicts	2	Conflicts between targets for individual ecosystem types are likely to occur as well	Definition of priority areas for restoration of individual ecosystem types
Data and Monitoring				
Establishing robust monitoring frameworks, data collection systems, and indicators to track the implementation of restoration measures, sustainable forest management practices, and the achievement of desired outcomes is complex and resource-intensive.	The NRL specifies a number of seven indicators for forest biodiversity, which are straightforward to address but may still need further refinement. Six out of seven indicators shall be chosen.	1	Though indicators have been defined, they may still need refinement to enable a straightforward monitoring	Monitoring and evaluating the outcomes of restoration activities to enable evidence-based improvements in restoration and management practices

Table S3f. Implementation challenges for meeting the biodiversity-related targets of the **Common Fisheries Policy (CFP)** and relation to NRL text. Column 'Is a similar challenge expected…': 0 = not relevant; 1 = partly relevant; and 2 = relevant.

Challenges of implementation	(How) does the NRL address this challenge?		Is a similar challenge expected for the implementation of the NRL?	Recommendations for NRL implementation to overcome the challenge
Policy				
Eco-schemes in most countries are not ambitious enough to generate a change in management (= restoration), funding periods not always align with the time needed to achieve environmental outcomes.	not addressed	1	The NRL will build on existing legislation, strategies and policies such as the possibility to set up eco-schemes. However, the target of restoration may result in higher ambitions of the EU-member states for eco-schemes.	Consider more ambitious, Europe-wide Eco-schemes
CFP alignment with MSFD objectives. Complex policy framework with numerous regulations can pose challenges for fishers understanding and complying with requirements.	Aiming to keep administrative burden as limited as possible, ensuring appropriate infrastructure for public access, reporting and data-sharing between public authorities	1	Although this point is regarded, the administrative burden is unlikely to disappear completely.	Better engage stakeholders and try to simplify, without oversimplifications
Finance, economy and capacity				

Challenges of implementation	(How) does the NRL address this challenge?		Is a similar challenge expected for the implementation of the NRL?	Recommendations for NRL implementation to overcome the challenge
Mechanisms to compensate fishers if areas are taken out of fishing especially for restoration and rewilding	Compensation through incentives and buyers of ecosystem services	2	Funding appears not sufficient to compensate for all nature restoration, as largely based on existing legislation, strategies and policies that lack funding.	Provision of funding needed, as long-term return expected
Overall CFP investments should be better balanced with biodiversity conservation.	not addressed	1	The NRL will build on CFP, so it will face a similar challenge.	Increase incentives for environmentally friendly and sustainable fishing methods.

Table S3g. Implementation challenges for meeting the biodiversity-related targets of the **Common Agricultural Policy (CAP)** and relation to NRL text. Column 'Is a similar challenge expected…': 0 = not relevant; 1 = partly relevant; and 2 = relevant.

Challenges of implementation	(How) does the NRL address this challenge?		Is a similar challenge expected for the implementation of the NRL?	Recommendations for NRL implementation to overcome the challenge
Policy				
The CAP's Good Agricultural and Environmental Conditions (GAEC) set too low requirements in terms of area and quality, and most farmers are exempt of these. Even with these, many farmers do not comply, due to tradition, resistance to change, and insufficient monitoring.	not addressed	1	Even though stakeholder involvement is planned and may contribute to improve and specify targets, the NRL is largely dependent on a more ambitious CAP	Joint involvement with farmers and environmentalists to work on the future of farming, including agricultural transition
Eco-schemes in most countries are not ambitious enough to generate a change in management (= restoration), the yearly funding approach does not align with the time needed to achieve many of the environmental outcomes	not addressed	1	The agri-environmental targets of the NRL are also largely dependent on member states' implementation, however restoration may result in higher ambitions of the EU-member states for eco-schemes, particularly in Natura2000 sites	Close cooperation of NRL and CAP jointly defining more ambitious, Europe-wide eco-schemes

Challenges of implementation	(How) does the NRL address this challenge?		Is a similar challenge expected for the implementation of the NRL?	Recommendations for NRL implementation to overcome the challenge
Conflict of CAP objectives (time-lag): CAP balances multiple objectives, including environmental protection, economic viability, social challenges - while trade-off occur among objectives. Long-term, short term economic considerations (by agricultural lobbies and pharma-industry) and farm income sometimes overshadow environmental priorities and consequently, dilute ambitions of environmental protection measures.	Focussing on Ecosystem Services and Economics of Biodiversity	1	Contrasting objectives with land users (in particular farmers) are expected	Ensure funding and independence of lobbyism and involve investment funds and insurance companies
CAP fails to integrate objectives of WFD, Sustainable Pesticide Use directives, SDGs, Aichi Targets, Green Deal. Complex policy framework with numerous regulations can pose challenges for small farmers understanding and complying with requirements and requires a lot of administrative capacity	Aiming to keep administrative burden for all entities as limited as possible, ensuring appropriate infrastructure for public access, reporting and data-sharing between public authorities	1	Although this point is regarded, the administrative burden is unlikely to disappear completely	Better engage stakeholders and try to simplify, without oversimplifications
Finance, economy and capacity				

Challenges of implementation	(How) does the NRL address this challenge?		Is a similar challenge expected for the implementation of the NRL?	Recommendations for NRL implementation to overcome the challenge
CAP lacks mechanisms to compensate farmers if land is taken out of production especially for restoration, including rewetting and rewilding	Compensation through incentives and buyers of ecosystem services	2	As funding appears not sufficient to compensate for all nature restoration, as largely based on existing legislation, strategies and policies that lack funding. The European Development Fund and Horizon Europe would need to step in.	Provision of funding needed, as long-term return expected
Agricultural Environmental and Climate Measures (AECM) have insufficient budgets and, in some areas (e.g. rich soils), low uptake by farmers	not addressed	2	Funding is not clear yet, so similar problems of funding are to be expected	Ensure enough funding (maybe in cooperation with insurance companies), as in the long term the investments are likely to be fruitful; incentivise Member States to seek coherence between CAP and NRL
Overall CAP investments are not balanced between business as usual farming and biodiversity-friendly farming, with predominant uniform payments least effective for biodiversity conservation	not addressed	1	Depending on how strongly the NRL will build on the CAP, similar problems are expected	Increase incentives for environmentally friendly farming methods
Implementation barriers				

Challenges of implementation	(How) does the NRL address this challenge?		Is a similar challenge expected for the implementation of the NRL?	Recommendations for NRL implementation to overcome the challenge
Lack of targeted measurements: CAP does not adequately address diverse environmental issues across the different European regions, failing to address region-specific challenges (e.g., regarding soil conditions, climate, etc.), nor focus on small farms (High Nature Value Farmlands)	By being based on several legislation, strategies and policies including EU soil strategy	0	No	Work interdisciplinarily
Missing indicators for impacts on biodiversity	NRL introduces new three well-established indicators within Article 9, and Article 8 addresses pollinators	1	The new indicators offer a significant improvement but the list of (non-pollinator) indicators is very short. Monitoring and reporting requirements (every 6 years) is insufficient to assess the status and respond appropriately.	Promote and incentivise yearly monitoring regardless of reporting requirements by the Commission

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Author contributions

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- Statistical analysis – not applicable
- Writing the manuscript – 15%
- Revision of the manuscript – 10%

Signature of the Doctoral Candidate | Signature of the Doctoral Supervisor

4 General Discussion

4.1 Achievements and their relevance

The research compiled in this thesis presents compelling evidence for the detrimental effects that present-day agriculture imposes on river ecosystems. The individual chapters uncover various influencing factors, analyse effects of agricultural practices in comparison to urban areas, which are often also identified as a major cause for freshwater deterioration, and other land use types, and unravel the pathways of agricultural stress. The agricultural effects likely differ based on specific pesticide- and nutrient application rates, with macroinvertebrates most strongly responding to pesticide stress, while diatoms appear more strongly impaired by nutrients. Additionally, these chapters offer viable mitigation strategies and offer suggestions on how to protect and restore healthy ecosystems.

In the first chapter, Schürings et al. (2022) analysed the existing literature in form of a meta-analysis, revealing a globally consistent negative effect of agriculture (North America, Europe, and Oceania), while particularly sensitive species are strongly impaired and tolerant species may potentially even benefit. The agricultural impact is influenced by several factors including ecoregions, the intensity, practices, and types of agriculture. The agricultural impacts assessed also differ based on the biological response measured, suggesting using environmental quality metrices such as the ecological status for the assessment of agricultural impacts. Also, agricultural effects differ between organism groups with strongest effects on macroinvertebrates. This gathered evidence can provide a basis for global legislation to alleviate the agricultural impacts on rivers. The identified influencing factors, particularly the difference among agricultural types and intensities should be taken under consideration when implementing mitigation measures. From the ecological point of view, this chapter suggests the use of quality metrics rather than using sheer richness and abundance metrices for agricultural stress assessment.

In the subsequent chapter, Schürings et al. (2023) worked on disentangling the effects of different agricultural practices by establishing a pan-European agricultural typology of 20 so called ‘Areas of Farming-induced Freshwater Pressures’, based on similarities in agricultural practices and types, agricultural production intensity, and exerted freshwater pressures. The compiled cumulative pressure map depicts the distribution of agricultural pressures among Europe with strongest freshwater pressures found for intensive cropland in the Mediterranean and Temperate regions. The presented individual pressure maps based on agricultural types, weighted by the freshwater pressures of pesticides, nutrients, water abstraction, and hydromorphological alterations, offer an overview for the regions, in which each stressor is most prevailing. The typology can be used to identify pressure hotspots, allow to relate agricultural intensity with environmental pressures and can hence be used to advice for pan-European management of freshwater ecosystems including concepts such as land-sharing and land-sparing (Fischer et al., 2014). Using the Areas of Farming-induced Freshwater Pressures and the specific freshwater pressures, also an agricultural intensity index was derived, which allowed to increase the correlations with the ecological nearly twofold. However, the correlations are small in general, owing to the study´s methodological limitations, which highlights the urgent need for data of higher resolution and comparability across the continental scale, to allow for more sophisticated advice for efficient and successful global environmental legislation. Still, the resulting correlation clearly exceeded other studies with Europe-wide datasets (e.g. Lemm et al., 2021), which highlights the great potential of considering difference in agricultural pressure intensity when addressing agriculturally caused environmental challenges at the continental scale.

In the third chapter, Schürings et al. (2024b) in depth investigated the effects of different agricultural crop types on macroinvertebrates, macrophytes and diatoms at the German-wide scale, utilizing recently available high-resolution agricultural land use maps and relating them with biological data derived from the Water Framework Directive monitoring. The findings show major crop-specific differences, likely associated with agrochemical application. Permanent crops and vegetables, which are linked to high pesticide application rates (Dachbrodt-Saaydeh et al., 2021), were associated with strongest effects on macroinvertebrates and macrophytes. For crop types with high nutrient application rates such as maize and cereals (Britz & Witzke, 2014), the strongest effects on diatoms were observed. These different associations between the pesticide and nutrient application rates and the effects on river biota suggest legislation to both reduce of agrochemical usage (Schleiffer & Speiser, 2022) to reduce the pressure, and to protect freshwater ecosystems with riparian vegetation (e.g. Palt et al., 2023).

In the fourth chapter, Markert et al. (2023) compiled evidence for the associations revealed in the third chapter, particularly the crop type-specific pesticide application rates, proposing a cause-effect relationship between agricultural land use and pesticides in the rivers by highlighting clear relations between the catchment land use and pesticide, industrial chemical, and pharmaceutical concentrations in rivers, measured in course of the micropollutant monitoring of the Water Framework Directive. This study revealed micropollutant concentrations exceeding the environmental quality standards and could show that while urban land related most strongly to pharmaceuticals and industrial chemicals, agricultural land use showed strongest relations with pesticides. Also, at the crop-specific level, the concentrations measured in the rivers directly mirrored pesticides applied. These results give advice to management and legislation by showing that pharmaceutical pollution may be alleviated with a fourth stage of wastewater treatment plants (Wang et al., 2020), while diffuse pesticide pollution from agriculture needs legislative and management action, such as in course of an alternative for the recently rejected EU sustainable use regulation (European Commission, 2022a).

In the fifth chapter, Schürings et al. (2024a) drew from the results of the second, third and fourth chapter, by weighting different agricultural types by crop-specific pesticide and nutrient application rates. It could be shown that accounting for agrochemical application intensity can improve the correlative strength of agriculture with the ecological status, yielding an R^2 of up to 0.43. They found that the agricultural effects are strongly depending on the river type and size, as well as the organism groups assessed. Macroinvertebrates and macrophytes appear more strongly affected by pesticides, while diatoms seem more nutrient dependent. Consequently, it is highly important to regard different organism groups for monitoring. Moreover, these results offer further evidence for the detrimental effects of agrochemicals in present-day agriculture and suggest accounting for differences in management intensity when addressing agricultural impacts and a shift to more sustainable agricultural practices with lower agrochemical usages such as organic farming (Gamage et al., 2023).

In the last chapter, Hering et al. (2023) assessed the proposed European Union´s Nature Restoration Law (NRL) in context of the other European pieces of environmental legislation: Fisheries Policy, Marine Strategy Framework Directive, Habitats Directive, the Birds Directive, Biodiversity Strategy, Forest Strategy, Water Framework Directive and Common Agricultural Policy. This study draws from experiences and challenges of implementation of the existing environmental legislation to advice successful implementation nature restoration and biodiversity protection, while circumventing potential challenges in advance. A particular focus lies on the conflict with the Common Agricultural Policy (CAP), which appears to have interfered with the successful implementation environmental legislation in the past (Alliance Environment, 2019; ECA, 2020). Cooperation between ecosystem restoration legislation such as of the NRL and the agricultural legislation of the CAP would likely enable unprecedented opportunities (Pe’er et al., 2020; Pe’er et al., 2022) and enable a societal transition to a more sustainable future, halting biodiversity decline and preserving ecosystem health.

4.2 Outlook

In view of this overwhelming evidence, a transition to more sustainable agricultural practices is no longer merely a choice – it is imperative to effectively respond to the human induced biodiversity and ecosystem health crisis. This transition should encompass a comprehensive paradigm shift that harmonizes human food production with ecological resilience (Wezel et al., 2020). Organic farming, permaculture, agroecology, mosaic farming and precision agriculture are a few examples of alternative approaches that try to integrate natural processes into farming practices and are likely less detrimental. Organic farming, for instance, eliminates synthetic pesticides and fertilizers, relying instead on organic matter to enhance soil fertility and microbial diversity (Reganold & Wachter, 2016; Gamage et al., 2023). Permaculture promotes self-sustaining ecosystems by mimicking natural patterns and using diverse crops to minimize the need for external inputs (Rhodes, 2012; Kebs & Bach, 2018). Similarly, agroecology is based on ecological concepts (González-Chang et al., 2020), for instance prioritizing crops in their inherent agro-climatic favorable regions facilitating higher yields and creating massive cropland sparing potential, particularly for biodiversity hotspot regions (Folberth et al., 2020). Mosaic farming, characterized by a mixture of various crops, land uses and peripheral zones including beneficial hedgerows, fosters biodiversity by creating habitats for a multitude of species (Précigout & Robert, 2021; Tscharntke et al., 2021). Precision agriculture uses technology to target inputs precisely, minimizing waste and environmental impact (Gebbers & Adamchuk, 2010). All those approaches oppose large-scale highly mechanized monoculture farming, but favor rather smaller farms, for which higher yields and more biodiversity have been identified by Ricciardi et al. (2021).

Such more sustainable agricultural practices inherently align with the principles of ecosystem-based management, where ecological integrity and resilience become integral components of agricultural systems (Koohafkan et al., 2012; Mabhaudhi et al., 2022). They capitalize on the services provided by ecosystems, such as pollination, nutrient cycling, and pest regulation (Andersson et al., 2012; Rutsch et al., 2016; Hirschfeld & van Acker, 2021), to promote both productivity and environmental health. With these practices a shift away from the monoculture and intensive agrochemical input farming towards mitigating pollution, conserving water, and restoring habitats may be achievable. Such an agricultural transition can foster biodiversity by creating habitats for a multitude of species, promoting genetic diversity, and allowing ecosystems to thrive (Wezel et al., 2020). While this shift in agricultural practices is likely to benefit all ecosystems, river protection can be further supported by ensuring continuous riparian vegetation adjacent to rivers and streams, for which several studies (Kail et al., 2022; Palt et al.,

2023) showed the high potential to successfully enhance the ecological health of river ecosystems.

It is essential to recognize that the transition to sustainable agriculture is not only important just for the sake of the environment, but also for the future viability of agriculture itself and such a change does not need to drastically reduce productivity. Tamburini et al. (2020) showed that agricultural diversification can go without compromising the agricultural yield by promoting multiple ecosystem services including pest control, water regulation, nutrient cycling, and soil fertility. Similarly, Dainese et al. (2019) revealed noteworthy biodiversity-mediated benefits for agriculture. Unsustainable agricultural practices on the other hand, such as omnipresent in conventional agriculture does compromise the very ecosystems and their services, on which agriculture depends on (Pilling et al, 2020). For instance, pesticides have been shown to impair pollinating insects (Serrão et al., 2022) and to reduce the symbiotic capacity of nitrogen-fixing rhizobia, essential for several agricultural crops (Fox et al., 2007). Soil erosion leads to declines in the soil fertility (Pimentel & Burgess, 2013) and agrochemical pollution of rivers has direct repercussions for agriculture benefitting ecosystem services and on agricultural productivity (Dainese et al., 2019; Racoviceanu et al., 2023). Consequently, transitioning to sustainable practices also is an investment in the long-term resilience of agriculture, which will improve the capacity of agriculture to respond to the rapid global change including climate change (El Chami et al., 2020), and is essential to safeguard human sustenance.

The transition to a sustainable agriculture could be supported by the reduction of food waste (Santeramo & Lamonaca, 2021) and sharing the target of feeding humanity with urban agriculture (Artman & Sartison, 2018; Langemeyer et al., 2021) and the revival of home gardening (Lal, 2020). Also, the use of robots in agriculture (Sparrow & Howard, 2021), particularly for labor intensive agricultural approaches such as permaculture and agroecology (Donovan & Coming, 2010; Akram-Lodhi, 2021) may prove useful. Lastly, also the social aspect must not be disregarded to ensure healthy, long-lasting relationships with the people contributing to food production and enable them to actively participate and resolve sustainability deficits of their production (Grenz et al., 2009). For instance, the dietary choices of society have an enormous capacity of reducing environmental impacts, as shown in a recent study highlighting three-to-four-fold higher dietary based environmental impacts of meat consumption in comparison to vegan diet on greenhouse gas emissions, land use, water use, eutrophication, and biodiversity (Scarbourough et al., 2023). Hence, the question arises if humanity can still afford to dedicate most of the agricultural land to grow fodder for meat production, given the apparent negative effects of present-day agriculture on biodiversity and ecosystem services. Also, the use of biomass from agricultural land for energy production is questionable, given the vast area needed (Reilly & Paltsev, 2009) and the urgency for nature restoration. A focus shift back to feeding humanity with reduction in meat consumption, while using sustainable practices would likely even allow for the dedication of agricultural areas to the urgently required nature conservation (Wirsenius et al., 2010; Dinerstein et al., 2020), jointly benefiting global environmental stability and biodiversity. However, the successful implementation of agricultural transition will strongly depend on the political and societal will (Loorbach & Rotmans, 2006; Frantzeskaki et al., 2012), as currently short-term economic goals as of the chemical industry and farmers are often circumventing (Cortina-Segarra et al., 2021). To convince society and politics, further research, whether the alternative, presumably more sustainable, approaches can successfully decrease the agricultural burden for freshwaters, biodiversity, and ecosystem health and how to implement such a transition in a social just manner, is urgently required.

At the European scale, an ambitious post-2027 renewal of the Common Agricultural Policy following suggestions of Pe’er et al. (2022), joint with a successful implementation of the Nature Restoration Law (European Commission, 2022b) and an alternative for the rejected Sustainable Use Regulation (European Commission, 2022a) could be a good first step to face the challenge of our time of environmental destruction. Since, in a world, in which living unsustainable is economically viable, a transition to more sustainable ways of living is strongly impeded. A shift to a more sustainable economy, where the most sustainable pathways coincide with the most economically sound ones, will be crucial to successfully halt the global environmental crises, but also to ensure sustainable food production and food security long. While conventional agriculture may be economically sound short term, long term a change in agricultural practices is imperative from both an ecological and economic perspective. For this shift, approaches such as the internalization of externalities (Pretty et al., 2001; Deke, 2004; McElwee et al., 2020; Nedopil, 2023), societal and farmer participation (Hauser et al., 2016; de Boon et al., 2022) and a focus on nature-based solutions (Williams et al., 2020; Seddon et al., 2020; Miralles-Wilhelm, 2021) would certainly be beneficial. In summary, the environmental burden of present-day agriculture necessitates a shift to a more sustainable agricultural food system. The above proposed alternative agricultural food systems can offer healthier food and long-term food security, while increasing ecosystem services and protecting or even enhancing biodiversity and ecosystem health.

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The curriculum vitae is not included in the online version for data protection reasons.

Declarations

Declaration

In accordance with § 6 (para. 2, clause g) of the Regulations Governing the Doctoral Proceedings of the Faculty of Biology for awarding the doctoral degree Dr. rer. nat., I hereby declare that I represent the field to which the topic "Effects of agricultural land use types on the ecological status of rivers in Germany: A large-scale analysis" is assigned in research and teaching and that I support the application of Christian Schürings.

Essen, date ____________ ____

Name and Signature of the scientific supervisor/member of the University of Duisburg-Essen

Declaration

In accordance with § 7 (para. 2, clause d and f) of the Regulations Governing the Doctoral Proceedings of the Faculty of Biology for awarding the doctoral degree Dr. rer. nat., I hereby declare that I have written the herewith submitted dissertation independently using only the materials listed and have cited all sources taken over verbatim or in content as such.

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Signature of the doctoral candidate

Declaration

In accordance with § 7 (para. 2, clause e and g) of the Regulations Governing the Doctoral Proceedings of the Faculty of Biology for awarding the doctoral degree Dr. rer. nat., I hereby declare that I have undertaken no previous attempts to attain a doctoral degree, that the current work has not been rejected by any other faculty, and that I am submitting the dissertation only in this procedure.

Essen, date ____________ ____

Signature of the doctoral candidate