# Targeted Reduction of Phosphorus Losses from Agriculture to Surface Waters Remains a Current Topic

Radoslav BUJNOVSKÝ[1] (✉)
Štefan KOCO[2,3]

## Summary

Phosphorus (P) losses from agricultural land contribute significantly to surface water eutrophication. The implementation of appropriate agricultural practices offers substantial potential for mitigating this impact. Effective mitigation requires both source-based and transport-based strategies. The source-based measures aim to regulate P inputs through fertilizers and prevent excessive P accumulation in the soil or gradually reduce/deplete high soil test P (STP) levels. Current fertilization schemes, which are primarily based on production limits may not be sufficient in terms of water protection. Since STP level is not a reliable indicator of dissolved reactive phosphorus (DRP) losses, the degree of soil P sorption saturation (DPS) is often proposed as an environmental indicator of soil P release into surface runoff or seepage water. Transport-based measures target the mitigation of particulate P (PP) and/or water-soluble P (WSP). While cover crops has been found to be a useful practice of mitigating both particulate and dissolved phosphorus losses, zero tillage has been demonstrated to be an effective method in reducing PP losses. Frequently recommended measures also include vegetated buffer strips at the edge/outside of the field and application of reactive media help mitigate losses of PP or WSP. From the perspective of increasing the environmental efficiency and cost-effectiveness of the measures taken, their allocation should focus on hot spots that consider source and transport factors. Understanding the distribution of P in loss pathways is essential for selecting effective measures in terms of response time and reduction efficiency.

## Key words

agriculture, phosphorus, surface waters, eutrophication, farm measures

[1] Water Research Institute, Nábr. arm. gen. L. Svobodu 5, 812 49 Bratislava, Slovakia

[2] National Agricultural and Food Centre – Soil Science and Conservation Research Institute, Trenčianska 55, 82109 Bratislava, Slovakia

[3] University of Presov, Faculty of Humanities and Natural Sciences, University of Presov, ul. 17. Novembra 1, 08001 Prešov, Slovakia

✉ Corresponding author: radoslav.bujnovsky@vuvh.sk

Received: May 5, 2025 | Accepted: September 24, 2025 | Online first version published: October 27, 2025

## Environmental consequence of long-term intensive use of phosphorus in agriculture

Phosphorus (P), one of the most important nutrients for living organisms to function, plays a key role in ensuring food production (Johnston, Dawson, 2005; Faucon et al., 2015; Khan et al., 2023). Following World War II, intensified crop cultivation led to significant increases in P application (Johnston, Dawson, 2005; Johnston et al., 2014; Bonsdorff, 2021). In Europe and U.S. this process culminated around the 1980s (Demay et al., 2023).

The production aspects of phosphorus use have been at the centre of public interest for a long time, with little attention being paid to its impact on water quality (Leinweber et al., 2002; Steinfurth et al., 2022). The consequence of P application to ensure crop yields has often been the formation of excessive phosphorus reserves in the soil. A common argument to justify P fertilization was the good sorption of applied P in the soil, in contrast to nitrogen.

Increased attention to the impact of agriculture, and phosphorus in particular, on water quality began to be paid after 1990 (e.g. Breeuwsma and Silva, 1992; Sharpley et al., 1994; Carpenter et al., 1998; Leinweber et al., 2002). Phosphorus losses from agricultural land to surface waters represent a global environmental problem due to the significant role of this nutrient in surface waters eutrophication and its manifestations – the development of algae and cyanobacteria (Smith et al., 1999; Ulén et al., 2007; Conley et al., 2009; Withers et al., 2014a; Schindler et al., 2016). Despite of generally lower intensity of crop and animal production compared to Western Europe countries/regions, this issue is also relevant for Central and Eastern Europe (CEE) countries (ICPDR, 2021b).

When reassessing phosphorus management for environmental (especially water) issues, similar to the nitrogen case, the thesis "Feed the crop rather than the soil" begins to be emphasized (Withers et al., 2014b; Duncan et al., 2017). It is therefore not about putting phosphorus at the level of nitrogen, but emphasizing the environmental aspects of its use.

Achieving good ecological status in surface waters requires both immediate and long-term reductions in phosphorus losses and efficiency improvement its application (Carpenter et al., 1998; EEA, 2018; ICPR, 2020; Johnes et al., 2022; EEA, 2024).

At the European Union level, the adoption/implementation of the Nitrates Directive 91/676/EEC and subsequently the Water Framework Directive 2000/60/EC represents a fundamental milestone in addressing the problem of water pollution by nutrients. Despite ongoing and increasing efforts (especially in the last two decades), progress has been slow, as documented by water monitoring results at both national and international levels (EEA, 2018; Cibulka et al., 2020; ICPDR, 2021b; Withers et al., 2024). It can be said that the achieved results have not always met the expectations of policy makers. Making food systems fair, healthy and environmentally-friendly through the reduction of nutrient losses by 50% and fertilizer use by 20% by 2030, as defined in the Farm to Fork Strategy (European Commission, 2020), can be seen as a further impetus to improve water status.

Nutrient inputs reduction in agriculture is usually achieved either through extensification or sustainable intensification. A typical example of (forced) extensification is the reduction of agricultural intensity in CEE countries during the early 1990s, consequent to the collapse of the centrally planned economies of Central and Eastern Europe (Redman et al., 2004). Consequently, in many of these countries, diffuse pollution of waters by phosphorus from agriculture has received little attention. It was assumed that the reduction in the use of mineral phosphorus fertilizers and animal manure after 1990, as reported by (Tóth et al., 2014; Van Dijk et al., 2016), would be sufficient to reduce the contributions of agriculture to surface water eutrophication.

Sustainable intensification is a concept that aims to balance agricultural production with environment protection (Buckwell et al., 2014; ICPDR, 2021a; European Commission, 2025). Although several concerns have been raised about the concept above – its complexity and practical applicability (e.g. Struik and Kuyper, 2017; Mahon et al., 2018; Helfenstein et al., 2020), – both the build-up approach and de-intensification are fully relevant justified for this concept as well. The issue of sustainable de-intensification is particularly justified in Eastern Europe, which is among the regions with the highest rates of phosphorus depletion. This is largely, due to lower affordability of mineral fertilizers and inefficient management of organic phosphorus sources (Allewell et al., 2020).

## Commonly recommended measures at farmland

From the perspective of reducing of water pollution caused by nutrients from agriculture, the farm level represents the basic spatial unit for implementing the relevant measures. The input of phosphorus from land to surface waters is determined by both, the source/supply of P and transport processes. In this context, measures can be classified as either source-oriented or transport-oriented (Schoumans et al., 2014; Collins et al., 2018; Osmond et al., 2019; Singh et al., 2020; McDowell and Haygarth, 2024).

### Source-based measures

4R nutrient stewardship, with respect to right source, right rate, right time and right place of nutrient application, can be considered a basic framework for increasing nutrient use efficiency and reducing their losses (Johnston and Bruulsema, 2014).

Determining the right P fertilizer rate is typically based on fertilization schemes that consider soil test P (STP) levels and crop requirements (Jordan-Meille et al., 2012; Collins et al., 2018; Singh et al., 2020). To prevent phosphorus accumulation in hydrologically active and connected soils, it is recommended to use environmental thresholds (if available) instead of agronomic thresholds when calculating P fertilizer rates (Kleinman, 2017).

To reduce water-soluble phosphorus (WSP) losses, the ideal time to apply P in mineral fertilizers is usually just before or at sowing. Farmyard manure is usually applied after harvesting the previous crop – either in autumn if the following crop is sown in the same season, or later in autumn if spring crop follows. Liquid manure, also with regard to minimizing nitrogen losses, is suitable to be applied in the spring.

Sufficient and safe storage capacities for animal manure are a basic prerequisite for avoiding its application to fields in high-risk times (Collins et al., 2018; Bujnovský et al., 2023).

On sloped fields, in order to reduce P losses through surface runoff, it is advisable to apply P fertilizers subsurface or incorporate them into the soil after their application (Collins et al., 2018; Singh et al., 2020; Bujnovský et al., 2023). To regulate the release of P into the soil solution, it is necessary to take into account the solubility of P fertilizers. Within the range of common mineral fertilizers, P solubility increases in the following order: rock phosphates > superphosphates > mono-/di-ammonium phosphates (Johnston and Dawson, 2005).

For soils with high P sorption saturation (i.e. critical source areas of P loss within a watershed), long-term P phytoextraction approaches should be implemented (Delorme et al., 2000; Svanbäck et al., 2015; Kleinman, 2017), under the condition that no additional P inputs in mineral fertilizers or animal manure are applied. If information on the degree of soil P sorption saturation (DPS) is not available, STP values should be reduced to a medium level as a preliminary measure.

For mineral P fertilizers, this means halting their purchase, while for animal manure, it requires either transporting animal manure out of the farm/watershed area, or relocating/reducing of livestock numbers (Amery and Schoumans 2014) – both of which impact farmers' practices and incomes. Manure separation can reduce the costs of its transporting to nearby, less intensively farmed area (Schoumans et al., 2014). As Sharpley et al. (2015) emphasize, in highly vulnerable areas, even a set of best management practices may not be sufficient to maintain current levels of agricultural intensity. Nevertheless, the regulation of livestock numbers is determined by social preferences.

### Transport-based measures

These measures usually aim to reduce the transport of sediment-bound P (also referred to as particulate P – PP) via soil erosion and movement of water soluble P through water-runoff, which primarily includes surface and subsurface outflow.

Soil erosion often represents a significant pathway for total phosphorus (TP) transport to surface waters (e.g. Kronwang et al., 2007; Kovács et al., 2012; ICPDR, 2021b; Panagos et al., 2022), particularly on sloping land. Usually, most of eroded sediment is redeposited within the field, and only a smaller portion reaches surface waters. It most often ranges up to 30% (Tetzlaff et al., 2013; Borelli et al., 2018; Remund et al., 2021).

The most common and widely available in-field anti-erosion measures, used to reduce PP losses include zero and reduced tillage, contour ploughing, cover/catch crops, and grassed waterways (Sharpley et al., 2013; Singh et al., 2020; McDowell and Haygarth, 2024). Improving soil structure by applying manure or liming also helps to increase the soil erosion resistance (Aviles et al., 2020; Fu et al., 2022; Cui et al, 2023).

It is generally known that on sloping land, zero/minimum tillage reduces erosion (and the associated PP transport), but on the other hand, it increases the transport of WSP, which is much more effective from the view of water eutrophication (Duncan et al., 2017; Jarvie et al., 2017; Tattari et al., 2017). Therefore, additional measures with focus on P fertilizer placement – such as subsurface application or injection and incorporation of applied fertilizer into soil, or one-time inverse ploughing – need to be introduced to reduce surface runoff P losses (Bengström et al., 2015; Smith et al., 2016; Carver et al., 2022).

The purpose of growing cover/catch crops is to provide continuous "green" soil cover, thereby reducing both soil erosion and losses of particulate and water soluble P (Melland et al., 2012; Aronsson et al., 2016; Carver et al., 2022). Their efficiency depends on climatic conditions – either sufficient heat in cold areas or sufficient moisture in dry sub-humid areas (Vach et al., 2009; Sharpley et al., 2015; Aronsson et al., 2016; Nielsen et al., 2016; Bujnovský et al., 2023).

P trapping at the edge of the field is a common measure aimed primarily at reducing the input of PP into surface water. The most accessible farmland measure in this category is the establishment/maintenance of vegetated barriers, also known as riparian buffers, vegetated buffer strips or riparian field margins (Hickey and Doran, 2004; Dorioz et al., 2006; Collins et al., 2009; Stutter et al., 2012; Muñoz et al., 2024). These are usually (unfertilized) strips of the same crop along the field margin, and zones with different vegetation – usually outside the fields, including stream bank vegetation. However, riparian zones outside the farmland are usually inaccessible to farmers unless they own the relevant lands.

Efficiency of vegetated buffer strips is influenced by several factors such as buffer width, plant community composition, slope or input loads (Zhang et al., 1010; Jiang et al., 2020; Prosser et al., 2020; Dunn et al., 2022; Ramler and Strauss, 2024). Over time, however, P accumulated in these strips becomes a potential source of legacy P (Collins et al., 2009; Sharpley et al., 2013; Cole et al., 2020). As Hille et al. (2018) suggest, annual harvesting of biomass from vegetated strips help mittigate P leaching to the aquatic environment.

Reducing P mobilization or capturing transported P using chemicals or a reactive media as soil amendments represents potential options for mitigating WSP losses (Udeigwe et al., 2011; McDowell and Nash, 2012; Ulén and Etana, 2014; Uusitalo et al., 2015; Singh et al., 2020; Ekholm et al., 2024; Perry et al., 2024). The application of the relevant substances or by-products is usually carried out as part of soil treatment in the field, or the edge of the field and near the ditches/streams. However, applying chemicals in ditches connected to watercourses may be unavailable to farmers unless they own the ditches.

Before using any by-product to regulate the WSP mobility, it is essential to evaluate material costs, P sequestration/immobilization efficiency and environmental toxicity, related to soil quality and aquatic biota (Udeigwe et al., 2011; Iho and Laukkanen, 2012; McDowell and Nash, 2012; Rantamo et al., 2022; Perry et al., 2024). As Iho and Laukkanen (2012) point out, materials rich in aluminum (Al), calcium (Ca) and iron (Fe) are mainly suitable for a temporarily reducing of water-soluble P at high STP values and should not be considered as a substitute for long-term high phosphorus reserves depletion.

When liming acidic soils with high P sorption saturation, both pH requirements of the crops and the target pH value must be considered. In terms of reducing phosphorus solubility, the target pH should not exceed 6.5 (Kleinman, 2017).

To enhance environmental efficiency and cost-effectiveness of the implemented measures, their allocation should focus on hot spots or critical source areas (CSAs) that consider source and transport factors (White et al., 2009; Sharpley et al., 2011; Iho and Laukkanen, 2012; Kovács et al., 2012; Macintosh et al., 2018). The accuracy of hot-spot definition (Thomas et al., 2016) depends on the availability of high-resolution data. The spatial resolution of input variables for calculating P input to freshwater is increasingly important, especially at the local level (Dupas et al., 2015).

## Farm measures efficiency begins with their setting – the case of source-based measures

Preventing P losses to waters through source regulation is directly linked to the application of mineral fertilizers and animal manure. Although P balance is often debated when assessing the level of fertilization with this nutrient and environmental risks (Kronwang et al., 2007; Stubenrauch et al., 2018; Einarsson et al., 2020; Amery et al., 2021; Panagos et al., 2022; Muntwyler et al., 2024), STP indicates a long-term balance of this nutrient. The conversion of various STP to Olsen P is presented by Steinfurth et al. (2021). Nevertheless, the relationship between P balance and plant available P content determined by various STP methods (e.g. Reid et al., 2019; Muntwyler et al., 2024) is particularly important for estimating the effect of fertilization on reducing soil P reserves at zero or low input fertilization strategy where such approaches are justified (Tyson et al., 2020; Steinfurth et al., 2024).

Taking STP values into account within fertilization schemes is a fundamental step in calculating P rates. Since plant available P classification systems have traditionally been based on a production aspect (Bujnovský and Fotyma, 2001; Jordan-Meille et al., 2012; Steinfurth et al., 2022), current fertilization schemes may not always be appropriate from a water quality protection perspective.

In general, maintaining of agronomic adequate but environmentally low STP levels can serve as an initial step in minimizing P loses (Osmond et al., 2019; Nair et al., 2020). The starting point is the determination of agronomic and environmental thresholds. The agronomic STP threshold (also referred to as critical level of STP for optimum crop yield) corresponds to the P concentration needed to achieve 95% of the maximum expected yield. In contrast, the environmental STP threshold is defined as the inflection point above which both the soil P concentration and the potential of P loss through runoff and leaching increase (Bai et al., 2013; Wang et al., 2023; Gu et al., 2024).

While available P level below the agronomic threshold STP may reduce crop yields, exceeding the environmental STP threshold negatively affects water quality (Regan et al., 2010). This is due to the fact that the critical soil P concentrations for crop growth are in order of magnitude higher than P concentrations causing eutrophication of surface waters (Daniel et al., 1998). Regarding surface water quality, total phosphorus (TP) concentrations exceeding 0.05 $mg{\cdot}L^{-1}$ indicate anthropogenic influence (Kristensen, 1996), while a threshold of 0.1 $mg{\cdot}L^{-1}$ is commonly considered to support good ecological status for rivers and certain types of lakes (Poikane et al., 2019).

For illustration, average agronomic thresholds for Olsen STP range from 7 to 30 $mg.kg^{-1}$ (Withers et al., 2019; Steinfurth et al., 2022) and critical Mehlich-3 STP values range from 30 to 80 $mg.kg^{-1}$ (Sharpley et al., 2012; Withers et al., 2019). Recent assessments suggest that the majority of European cropland exceeds agronomic Olsen P thresholds for individual crop groups (McDowell et al., 2024).

Environmental STP thresholds either exceed the critical STP values for optimum yield (e.g. Regan et al., 2010; Bai et al., 2013; Xi et al., 2016; Johnston and Poulton, 2019; Steinfurth et al., 2022; Wang et al., 2023) or overlap with them. In this context, Johnston and Poulton (2019) state that phosphorus use efficiency is highest near the agronomic thresholds. Conversely, Withers et al. (2019) state that maintaining agronomic optimum levels may still pose a risk of eutrophication, and call for more research focused on agronomic STP thresholds.

Soil properties such as iron and aluminium (hydr)oxides, organic matter, pH, calcium content, and clay influence soil P sorption/desorption processes and, as a result, both agronomic and environmental STP thresholds (Lins and Cox, 1989; Penn et al., 2018; Tandy et al., 2021; Møller et al., 2023). While the environmental STP thresholds decrease in the order clay > loam > sandy soils, as a result of correlation between P adsorption capacity and clay content (Penn et al., 2006), agronomic STP thresholds have often show the opposite trend (Pöthig et al., 2010; Wang et al., 2023). Accordingly, agronomic STP thresholds on sandy soils exceed environmental thresholds first. Consequently, reducing soil P mobility or extensifying of crop cultivation should also be considered. This aligns with the view of Huang and Hartemink (2020), who emphasize that the production and environmental aspects of crop fertilization on sandy soils require increased attention, particularly under intensive use.

Updating STP critical values for yield optimization, as suggested by several authors (Withers et al., 2019; Steinfurth et al., 2022), can also help calibrate P fertility classes and revise fertilization schemes, directly contributing to improve phosphorus use efficiency and a reduction in environmental losses (Jordan-Meille et al., 2012; Rupp et al., 2018; Steinfurth et al., 2022; Wiesler et al., 2018). To avoid adverse impacts on crop yields, Muntwyler et al. (2024) propose targeting areas (on European scale) where Olsen STPs exceeding 40 $mg{\cdot}kg^{-1}$, with recommendation to reduce or suspend P inputs as a first step.

As stated by Sharpley et al. (2012) and Duncan et al. (2017), STP is a useful screening method for identifying fields at higher risk of P losses, but it is not a reliable predictor of dissolved reactive phosphorus (DRP) losses on its own. Therefore, the degree of soil P sorption saturation (DPS), calculated as the ratio of soil P concentration (determined by STP methods) to soil P sorption capacity (PSC), is often proposed as an environmental indicator of soil P release into surface runoff or seepage water (Breeuwsma and Silva, 1992; Nair et al., 2004; Sharpley et al., 2012; Fischer et al., 2017; van Doorn et al., 2024).

However, determination of PSC in commercial soil test laboratories and the subsequent DPS calculation for routine fertilization is not yet feasible in standard practice. Therefore, alternative methods of PSC and DPS prediction are being developed, primarily through the pedo-transfer functions (Pöthig et al., 2010; Fisher et al., 2017; Penn et al., 2018; Welikhe et al., 2020; Kedir et al., 2022), with subsequent update of STP categories for fertilization schemes (Fisher et al., 2017). Although there is an effort to predict and map parameters relevant to PSC detection (Møller et al., 2023), a more practical approach may lie in the modified method for PSC determination using Mehlich-3 extractant as a substitute for acid ammonium oxalate (Keinman and Sharpley, 2002). Other extraction reagents for detecting PSC are also being tested (Blombäck et al., 2021).

The combined measurement of STP, PSC, and DPS is also recommended for estimating high levels of available P in the soil (legacy soil P) and assessing its potential to become a source of P to surface and subsurface runoff (Rowe et al., 2016; Van Grinsven et al., 2016; Nair et al., 2020; Sharpley et al., 2020).

## Water protection goes beyond the farm – a source and time perspective

In order to effectively reduce P inputs to surface waters at farm and watershed level, it is necessary to understand the contribution of this nutrient from different sources and various transport pathways (Sharpley et al., 2015; Ta et al., 2020; Fuchs et al., 2022). The apportion of total P surplus across individual sectors and input pathways is therefore the starting point for setting measures to achieve target water quality. Without such data, reducing surface water pollution remains a matter of conjecture or biased opinion.

In terms of source, diffuse pollution from agriculture and municipal wastewater discharges are typically the most significant P contributors to surface water pollution and eutrophication (ICPDR, 2021b; Fuchs et al., 2022; Withers et al., 2024). There is growing support for shifting the focus from municipal measures to diffuse sources, especially agricultural land (Westphal et al., 2019; McDowell and Haygarth, 2024). Even though agriculture is often the main source of degradation of the state of the waterbody (EEA, 2018), mixed pollution sources are often involved, including sewage water overflows and emissions from industrial sites (Fuchs et al., 2022; Withers et al., 2024; Wuijts et al., 2024). Moreover, P discharges in municipal wastewater can significantly contribute to the pollution and eutrophication of surface water, particularly where advanced P removal from urban sewage is not applied (Jarvie et al., 2006; Carey and Migliaccio, 2009; Preisner et al., 2020), and where receiving water bodies have limited dilution capacity (Büttner et al., 2022) or ecologically significant times related to receiving (surface) waters (Melland et al., 2012) are not adequately considered.

Although P inputs to surface waters are commonly calculated and reported as TP (Malagó et al., 2015; ICPDR, 2021b; Tetzlaff et al., 2024), P bioavailability is an important aspect when assessing the significance of P input from various sources. From the perspective of surface water eutrophication, greater attention should be given to algal-available P (Sharpley, 1993; Uusitalo et al., 2003; Baker et al., 2014; Dupas et al., 2015; Trentman et al., 2021) or bio-available P which may serve as a reference parameter for evaluating impacts of P inputs to surface water.

Developing a sustainable agricultural system requires better understanding, a reassessment of traditional recommendations, and broader adoption of the most effective practices by farmers (Smith et al., 2018). Without sufficient research support and a clear understanding of why measures succeed or fail, there is a risk of misdirected efforts (Sharpley et al., 2015). In this context, better mapping and understanding the impact of P accumulated in the soil-water continuum in the past (P legacy) is often important for reducing P inputs to surface waters and their eutrophication. This applies to both primary sources (e.g. agricultural land) and secondary P sources, such as riparian zones and fluvial sediments outside farmland (Jarvie et al., 2013; Sharpley et al., 2013; Withers et al. 2014a; Powers et al., 2015; Rowe et al., 2016; Westphal et al., 2019). Secondary P sources may mask the effect of measures taken to reduce the input of this nutrient into the aquatic environment. Remobilization of river sediments poses a potential risk for the continuation or recovery of eutrophication and its manifestations in downstream water bodies (Jarvie et al., 2013; Westphal et al., 2019).

Reducing phosphorus input into surface waters requires action that goes beyond the agricultural sector (e.g. Withers et al., 2015; Helfenstein et al., 2020; Johnes et al., 2022). The creation of objective knowledge (Sharpley et al., 2015; Bol et al., 2018) and evidence, together with effective policies and legislation (Amery and Schoumans 2014; Johnes et al., 2022; Wuijts et al., 2024) is a fundamental prerequisite for achieving this goal.

## CRediT authorship contribution statement

**Radoslav Bujnovský:** Conceptualization, investigation (collection of information), supervision, writing original draft, writing – review & editing of the manuscript. **Štefan Koco:** Investigation (collection of information), and writing – review & editing of the manuscript.

## 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.

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