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Current challenges, methods, and strategies for reducing the transfer of nonpoint source pollution from agricultural areas to surface water

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Abstract: Agricultural diffuse pollution, also referred to as nonpoint source pollution, is widely recognised as one of the primary challenges to achieving good ecological status in surface waters. This paper synthesises current knowledge on strategies and technical approaches designed to reduce nutrient and contaminant transfers from agricultural landscapes to rivers, lakes, and reservoirs. A broad spectrum of mitigation measures is assessed, including riparian buffer zones, vegetated filter strips, grassed waterways, constructed wetlands, denitrifying bioreactors, permeable barriers, stormwater management on agricultural land, and Good Agricultural Practices. The effectiveness of these measures is examined in relation to biophysical, hydrological, and geomorphological conditions, as well as their ability to provide additional ecosystem services, such as biodiversity enhancement and flood mitigation. Particular emphasis is placed on emerging global challenges, including climate change, which alters precipitation patterns and increases the frequency of extreme weather events, and the growing presence of microplastics and nanoplastics. Persistent barriers to implementation are identified, including fragmented governance frameworks, economic constraints, slow ecological responses, and limited stakeholder engagement. Advances in remote sensing, and geographic information systems are highlighted as essential tools for identifying critical source areas, optimising land management strategies, and improving spatial planning at the catchment scale. A prevention-focused hierarchy of measures is proposed, supported by adaptive and integrated water resource management principles. This paper offers a comprehensive synthesis of scientific and practical insights intended to support policy development, guide effective environmental management strategies, and contribute to achieving Sustainable Development Goal 6 and related European Union water quality objectives.

Keywords: agricultural diffuse pollution, buffer zones, constructed wetlands, nature-based solutions, nutrients, SDG 6, sustainable water management, water quality

INTRODUCTION

According to the Water Framework Directive (2000/60/EC) and the Floods Directive (2007/60/EC) described by European Commission (2019), over 60% of rivers and lakes in Europe fail to meet the criteria for good ecological status. In various locations across Europe, water quality does not meet the targets of the EU Water Framework Directive due to agricultural intensification. The development of agriculture to meet human food needs has increased the use of fertilisers and pesticides, leading to the loss of mineral components in water from agricultural areas to surface water through runoff, drainage and causing eutrophication (Fan *et al*., 2019; Zhang *et al*., 2020; Zhong *et al*., 2020; Mararakanye, Le Roux and Franke, 2022; Fudała, Bogdał and Kowalik, 2023).

Agricultural pollution can be classified into two main types: point source and nonpoint source (NPS) pollution. Point source pollution originates from identifiable, discrete sources, such as discharge pipes releasing contaminants into water bodies. In contrast, NPS pollution comes from widespread areas, making it harder to trace to a single origin. It is typically associated with runoff from agricultural fields, where rainwater or irrigation washes fertilisers, pesticides, and sediments into nearby rivers, lakes or groundwater. Nonpoint source pollution is a major challenge in agricultural management due to its diffuse nature and dependence on various environmental factors such as weather patterns and land management practices (Stevens and Quinton, 2009; Kiedrzyńska *et al*., 2014; Wiering, Boezeman and Crabbé, 2020; Yuan, Sinshaw and Forshay, 2020; Kornijów, 2024).

Diffuse pollution in agricultural areas still greatly affects the quality of receiving water resources and biodiversity (Sojka *et al*., 2017; García-Galán *et al*., 2018; Górski, Dragon and Kaczmarek, 2019; Boardman, 2021; Mondon *et al*., 2021). Decades of research in the agricultural sector have shown that the main forms of NPS pollution, nitrogen (N) and phosphorus (P), from agricultural areas, significantly reduce the quality of water, impacting both the chemical and ecological status (Zhang, Ni and Xie, 2016; Mateo-Sagasta *et al*., 2017; Carstensen *et al*., 2020; Mojiri *et al*., 2020; Duan *et al*., 2023; Ma *et al*., 2023).

According to the European Environment Agency (EEA), nutrient pollution affects 27% of rivers, 25% of lakes and 37% of transitional waters of the European Union (EEA, 2018; EEA, 2021). Although EU countries have applied measures to improve nutrient concentrations in surface waters, the ecological status remains below 'good' at 59% of river monitoring sites for total nitrogen (TN), 57% for total phosphorus (TP), 64% of lake sites for TN, and 61% for TP (Nikolaidis *et al*., 2022). In Europe, diffuse agricultural pollution from agricultural lands to surface water is mainly due to the transfer of N and P (Withers and Haygarth, 2007; Wiering, Kirschke and Akif, 2023), as well as other products such as pesticides (Syafrudin *et al*., 2021). Notably, according to some studies such as that of Lassaletta *et al.* (2014), in most of the United States and Europe, the amounts of N transferred from agricultural areas through surface runoff are quite high (50 kg N·ha^{-1}).

The reasons why NPS pollution from agricultural lands is more difficult to treat than other types are: the intensity of pollutants varies over time and space, making it difficult to identify major sources, the transport and fate of pollutants depends on dynamic natural processes, such as precipitation and soil infiltration, as well as on complex chemical and physical processes. Moreover, effective control methods require costly changes in agricultural practices and long-term strategies that take into account the cumulative effects of past agricultural activities (Xia *et al*., 2020; Zhang *et al*., 2020; Mason *et al*., 2021). The runoff of agricultural nutrients affects both the environment and the economy. It is linked to fertiliser loss and environmental damage. These include the costs of lost ecosystem services and the expenses associated with implementing and maintaining remedial measures (Chen *et al*., 2023; Luna Juncal *et al*., 2023; Zhang, Luo and Zhang, 2024). Zalewski (2014) emphasises that the resilience of ecosystems is declining due to increasing anthropogenic stress, necessitating new approaches to achieve the sustainability of water and environmental resources. One potential solution lies in the integration of engineering, biotechnology, and ecohydrology.

Incorporating good agricultural practices (GAPs) is essential for mitigating agricultural runoff into water bodies. These practices include a range of technical methods of nutrient runoff mitigation, among others: riparian buffer zones (filter strips, vegetated streams, vegetated filter strips), grassed waterways, wetlands, constructed wetlands, woodchip bioreactors, conservation tillage, crop rotation, cover crops, contour farming, drip irrigation, drainage ditch vegetation, controlled release fertiliser, leaching fraction irrigation, land use change, no-till farming, biogeochemical nutrient removal, high-performance barriers and bioreactors, reduced fertiliser usage, reduced tillage, nature-based solutions, ecological floating beds (floating islands), breaking or shaping pathways of pollutant transfer to surface waters (Reichenberger *et al*., 2007; Zalewski, 2014; Dąbrowska *et al*., 2016; Pignalosa *et al*., 2022; Kumwimba *et al*., 2023; Luna Juncal *et al*., 2023). Various types of pollutants enter water from agricultural land, mainly N and P, but also suspended solids, pesticides, heavy metals, organic compounds, pathogens from livestock, emerging pollutants such as veterinary antibiotics, microplastics, nanoplastics, etc. (Chu *et al*., 2010; Lin *et al*., 2011; Astner *et al*., 2023; Mishra *et al*., 2023; Hoang *et al*., 2024; Tan *et al*., 2025).

In recent years, nature-based solutions (NBS) have gained increasing attention as effective tools to mitigate the environmental impacts of agricultural activities. Many studies have highlighted their potential to reduce the transfer of diffuse pollutants to surface waters, especially through approaches that integrate natural processes into land and water management (Mancuso *et al*., 2021; Bianciardi, Becattini and Cascini, 2023; Rizzo *et al*., 2023). In the context of agriculture, NBS include well-established measures such as buffer zones, constructed wetlands, and various techniques for sustainable stormwater management. These solutions not only support water quality improvement but also contribute to broader ecosystem services and biodiversity conservation.

The methods that limit the transfer of agricultural nonpoint pollutants to water are based on: limiting the generation of pollutants, nutrient retention, nutrient reuse, reducing their transport to waters, and restoration of polluted water resources according to the sequence of nutrient pollution generation and evolution (generation-flow-sink) (Dąbrowska, Dąbek and Lejcuś, 2018; Kumwimba *et al*., 2023). Thus, to prevent water pollution from agricultural activities, it is important to not only implement farming changes but also reduce the transport of pollutants in runoff (Aubertot *et al*., 2011). Preventative, source-based measures for nutrient management can significantly impact agricultural operations and have economic consequences. Performance-based interventions, such as buffer zones and constructed wetlands are reactive measures implemented after fertiliser application (Villamizar *et al*., 2020; Wiering, Boezeman and Crabbé, 2020).

In the EU, the problem of agricultural diffuse pollution is regulated and addressed by several documents, e.g. Water Framework Directive 2000/60/WE, Nitrates Directive 91/676/EWG, Directive on the sustainable use of pesticides 2009/128/WE, the EU's Common Agricultural Policy, and the European Green Deal (Harrison *et al*., 2019; Cole, Stockan and Helliwell, 2020; Wiering, Boezeman and Crabbé, 2020; Englund *et al*., 2021; Wiering, Kirschke and Akif, 2023). On September 25, 2015, in New York, the 2030 Agenda for Sustainable Development, containing the Sustainable Development Goals (SDGs), was adopted by all 193 United Nations member states with a General Assembly Resolution. The subject of nonpoint pollution of the agricultural type relates directly to the SDG 6 – clean water and sanitation, SDG 2 – zero hunger, SDG 12 – responsible consumption and production, SDG 14 – life below water, and SDG 15 – life on land (UN General Assembly, 2015).

Numerous authors, when discussing agricultural diffuse pollution, use the term "wicked problem", due to the complexity of pollution sources and mechanisms, the dynamic interaction between environmental and social factors, conflicts of interest between different social groups, scientific uncertainty and difficulty in monitoring long-term and global impacts. There is no universal solution that could effectively address all the challenges involved. Solving the problem requires an integrated approach, taking into account, i.e. socio-economic, technical, environmental and regulatory limits (Collins *et al*., 2016; Kumwimba *et al*., 2023; Wiering, Kirschke and Akif, 2023; Vega *et al*., 2024).

Although many studies have examined strategies to reduce pollutant runoff from agricultural watersheds to surface waters, few have systematically assessed these approaches in a comprehensive manner. This review aims to evaluate and synthesise existing methods for mitigating nonpoint source pollution from agriculture, with attention to their scientific, environmental, technical, economic, social, legal, and political dimensions. Current challenges in agriculture, including barriers to the implementation of sustainable development principles, are also taken into account. By critically analysing these factors, the review provides an integrated assessment of the effectiveness and feasibility of different measures, offering practical insights for water quality management at various scales.

SOURCES OF DIFFUSE AGRICULTURAL POLLUTION

As mentioned above, agricultural pollutants have been classified into main categories: nutrients, plant protection products, sediments, organic matter and emerging pollutants. However, the research will focus on nutrients and measures to reduce the transport of nutrients from agricultural catchments to surface waters.

For plants, N and P are essential nutrients for growth (Moe *et al*., 2019). From 2000 to 2018, the amount of N fertiliser increased by 33.1% and P increased by 18.4% globally to meet the sharp increase in food demand. However, promoting the use of agricultural chemicals causes significant harm to the environment and has been identified as a major source of water pollution in many countries, accounting for a higher proportion of both industrial and urban water pollution (Mateo-Sagasta, Marjani Zadeh and Turral, 2018; Pellegrini and Fernández, 2018, Jayasiri *et al*., 2022; FAOSTAT, 2024). The problem of biodiversity loss, lack of dissolved oxygen in surface water and eutrophication in rivers and lakes is due to N and P after fertilisation and its loss caused by surface runoff (Sharpley *et al*., 2015; Bechmann and Stålnacke, 2019; Xu *et al*., 2020; Li *et al*., 2023a).

Phosphorus (P) is the primary limiting nutrient in freshwater systems, whereas nitrogen (N) regulates primary production in marine environments. The transfer of nutrients from nonpoint sources to water bodies occurs through overland quickflow and belowground slowflow pathways. Overland flow plays a crucial role in transporting agricultural pollutants, serving as the primary route for phosphorus compounds – both dissolved and particulate forms – and as an important pathway for nitrogen compounds, including organic forms bound to eroded soil and soluble mineral forms (Rabalais, 2002; Dąbrowska, Dąbek and Lejcuś, 2018).

Phosphorus is a vital macronutrient for plant growth and development – it is involved in energy transfer, photosynthesis, transformation of sugars and starches, nutrient movement within the plant and the transfer of genetic characteristics from one generation to the next. Its sources are mainly fertilisers, manure, sewage and sludge. Phosphorus diffusion from agricultural soils into surface water is a major factor in water quality degradation, posing risks to public health and causing eutrophication (Ahemad *et al*., 2009; Dupas *et al*., 2015; EEA, 2019; Mallin and Cahoon, 2020; Penuelas *et al*., 2020). Over recent decades, intensive farming practices with high levels of organic and mineral P fertilisers have led to phosphorus accumulation in some European soils (Eurostat, 2013; Einarsson, Pitulia and Cederberg, 2020).

Nitrogen is a crucial nutrient for plant growth, and its main sources in agriculture are synthetic fertilisers, animal manure, and plant residues. It is a component of proteins, nucleic acids, chlorophyll, and growth hormones. Up to 40% of the N applied to agricultural soils in Europe is lost into the environment (Leip *et al*., 2011; Leghari *et al*., 2016). Elevated N levels in water can cause a decline in water quality within agricultural catchments, posing risks to both human health and the balance of natural ecosystems in those watersheds (Camargo and Alonso, 2006; Luo *et al*., 2023), and eutrophication in freshwater systems (Chaffin *et al*., 2018; Aubriot, 2019). A notable concern is that the EU must reduce annual nitrogen fertiliser use by approximately 43% to effectively limit surface water pollution (Vries de *et al*., 2021).

BEYOND ISOLATED INTERVENTIONS: THE IMPERATIVE OF A HOLISTIC APPROACH TO WATER MANAGEMENT

Effectively addressing water pollution from diffuse agricultural sources necessitates a holistic and integrated approach that considers the interplay of environmental, technical, social, economic, and policy factors. The complexity of nonpoint source pollution challenges, compounded by evolving threats such as climate change and emerging pollutants, requires solutions that go beyond isolated interventions. While technical measures like buffer zones and constructed wetlands offer localised benefits, their success depends on broader frameworks that integrate land-use planning, community engagement, economic incentives, and regulatory support. Furthermore, fostering interdisciplinary collaboration among scientists, policymakers, and practitioners is critical to developing scalable and sustainable strategies. In water resources management, the active involvement of society is essential. Ecohydrology advocates for aligning societal needs with the enhanced catchment's carrying capacity. A systems-based approach can bridge the gaps between fragmented efforts, ensuring that solutions are both locally adapted and globally relevant. Such a perspective aligns with the overarching principles of integrated water resources management (IWRM), which emphasise balancing the needs of ecosystems, agriculture, and society for long-term water security. It is crucial in IWRM to consider the drivers-pressures-state-impacts-responses (DPSIR) framework, linking the environmental and human systems, as well as adaptive water resource management based on the theory of ecohydrology (Zalewski, 2015; Dąbrowska, Dąbek and Lejcuś, 2018; Izydorczyk *et al*., 2019; Markowska *et al*., 2020; Morón-López, 2021).

The hierarchy for targeting nonpoint source pollution effectively (Jain and Singh, 2019), as depicted in Figure 1, emphasises the critical importance of preventing pollution at its source, with the reduction of chemical use being the primary and most effective measure. Eliminating or minimising the use of harmful substances is the cornerstone of sustainable water management practices. However, recognising the realities of agricultural systems and the challenges of achieving zero pollution, it is essential to implement and integrate all subsequent steps in the hierarchy. These include substituting synthetic chemicals with environmentally friendly alternatives, employing engineering controls to mitigate pollution, enforcing administrative controls, and ultimately utilising endpoint solutions like constructed wetlands to manage residual contamination. While prevention remains the primary goal, the inclusion of these additional strategies ensures a comprehensive and adaptive approach to addressing nonpoint source pollution.

The solutions presented in this study are positioned at various levels (ranging from prevention to mitigation) within the triangle illustrating the hierarchy for effectively targeting non-point source pollution. It is important to note, however, that they must be integrated into a broader holistic framework.

SELECTED METHODS TO REDUCE THE TRANSFER OF NONPOINT SOURCE POLLUTION FROM AGRICULTURAL AREAS TO SURFACE WATERS

BUFFER ZONES

A buffer zone (BZ) is an area of land with continuous vegetation, strategically located to separate agricultural areas from water sources, as defined by Wasilewski (2012). The problem in discussing BZs lies in the fact that different names are used for this type of edge-of-field measures. While terms such as buffer zones, riparian buffers, filter strips, buffer strips, vegetative filter strips, grass filters, vegetative buffer strips are used interchangeably or separated in the literature, their specific functions and configurations can vary to address water quality protection, erosion control, and habitat conservation, specific functions may also overlap (Mancuso *et al*., 2021; Kumwimba *et al*., 2023).

The efficiency of buffer zones depends on a number of factors – topography and geomorphology, their lateral, longitudinal, and vertical dimensions, species composition, density, age and condition of overgrowing plants, soil type, pollutant load and type, buffer management practices, local climate conditions and seasonal variation. The effectiveness of a BZ is influenced not only by its physical characteristics and the types of pollutants it needs to manage, but also by the intensity of pollutant transport. This is because surface runoff must reach the BZ as a broad, shallow overland flow, rather than as concentrated (channelised) flow (Mayer, Reynolds and Canfield, 2005; Reichenberger *et al*., 2007; Dąbrowska, Dąbek and Lejcuś, 2018; Ghimire *et al*., 2021; Wu *et al*., 2023). In addition to their main task of trapping pollutants and sediments, BZs also play a major role and provide

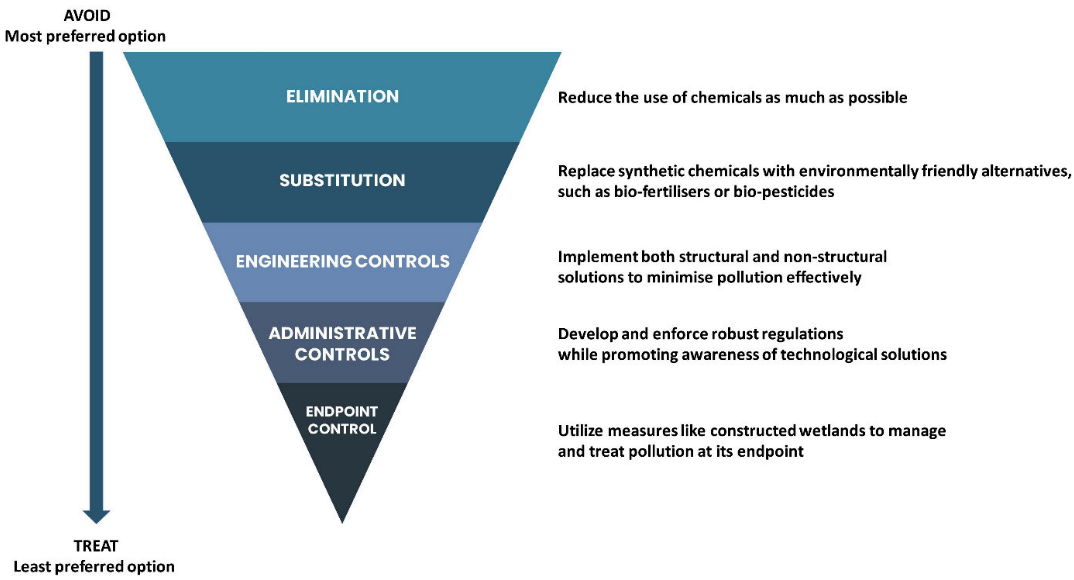

Fig. 1. Hierarchy for the effective targeting nonpoint source pollution; source: own elaboration based on Jain and Singh (2019)

a wider range of ecosystem services e.g. ensuring biodiversity, flood protection, regulation of aquatic thermal properties, bank stabilisation, landscape values etc. (Cole, Stockan and Helliwell, 2020; Graziano, Deguire and Surasinghe, 2022). The benefits, processes, and interactions of the riparian buffer zone with agricultural surroundings are illustrated in Figure 2.

Furthermore, they actively regulate runoff and reduce the flow of pollutants from land to water, as Miałdun and Ostrowski emphasised in 2010 (Miałdun and Ostrowski, 2010). Overall, buffer zones offer integrated physical, biological, and chemical protection to water bodies from both surface and subsurface pollution, thus contributing to the preservation of aquatic ecosystems (Wasilewski, 2012; Łaskawiec, 2015). To effectively purify water, BZs should consist of plant communities with diverse species compositions, including at least nine different plant species, according to Wasilewski (Wasilewski, 2012). The findings from the referenced studies show that buffer zones are effective in reducing pollution from pesticides, mineral fertilisers, and heavy metals. Therefore, BZs should be maintained and extended in agricultural regions (Helmers *et al.*, 2012; Hernandez-Santana *et al.*, 2013; Łaskawiec, 2015).

The BZs effectively reduce total phosphorus (TP) runoff from agricultural land, with nutrient retention rates reaching 97%. Wider BZs may be needed for higher surface runoff and erosion loads. In relation to that, Reichenberger *et al.* (2007) mentioned that mitigation measures for runoff/erosion and spray drift are more numerous than those for drainage and leaching. Vegetated buffer strips, particularly edge-of-field buffers, have been extensively studied and are more effective than riparian buffers in reducing pesticide runoff and erosion into surface waters.

The BZs come in various types and widths, each with differing capacities for removing biological pollutants, as reported by Blanco-Canqui and Lal (2010), including: deciduous forests with a buffer zone width of 10 m, achieving 97% efficiency in N removal and 78% in P removal; zones with deciduous trees and grasses, 75 m wide, which have N removal efficiency of 27% and P removal efficiency of 56%; areas containing trees, grasses, and shrubs with a width of 16 m, providing 94% efficiency in N removal and 91% in P removal; shrubs and weeds in an 18 m wide buffer zone, removing 32% of N and 30% of P.

Many researchers worldwide have extensively studied the width of riparian buffer zones, which range from 15 m to 200 m. Yet, there is no consensus on a standard width, as different needs may necessitate varying widths. It's commonly believed that wider BZs are more effective, although there are dissenting views (Lind, Hasselquist and Laudon, 2019; Luke *et al.*, 2019).

A buffer zone consisting of forests, grasslands, and shrubs serves a vital function in stopping pollutants from runoff reaching water sources. Its importance stretches to the sustained health of water and ecosystems, attracting considerable interest from scholars and governmental bodies alike. The efficacy of these BZs depends on factors like their width, the types of plants present, and how they are arranged (Wang *et al.*, 2024). Although

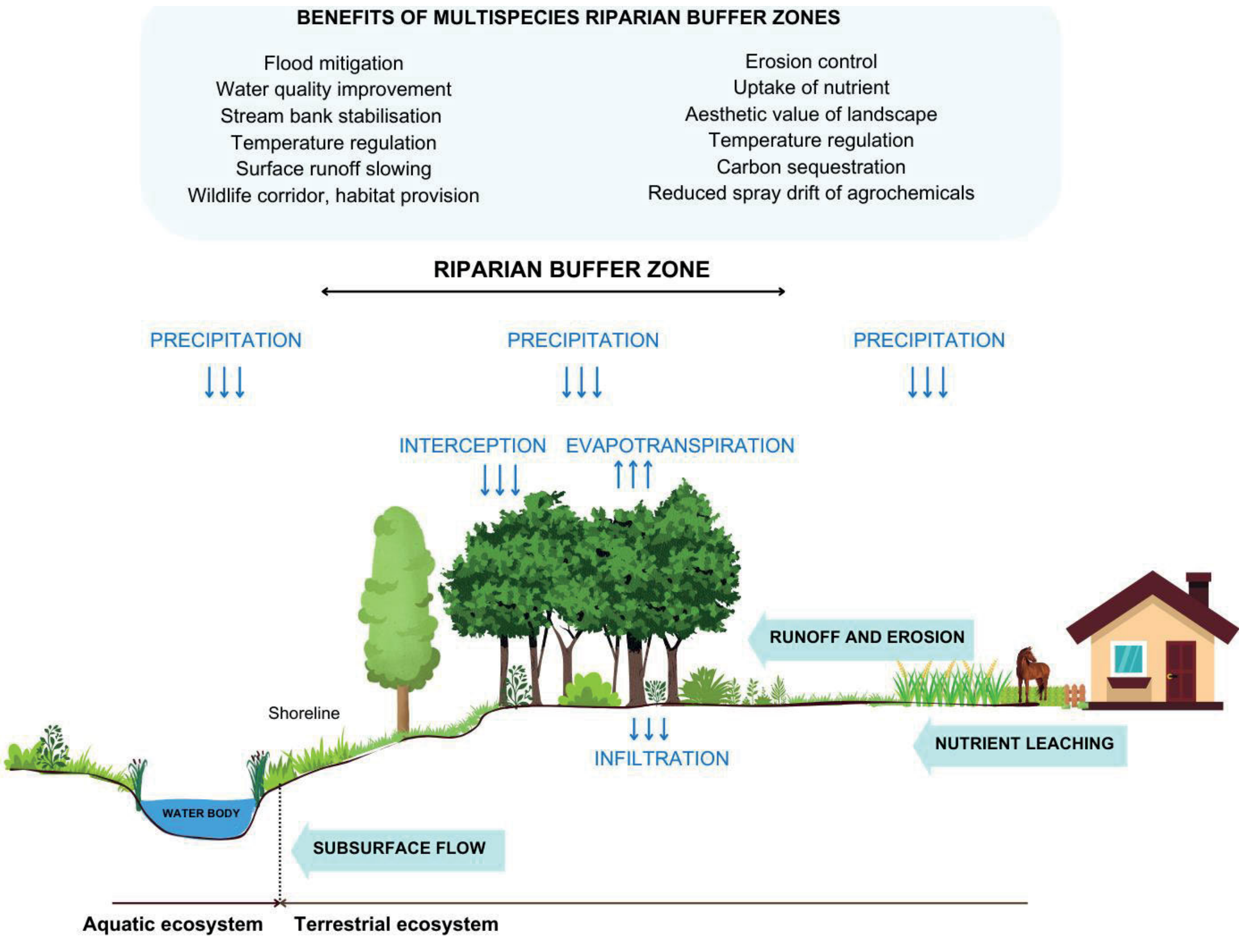

Fig. 2. Multispecies riparian buffer zones: benefits, processes, and interactions with agricultural surroundings; source: own elaboration

considerable research has examined the width and types of vegetation in buffer zones, there has been comparatively less emphasis on their length. Stanford *et al.* (2020) found that in scenarios where land is scarce, elongated and narrow riparian buffer zones can effectively mitigate pressures on aquatic environments.

In studies conducted by Abu-Zreig *et al.* (2003), field experiments examined the efficiency of vegetated filter strips in removing phosphorus from cropland runoff. The experiments involved 20 filters of varying lengths (ranging from 2 to 15 m), slopes (2.3% and 5%), and degrees of vegetation cover, with bare-soil plots serving as controls. The results showed an average phosphorus trapping efficiency of 61% across all vegetated filter strips, with individual values ranging from 31% for a 2 m filter to 89% for a 15 m filter. These findings indicate that filter length is the primary factor influencing phosphorus removal efficiency. In turn, Blanco-Canqui *et al.* (2004) emphasised that vegetative filter strips are not very effective at reducing sediment and nutrient losses in concentrated flow areas. In contrast, stiff-stemmed grass barriers are much more efficient, capturing 4.9 times more organic nitrogen, 2.3 times more ammonium nitrogen, and 3.7 times more particulate phosphorus than the vegetative filter strips at a distance of 0.7 m. The efficiency of sediment and nutrient trapping significantly improves as the length of the vegetative filter strips increases. Barriers could be a promising conservation approach for restoring lands impacted by concentrated flow, especially in cases where conventional methods are insufficient.

Buffer strips are an effective and cost-efficient method for reducing agricultural pollution. They remove agricultural contaminants such as nitrates and phosphates primarily through processes like microbial mineralisation, biological assimilation, and, specifically for nitrogen, denitrification (Krutz *et al.*, 2006; Chu *et al.*, 2010; Lin *et al.*, 2011). Local studies are needed to understand their performance, especially with narrow buffers. Studies conducted at an experimental farm of Padova University (Borin *et al.*, 2005) demonstrated the high effectiveness of a 6 m wide buffer strip. At the end of the monitoring period, cumulative mass losses with and without buffer strips were as follows: 0.4 and 6.9 $Mg \cdot ha^{-1}$ for total suspended solids, indicating a 94% reduction; 2.9 and 13.4 $kg \cdot ha^{-1}$ for total nitrogen, showing a 78% reduction; 1.3 and 3.1 $kg \cdot ha^{-1}$ for nitrate nitrogen, reflecting a 58% reduction; 0.3 and 0.8 $kg \cdot ha^{-1}$ for ammonium nitrogen, indicating a 63% reduction; 0.6 and 3.2 $kg \cdot ha^{-1}$ for total phosphorus, showing an 81% reduction; and 0.1 and 0.6 $kg \cdot ha^{-1}$ for phosphate phosphorus, indicating an 83% reduction. Although an increase in the concentration of all N forms was observed as water passed through the buffer strips, total nitrogen losses were reduced in terms of overall mass balance. The buffer strip consisted of two rows featuring a regular alternation of trees (*Platanus* × *hybrida* Brot.) and shrubs (*Viburnum opulus* L.), with grass (*Festuca arundinacea* L.) planted in the spaces between the rows. Other studies conducted at the same research site (Borin *et al.*, 2010) showed that, over a span of 3–5 y, young buffer strips decreased total runoff by 33%, nitrogen loss by 44%, and phosphorus loss by 50%.

Grass buffer strips for controlling diffuse phosphorus transfer are widely accepted by agricultural professionals. However, Dorioz *et al.* (2006) caution that their effectiveness is short-term, necessitating further research for long-term solutions.

Zhang *et al.* (2010) detailed how various factors influence phosphorus retention in vegetated buffer strips (VBSs). They found that the width of VBSs had the most significant effect on retaining P from overland flow. According to their analysis, a 30 m buffer on a favourable slope of about 10% can remove over 85% of the pollutants studied. Buffers consisting of trees are more effective at removing nitrogen and phosphorus compared to those made up of grasses or a mix of grasses and trees.

Diffuse pollution continues to pose a significant threat to surface waters due to eutrophication from phosphorus runoff from agricultural lands. VBSs are increasingly utilised to reduce diffuse P losses, effectively decreasing the transfer of particulate P from agricultural areas (Roberts, Stutter and Haygarth, 2012). They also help prevent suspended solids from entering water bodies by stabilising stream banks, reducing impacts on aquatic ecosystems like light penetration, temperature, and benthic structure. Moreover, they reduce the amount of pesticides and nutrients reaching surface waters from agricultural fields during rainfall or spring thaw events (Lovell and Sullivan, 2006; Bilotta and Brazier, 2008; Sweeney and Newbold, 2014; Zhou *et al.*, 2014; Franco and Matamoros, 2016; Hladik *et al.*, 2017; Wang *et al.*, 2018).

The effectiveness of buffer zones in reducing nutrient runoff is influenced by factors such as topography, vegetation type, buffer width, soil characteristics, climate, and the extent of nutrient loading (Hefting *et al.*, 2005; Mayer, Reynolds and Canfield, 2005; Dosskey *et al.*, 2010; Lam, Schmalz and Fohrer, 2011). The reported effectiveness of vegetated buffers in reducing the movement of pesticides and nutrients ranged from 10 to 100% and 12 to 100%, respectively (Prosser *et al.*, 2020). The type of vegetation significantly affects the removal of total suspended solids (TSS), total phosphorus (TP), and total nitrogen (TN). These findings indicate that well-designed and implemented grass-shrub buffers, even with widths as narrow as eight meters, can significantly improve water quality, especially if adequate infiltration is ensured (Mankin *et al.*, 2007). Mankin *et al.* (2007) emphasised that riparian buffer forests and vegetative filter strips are commonly recommended for enhancing surface water quality, whereas grass-shrub riparian buffer systems (RBSs) have received less attention. These buffers are highly effective in removing sediments, N and P, with their efficiency closely linked to infiltration rates. Buffer strips demonstrated high efficiency in removing sediments, nitrogen, and phosphorus, with removal effectiveness closely associated with infiltration processes. Moreover, Mankin *et al.* (2007) reported that average mass and concentration reductions were 99.7 and 97.9% for TSS, 91.8 and 42.9% for TP, and 92.1 and 44.4% for TN. Infiltration alone accounted for over 75% of TSS removal, and more than 90% of TP and TN removal. On the other hand, grass buffer strips can be more effective than wooded buffer strips at intercepting sediments and sediment-bound pollutants. While wooded buffer strips offer various ecosystem services, particularly in mitigating climate change impacts, their multifunctional nature makes developing management strategies more complex (Cole, Stockan and Helliwell, 2020).

As regards the water quality improvement effect, it is higher in multi-zone buffer zones than in those that include only one type of vegetation (Jiang *et al.*, 2020). Research indicates that the size of riparian forest buffers plays a crucial role in preserving water quality, as land use along riverbanks significantly influences water quality for at least one kilometre downstream. Even broad riparian forest buffers fail to enhance river water quality if they are too short. The study recommends riparian forest buffers of at least 15 m in width and extending 500 m in length to protect river water quality in tropical forest environments. This model could be applied worldwide to evaluate riparian buffer setups and balance water quality with broader human development goals (Brumberg *et al.*, 2021). Moreover, riparian buffers can disrupt the transport pathways of *E. coli*, influencing its concentration–discharge relationship. They are effective in reducing *E. coli* concentrations during both dry and wet weather conditions (Lim *et al.*, 2022).

Vegetative filter strips are an affordable best management practice that frequently offer substantial runoff reduction in agricultural areas with moderate slopes under 15% (Zhang, Bhattarai and Muñoz-Carpena, 2023). Agricultural practices in watersheds can substantially raise sediment and nutrient levels, posing risks to aquatic ecosystems. Riparian vegetated buffer strips offer a hopeful approach to capturing and storing these pollutants. While prior studies emphasise the effectiveness of woody vegetation in lowering nutrient levels, there's a key knowledge gap regarding how various types of vegetation (woody, shrubs, and grasses) reduce nutrient transfer from agricultural catchments to surface water (Kumwimba *et al.*, 2024).

Grassed waterways, which are conceptually similar to buffer zones, function as vegetated buffer areas primarily aimed at slowing the water flow and capturing sediments and pollutants from surface runoff before reaching watercourses. Fiener and Auerswald (2006) noted that grass-fed waterways (GWWs), which may be highly effective at the fluvial scale, have received little scientific attention. The GWWs are broad, shallow, grass-lined channels that drain surface runoff from large fields, preventing gullying. They are a common best management practice but are rare in regions with small fields, like many European countries (Fiener and Auerswald, 2017). However, most research has focused on buffer zones, assessing their sediment trapping, flow reduction, and pollutant trapping capabilities.

CONSTRUCTED WETLANDS

Vegetated constructed wetlands function as biofilters, with plants removing nitrogen and phosphorus and preventing eutrophication in water (Braskerud *et al.*, 2005). The review by Uusi-Kämppä *et al.* (2000) indicates that retention of TP increases with the surface-area to watershed-area ratio. Constructed wetlands (CWs), due to their shallow depth and dense vegetation, are more effective than ponds in retaining TP. Constructed and natural wetlands are utilised globally for nutrient removal from various effluents. However, denitrification within these systems may face limitations due to the availability of carbon for denitrifying bacteria, as noted by Kadlec (2005).

Constructed wetlands can significantly reduce agricultural contaminants like nitrate and pesticides. Nitrogen removal in CWs involves various processes, including volatilisation, nitrification, denitrification, and microbial uptake. While many processes convert nitrogen to different forms, only a few completely remove it from wastewater. Average nitrate removal efficiency ranges from 20 to 90%, though the effectiveness varies with season, hydrological flows, and pollutant properties. Other factors mainly support the natural microbiological purification processes. The objective of achieving 50% nitrate removal may be attainable with a wetland-to-catchment ratio of 1% (Tournebize, Chaumont and Mander, 2017). According to another study, TN removal in CWs ranges from 40 to 55%, and TP removal ranges from 40 to 60% (Vymazal, 2007). Research by Stevens and Quinton (2009) highlighted that these wetlands retain, on average, 69% (43–88%) of sediments, 35% (1–91%) of phosphorus, and 29% (11–42%) of nitrogen in agricultural catchments. Wetlands efficiently removed nitrate (22–99%) and total suspended solids (31–96%). Key factors influencing reductions were evapotranspiration, seepage, vegetation characteristics, and hydrologic residence time (Díaz, Ogeen and Dahlgren, 2012). In a retention pond in Canada, the mean removal efficiency ratios, calculated based on event-mean concentrations and event-total loads, were comparable to values reported for urban areas, showing efficiencies of 50–56% for total suspended solids, 42–52% for TN, and 48–59% for TP (Chrétien *et al.*, 2016). In turn, Gaballah and Lammers demonstrated that free water surface constructed wetlands remove, on average, 37.9% of TN based on data from 24 U.S. wetlands, and 55.1% of TP based on data from 38 wetlands (Gaballah and Lammers, 2025).

It is worth noting that treatment wetlands enhance water quality by employing biological processes to remove nitrogen, phosphorus, and suspended solids from runoff, sometimes matching the effectiveness of land management strategies in reducing nitrogen loads (Rousseau *et al.*, 2008; Land *et al.*, 2016; Haritash, Dutta and Sharma, 2017; Hansen *et al.*, 2018). Consequently, there has been substantial global investment in treatment wetlands aimed at enhancing water quality in agricultural environments (Jones, Hole and Zavaleta, 2012). An example from research by Cooper *et al.* (2019) shows that after 12 mo, a roadside constructed wetland retained 305 $kg \cdot ha^{-1} \cdot y^{-1}$ of sediment, 0.5 $kg \cdot ha^{-1} \cdot y^{-1}$ of total phosphorus, 1.3 $kg \cdot ha^{-1} \cdot y^{-1}$ of total nitrogen, and 17 $kg \cdot ha^{-1} \cdot y^{-1}$ of organic carbon. With an estimated payback time of eight years, it proves to be a cost-effective solution for mitigating road runoff, suitable for catchment-scale adoption.

Limited studies have explored the treatment wetlands' ability to mitigate nitrogen pollution in the tropics. Kavehei *et al.* (2021) show that they exhibit optimal nitrogen removal rates with NO_3-N > 0.4 $mg \cdot dm^{-3}$ and slow water flows. Adequately managed treatment wetlands in tropical regions can achieve high removal rates for nitrogen and other pollutants.

HIGH-PERFORMANCE BARRIERS AND BIOREACTORS

Compared to current solutions, denitrification barriers represent a promising and highly effective emerging technology that requires further in-depth research and evaluation, as they offer a low-cost, flexible approach and can be combined with other ecological methods to enhance nitrate removal efficiency (Bednarek *et al.*, 2010; Bednarek, Szklarek and Zalewski, 2014; Zalewski, 2014). A denitrification bioreactor uses a solid carbon substrate, typically fragmented wood, to treat contaminated water by providing carbon and energy for denitrification, converting nitrate (NO_3^-) into nitrogen gas. Key types include denitrification walls (intercepting shallow groundwater), denitrification beds (intercepting concentrated discharges) and denitrification layers (intercepting soil leachate); NO_3^- removal rates range from 0.01 to 3.6 g $N \cdot m^{-3}$ per day for walls and from 2 to 22 g $N \cdot m^{-3}$ per day for beds (Schipper *et al.*, 2010). According to a study by Schipper and McGill (2008), denitrification barriers, incorporating 100 mm layers of organic matter beneath the topsoil in a dairy factory effluent irrigated site, resulted in reduced total nitrogen leaching. Control plots leached 296 kg N·ha^{-1}, whereas plots with denitrification layers leached 238 kg N·ha^{-1}, a total of 798 kg N·ha^{-1} was applied in effluent. Over 50% of the N leached to a depth of 40 cm was in the form of organic nitrogen, likely as a result of bypass flow. Denitrification occurred in a 100 mm organic layer receiving effluent but was insufficient to significantly reduce nitrate leaching in a large-scale land treatment system, likely due to the short leachate residence time. Studies suggest that thicker layers (300–500 mm) are more effective in removing nitrate beneath septic tank drainage fields. Another study showed that using waste cellulose solids (wood mulch, sawdust, leaf compost) as a carbon source, the reactive media ranged from 15% to 100% volume. These trials effectively reduced influent NO_3^- concentrations by 58–91% (Robertson *et al.*, 2005a). Further research involved laboratory tests on coarse wood particle media (woodchips), including fresh samples and those used for 2 and 7 years in subsurface denitrifying bioreactors. The 7-year-old media demonstrated a mean NO_3-N removal rate of 9.1 mg N·dm^{-3}·d^{-1}, 75% for the 2-year-old media (12.1 mg N·dm^{-3}·d^{-1}) (Robertson, 2010). In turn, another study showed that within two days, nitrogen levels in the treatment stream dropped from 6.7 to 3.9 mg·dm^{-3} with no change in the control area. Covering only 10–11% of the site edge, denitrification walls treated about 60% of the flow and significantly reduced nitrogen loads in adjacent waters (Schmidt and Clark, 2012).

In another study, Robertson and Merkley (2009) investigated a new in-stream bioreactor using wood chips as a carbon source for denitrification. Their findings indicated that nitrate mass removal typically increased with higher flow rates. When nitrate availability was not a limiting factor, the areal mass removal varied from 11 mg N·m^{-2}·h^{-1} at 3°C to 220 mg N·m^{-2}·h^{-1} at 14°C in agricultural drainage systems. Woodchip denitrifying bioreactors (WDBs) are an edge-of-field nitrate mitigation method increasingly used as permeable reactive barriers (Schmidt and Clark, 2012; Christianson and Schipper, 2016; Manca *et al.*, 2021).

Similarly, Addy *et al.* (2016) conducted an analysis using data from existing research to evaluate the nitrate removal efficacy of various denitrifying woodchip bioreactors, including walls, beds, and laboratory columns. Their findings indicated that the nitrate removal rates in bed and column studies were comparable and both were significantly higher than those observed in wall studies. Additionally, the type of wood used in denitrifying beds did not have a significant impact on nitrate removal rates. However, bed temperature was found to significantly influence the nitrate removal capacity and efficiency.

It is widely agreed that engineering technologies such as denitrifying wood chip bioreactors exist to intercept and remove nitrogen from agricultural runoff (Schipper *et al.*, 2010; Christianson and Schipper, 2016; Lepine *et al.*, 2018; Dougherty *et al.*, 2020; Audet *et al.*, 2021) and Aalto *et al.* (2022) mentioned that denitrifying bioreactors are designed to speed up the microbial process of denitrification. Plenty of evidence supports bioreactors' efficacy in enhancing runoff water quality and reducing nitrogen in humid tropic runoff (Manca *et al.*, 2021; Wegscheidl, Robinson and Manca, 2022).

A wide bioreactor with baffles can reduce N loss by 22–24% (Dougherty *et al.*, 2020). Oxygen depletion prompts rapid denitrification as water flows through the filter, yet their efficacy varies greatly between systems and seasons. Audet *et al.* (2021) investigated eight subsurface flow bioreactors with varying flow designs, observing N removal rates ranging widely from 17 to 73%. The study indicates that nitrogen removal efficiency is influenced by factors such as hydraulic residence time and water temperature. Denitrification bioreactors efficiently treat nitrate–nitrogen in various water matrices. For example, in the U.S. Midwest, subsurface-drained bioreactors remove an estimated 20–40% of annual NO_3-N losses (Christianson *et al.*, 2021).

An in-drain denitrifying wood chip bioreactor has the potential to reduce NO_3-N concentrations by 41% when evaluated between agricultural fields and downstream aquatic ecosystems (Cheesman *et al.*, 2023). As reported by Perera *et al.* (2024), the denitrifying woodchip bioreactor (DBR) initially exhibited high inferred phosphorus removal rates (75–100%) in season 1, which decreased notably later (around 3–67%). A pilot-scale DBR located in Tatuanui (New Zealand) was studied over two seasons: 2017 and 2018.

A highly permeable wood particle layer has been successfully tested for nitrate remediation in a shallow sand and gravel aquifer, reducing nitrate levels from 1.3–14 mg·dm^{-3} to less than 0.5 mg·dm^{-3}. The high-K reactive media allow for effective pollutant removal via permeable reaction barriers installed horizontally at the shallow water table without penetrating the entire depth of contaminant plume (Robertson *et al.*, 2005b). A study by Gibert *et al.* (2008) focused on the selection of organic substrates as potential reactive materials for use in a denitrification permeable reactive barrier, demonstrated significant reductions in TN across all tested substrates. Notably, TN removal rates exceeded 90% in softwood, ranged between 80–90% for hardwood, coniferous, compost, leaves, and mixture, and were slightly lower, at 70–80%, for mulch and soil, with willow showing removal rates below 70%. The efficiency of nitrate removal varied according to substrate type, with denitrification percentages surpassing 98% for softwood, hardwood, and mixture. Other materials, such as coniferous (95%), leaves (94%), and compost (93%), also achieved high denitrification rates. Moderately lower denitrification rates were observed for mulch (89%), willow (86%), and soil (73%), underscoring the role of substrate selection in optimising denitrification outcomes within permeable reactive barriers.

Barriers are used not only to capture nitrogen compounds but also phosphorus. Permeable reactive barriers (PRBs) offer an *in situ* solution for removing phosphate from agricultural runoff. Bus, Karczmarczyk and Baryła (2019) examined the potential of PRBs as an *in situ* method for removing phosphates from agricultural runoff. Testing four reactive materials (autoclaved aerated concrete, Polonite®, zeolite, and limestone) revealed phosphate removal rates ranging from 65 to 99%. Variations in performance may stem from climatic factors, phosphate discharge load, contact duration between the reactive material and treated fluid, and hydraulic conditions (Bus, Karczmarczyk and Baryła, 2019).

STORMWATER MANAGEMENT ON AGRICULTURAL LAND

Stormwater runoff, laden with pollutants, significantly contributes to groundwater contamination and surface water quality deterioration, necessitating effective management practices. Rain gardens, also referred to as bioretention systems or green infrastructures, with their physico-chemical and biological features, play a crucial role in remediating contaminants, storing runoff, reducing peak-flow rates, facilitating nutrient cycling, sequestering heavy metals, and providing supplementary benefits such as recreational amenities. Land use changes alter natural flow regimes, leading to increased stormwater runoff volume and peak flows, while reducing post-storm peak flow durations. This heightens the risks of flooding, erosion, and elevated pollutant loading in waterways. Rain gardens and stormwater harvesting systems emerge as viable methods to manage stormwater runoff, offering sustainable and economical solutions to decrease water volume flowing into waterways from impervious areas during storms (Schlea, 2011; Malaviya, Sharma and Sharma, 2019; Umukiza *et al.*, 2023). Retention ponds serve multifaceted roles in stormwater management, including peak runoff reduction, sediment trapping, erosion prevention, and water quality improvement. Additionally, they address agricultural runoff concerns. However, despite their efficacy, challenges persist, such as limited organic nitrogen removal efficiency (20–40%) and elevated total phosphorus concentrations (1 mg·dm^{-3} at the inflow) compared to similar studies (Rushton and Bahk, 2001).

Research results indicate that stormwater harvesting efficiently reduces farm field runoff and decreases rapid water flow to rivers during severe rainstorms (Verbist *et al.*, 2009). Besides, Sample and Liu (2014) also highlight the benefits of stormwater harvesting, which primarily serves as a water conservation measure. Its implementation also yields water quality advantages by mitigating runoff volume. Moreover, according to Tamagnone, Comino and Rosso (2020), stormwater harvesting techniques increase basin's water-holding capacity, reducing runoff and mitigating downstream flood risks. They also decrease sediment transport, preventing topsoil erosion and nutrient loss.

Since the 1960s, the term small retention has been coined and utilised exclusively in Poland, denoting diverse human interventions aimed at mitigating rapid water runoff following snowmelt and heavy rainfall events. It is posited that retained water during periods of surplus can replenish watercourses during drier seasons, thereby enhancing water availability for agricultural purposes and fostering biodiversity in rural landscapes. Various techniques for water retention have been delineated, including the construction of reservoirs or dams on rivers and lakes to augment the potential retention capacity of surface waters (Mioduszewski, 2014). In recent years, alongside the term "sustainable urban drainage systems" (SUDS) used for urban solutions, the term "rural sustainable drainage systems" (RSuDS) has also emerged (Dąbrowska, Dąbek and Lejcuś, 2018; Robotham *et al.*, 2021).

Ponds and wetlands exhibit potential for ameliorating stream water quality degradation caused by diffuse agricultural pollution by retaining dissolved nitrate, soluble reactive phosphorus, and suspended solids, particularly during baseflows. They also effectively reduce peak concentrations and loads of suspended solids and phosphorus during small to moderate storm events, with superior filtration of larger particles. An example of such a solution is presented in the work by Robotham *et al.* (2021), where on-line ponds are used to intercept pollutants like sediment, nutrients, and pesticides in watercourses. These features, known as constructed wetlands, retention ponds, or RSuDS, vary in design. In the predominantly arable Littlestock Brook sub-catchment (16.3 km^2) within the Evenlode catchment in southern England, the study shows that the effectiveness of on-line ponds in reducing diffuse agricultural pollution on clay soils with a 2.5% slope varies significantly, depending on sediment and nutrient retention capacities under different hydrological conditions. The ponds were estimated to have trapped 7.6% of the suspended sediment flux exiting the 340 ha catchment during the study period. Research underscores the intricate dynamics of pollutant retention and highlights the impact of event timing and magnitude (Robotham *et al.*, 2021).

GOOD AGRICULTURAL PRACTICES

To address nitrate pollution, the European Union has promoted a Good agricultural practices (GAP) code to improve the chemical and ecological status of waterbodies. Ilic *et al.* (2012) suggest that the decrease in contamination risks is directly associated with the implementation of GAP. While the effectiveness of GAP has been tested at different scales, at the basin scale, there is still very limited research and few study topics (Schnebelen *et al.*, 2004). Moreover, the impacts of GAP remain uncertain because studies indicate that leached nitrate from soil can take decades to reach surface water due to its storage and potentially lengthy travel time through unsaturated and saturated zones. However, this delay is often overlooked in current nitrate management and policy formulation regarding water resources (Wang *et al.*, 2013).

Furthermore, many agri-environmental measures emphasise method compliance over results, which does not motivate farmers (Sabatier, Doyen and Tichit, 2012). GAP regulations have been criticised for hindering farmer initiatives and involving cumbersome control procedures. Action plans are often delayed and focus on strict compliance with preset measures, resulting in minimal practice changes. This approach generally fails to adequately protect or restore water quality (Chantre *et al.*, 2016).

Schnebelen *et al.* (2004) employed a technique to assess the efficacy of GAP implemented in one region over seven years. Simulation outcomes indicated a decrease in nitrate concentration by approximately 30% (36 mg NO_3^-·dm^{-3}). Nevertheless, the rate of nitrate leaching remains excessive, necessitating additional enhancements to agricultural practices. Beaudoin *et al.* (2021) emphasised that implementing GAP can yield enduring benefits for water quality at a low cost–effectiveness ratio. There is merit in integrating GAP as the initial stage in transitioning towards agro-ecological systems. Additionally, Beaudoin *et al.* (2021) hypothesised that systematically applying GAP over time and space could achieve the EU nitrate concentration standard (50 mg NO_3^-·dm^{-3}) in arable cropping systems. GAP management primarily involved adjusting nitrogen fertilisation rates and establishing catch crops. Over 22 years, water and nitrogen fluxes were monitored in a 187 ha agricultural catchment. The observed nitrate concentration in the main spring declined and stabilised at 49 mg NO_3^-·dm^{-3} (Beaudoin *et al.*, 2021) 11 years after GAP implementation. The findings indicated that farmers were embracing various GAPs, including changes in cropping methods, integrating livestock, managing soil fertility, and adopting integrated pest management strategies. With the implementation of these practices, farmers decreased their usage of agrochemicals by over 40%. Key motivating factors for farmers to adopt GAP included enhancing soil health and minimising agrochemical usage (Kharel, Raut and Dahal, 2023).

Minimising nutrient loss, and nutrient leaching from croplands is crucial to mitigating nonpoint source pollution, preserving soil fertility, and enhancing soil and water quality. Implementing cover crops can be an effective strategy to tackle these issues. Restovich, Andriulo and Portela (2012) demonstrated that cover crops reduce soil nitrate levels by 50–90% by the time they are terminated, compared to soil that was left fallow. In another study, cover crops reduced nitrogen leaching by an average of 43%, but they did not significantly decrease total phosphorus losses from runoff and leaching. Winter freeze–thaw conditions heightened the risk of dissolved phosphorus loss from cover crop biomass (Aronsson *et al.*, 2016).

Combining multiple practices to mitigate soil phosphorus loss may improve reduction efficiency, though it is less studied than individual methods. Cover crops and drainage water management (DWM) decreased particulate phosphorus in runoff by 26% and total phosphorus by 12% (Zhang *et al.*, 2017). Aronsson *et al.* (2016) also especially emphasised that cover crops are mandatory in Denmark and subsidised in Norway, Sweden, and Finland, but farmer interest is limited. There is potential for wider use, but effective implementation strategies must be developed. More research is needed, especially on the impact of cover crops on phosphorus losses, including species differences and biomass harvesting effects. Grass cover crops like rye (*Secale cereale* L.) can reduce nitrate leaching by 18 to 95%. The effectiveness of cover crops follows this order: nitrate leaching ≥ sediment > runoff > dissolved nutrients in runoff. This suggests that cover crops are very effective at reducing nutrient leaching but have limited impact on reducing the transport of dissolved nutrients (Blanco-Canqui, 2018).

Despite efforts, nitrogen export from tile-drained agricultural watersheds continues. Effective agricultural conservation practices can reduce nitrogen loss at the field level, but their impact on watershed-scale reductions is less understood. In one study, planting cover crops on over 60% of croppable acres in a small agricultural watershed resulted in median nitrate-nitrogen losses from tiles draining fields with cover crops being 69–90% lower during winter/spring compared to those without cover crops (Hanrahan *et al.*, 2018). These findings indicate that frost exposure should influence cover crop selection in cold regions, while in temperate regions with snow cover insulating the soil, P release from vegetation may not significantly increase P runoff (Cober *et al.*, 2019). Cover crops improve water and nutrient absorption during otherwise fallow periods, reducing N and P losses and their transport to water bodies. They decrease N losses from subsurface drainage by about 50% but have variable effects on P loss. Surface N and P loss, which is less than subsurface loss, generally decreases with cover crops (Hanrahan *et al.*, 2021).

MANAGING AGRICULTURAL NONPOINT SOURCE POLLUTION: KEY ISSUES, CHALLENGES, AND OPPORTUNITIES FOR SURFACE WATER PROTECTION

CLIMATE CHANGE

Climate change has a significant impact on the transport of pollutants from agricultural land to water and on the effectiveness of constructed wetlands or riparian buffer zones. Constructed wetlands or riparian buffer zones alone provide multiple benefits for the adaptation to climate change (Shepherd *et al.*, 2011; Dąbrowska, Dąbek and Lejcuś, 2018; Varma *et al.*, 2021; Graziano, Deguire and Surasinghe, 2022; Jamion *et al.*, 2024). Increasing temperatures, changes in the distribution and intensity of precipitation, as well as longer periods of drought, affect the processes of release and transport of agricultural pollutants as well as their treatment. Heavy rains affect the increased leaching of fertilisers, pesticides and suspended solids. Prolonged periods of drought make the soil more vulnerable to erosion in the event of sudden rainfall, and any plant-based runoff reduction and water treatment methods are less effective after periods of water shortages. Drought has a negative impact on both above- and below-ground plant parts, but also on the concentration of pollutants, as reduced water flow may decrease the dilution of pollutants in watercourses. Higher temperatures, in turn, lead to an intensification of chemical and biological processes in the soil. The result is an increased mineralisation of fertilisers, which may lead to more rapid release of pollutants into waterways. Greater evaporation, in contrast, increases the concentration of pollutants in limited water resources. Increased consumption of chemicals in agriculture is also forecast under global change (Falloon and Betts, 2010; Brevik, 2013; IPCC, 2021; Seleiman *et al.*, 2021; Eekhout and Vente de, 2022; Hader *et al.*, 2022; Li *et al.*, 2023b). Numerous studies, i.e. Rolighed *et al.* (2016), prove that, in the face of projected climate change, a reduction in the load of pollutants entering water bodies is needed to maintain the current, albeit not very good, water status. Decisive action is needed to improve water quality.

Besides the fact that constructed wetlands or riparian buffer zones must be designed according to climatic conditions and pollutant loading (Syversen, 2005; Luke *et al.*, 2019; Jiang *et al.*, 2020; Nan *et al.*, 2023), new approaches to design and perhaps retrofitting parts of existing buffer zones and constructed wetlands in the face of climate change are needed (Ghimire *et al.*, 2021). Changes are necessary in the design and management of buffer zones so that they can function more effectively under conditions of increased rainfall, temperature, drought and a changing climate. This involves, for example, the introduction of more resistant plant species or additional elements of these solutions. So far, there is a scarcity of scientific articles in this area; only single papers can be found, e.g. Ghimire *et al.* (2021).

MICROPLASTICS AND NANOPLASTICS

Microplastics and nanoplastics in agricultural runoff are becoming a growing environmental concern. In agriculture, microplastics and nanoplastics can come from a variety of sources, such as plastic greenhouses and small tunnels, plastic packaging, protective nets, twine and stretch films intended to wrap straw, irrigation pipes and drippers, mulching with plastic sheeting, compost-based soil remediation, biosolid-amended croplands, sewage sludge, irrigation, using plastic carriers in seed coatings, coated fertilisers and pesticides, as well as from atmospheric deposition. Microplastics and nanoplastics are transferred from agricultural land to surface and groundwater through surface runoff, especially during heavy rainfall events (Lwanga *et al.*, 2022; Astner *et al.*, 2023; Naderi Beni *et al.*, 2023; Quilliam *et al.*, 2023; Hoang *et al.*, 2024).

When microplastics and nanoplastics reach waters, they may have a negative impact on aquatic ecosystems, disrupting biological processes in aquatic organisms and also accumulating in the food chain. Additionally, they can bind other chemical pollutants such as pesticides and heavy metals, increasing their toxicity (Rodríguez-Cruz *et al*., 2023; Lin *et al*., 2024; Oliveri Conti, Rapisarda and Ferrante, 2024). This issue becomes relevant in the context of the aforementioned climate change. Studies are already appearing on the effectiveness of plastic litter trapping by constructed wetlands or riparian buffers (Cesarini and Scalici, 2022; Zhang *et al*., 2024a).

PROBLEMS WITH IMPLEMENTATION OF POLICIES, STRATEGIES AND TECHNICAL SOLUTIONS

The problem with the effective implementation of policies, strategies and technical solutions has been observed worldwide for years. In most countries, the expected improvements in surface water quality have not been achieved (Luke *et al*., 2019; Wiering, Boezeman and Crabbé, 2020; Graziano, Deguire and Surasinghe, 2022; Wiering, Kirschke and Akif, 2023; McDowell *et al*., 2024).

The European Green Deal imposes strict requirements on farmers. The goal is to reduce pesticide use by 50% and nutrient losses by 50% while maintaining soil fertility, cut the use of mineral fertilisers by at least 20% by 2030, and promote organic farming to cover 25% of all arable land in the EU (Pańka *et al*., 2021). However, in 2024, farmers across Europe (in Poland, Czechia, Belgium, the Netherlands, Germany, and France, among others) staged protests against the European Green Deal, objecting to the proposed environmental regulations. Czech farmers demanded that the government withdraw from the European Green Deal, citing high energy costs, among others. In Poland, farmers pushed back against EU climate policies, particularly those reducing pesticide usage. In response to mounting pressure, the European Commission has made some concessions, such as pausing a bill that would halve pesticide use by 2030. However, dissatisfaction continues as farmers claim that these policies are being imposed without adequate support to help them transition to more sustainable practices. Surveys in Poland show that farmers often oppose the so-called "greening" of agriculture, which involves additional costs, and the available subsidies are insufficient to compensate for these changes. As a result, farmers may prioritise cost-minimising strategies, which can conflict with environmental goals (Świtek and Sawinska, 2017).

The limited acceptance of environmental regulations among farmers may be associated with a broader pattern of climate scepticism. Studies show that farmers are generally more sceptical of climate change impacts than non-farmers, and that reducing this scepticism is crucial for the effective implementation of mitigation and adaptation measures. At the same time, research indicates that even in the absence of strong belief in anthropogenic climate change, farmers often adopt practices with climate-mitigative effects – motivated primarily by expectations of economic benefits, improved soil quality, and biodiversity rather than environmental concern (Davidson *et al*., 2019; Kröner *et al*., 2025).

Additionally, the Russian invasion of Ukraine highlighted the fragility of unsustainable, input-intensive food systems, prompting a market crisis that threatened global food security and led to a rapid reversal of recent environmental progress (Cuadros-Casanova *et al*., 2023). During protests in Chechia and Poland, farmers opposed agricultural imports from Ukraine as additional threats to local production and thus their financial viability.

MODERN REMOTE SENSING TECHNIQUES AND GIS

Modern remote sensing techniques and Geographic Information Systems (GIS) support research on water transfer and protection against agricultural pollution in several key areas: identifying areas within a catchment that are susceptible to generating runoff, known as hydrologically sensitive areas (HSAs); determining critical source areas (CSAs) with the highest risk of pollutant transfer, where potential pollutant loading is linked to susceptibility to runoff (HSA combined with high pollutant mobilisation risk); pinpointing interfaces between pollution source areas and buffer zones or surface waters; simulating the impact of various agricultural practices on water quality and optimising water resource management, including the establishment of effective buffer zones and the support of nonpoint source pollution models (i.e., agricultural nonpoint source pollution model (AGNPS), annualised agricultural nonpoint source pollution model (AnnAGNPS), areal nonpoint source watershed environment response simulation model (ANSWERS), soil and water assessment tool model (SWAT), hydrologic simulation program FORTRAN (HSPF), chemicals, runoff, and erosion from agricultural management systems model (CREAMS) (Xiang, 1996; Dąbrowska, Dąbek and Lejcuś, 2018; Yuan, Sinshaw and Forshay, 2020; Robotham *et al*., 2021; Eishoeei, Miryaghoubzadeh and Shahedi, 2022). These methods are based on light detection and ranging based high-resolution digital elevation models (LiDAR DEMs), satellite imagery, i.e. Landsat, Sentinel-2, analysis of spectral indices such as e.g. normalised difference vegetation index (NDVI), normalised difference water index (NDWI) (Dąbrowska, Dąbek and Lejcuś, 2018; Jaskuła, Sojka and Wicher-Dysarz, 2019; Eishoeei, Miryaghoubzadeh, and Shahedi, 2022; Mykrä *et al*., 2023; Zhang *et al*., 2024b). High-resolution Digital Terrain Models (DTMs), particularly those derived from LiDAR data, are essential for identifying surface runoff pathways and zones of flow accumulation in agricultural landscapes. When combined with detailed land cover data, they enable accurate assessment of areas most vulnerable to nonpoint source pollution, especially where arable land, roads, and sparse vegetation interact to facilitate runoff. This integrated approach supports spatially targeted mitigation measures by identifying critical source areas and gaps in natural buffer systems (Dąbrowska, Dąbek and Lejcuś, 2018). Using satellite and aerial remote sensing (hyperspectral and multispectral imaging) and GIS allows for more precise and comprehensive information, leading to better environmental management and conservation of natural resources. These solutions allow analysis of changes over time and space, accurate modelling – the integration of remote sensing data with GIS enables the creation of advanced models of pollution spread that take into account many factors, e.g. topographical, hydrological and meteorological factors, as well as the creation of interactive risk maps, and the reduction of costs associated with the creation of traditional monitoring networks.

CONCLUSIONS

Effective reduction of nonpoint source pollution from agricultural areas to surface waters requires integrated, system-based approaches that combine technical, nature-based, and organisational measures. Key barriers include fragmented policy implementation, insufficient financial incentives, limited farmer engagement, and the challenge of adapting mitigation strategies to diverse environmental and climatic conditions. Enabling factors involve aligning subsidies with environmental objectives, involving land users in the co-design of measures, and applying geospatial tools for targeted planning and monitoring. Research priorities should focus on the long-term effectiveness of combined interventions, management of emerging pollutants (e.g., microplastics), and the climate resilience of nature-based solutions. Recommended actions include shifting from reactive to preventive measures, improving the targeting of agri-environmental support, and promoting decision-support systems based on satellite data and spatial modelling. Only an integrated and adaptive strategy, addressing environmental, technical, and socio-economic dimensions, can ensure meaningful reductions in agricultural water pollution and support the achievement of EU policy goals and Sustainable Development Goal 6.

CONFLICT OF INTERESTS

All authors declare that they have no conflict of interest.

DECLARATION OF GENERATIVE AI AND AI-ASSISTED TECHNOLOGIES

During the preparation of this work, the authors used OpenAI service in order to improve the readability and language of the manuscript. After using this service, the authors reviewed and edited the content as needed and take full responsibility for the content of the publication.

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