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PERSPECTIVE

Streambank erosion and phosphorus loading to surface waters: Knowns, unknowns, and implications for nutrient loss reduction research and policy

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Abstract

To monitor and meet water quality objectives, it is necessary to understand and quantify the contribution of nonpoint sources to total phosphorus (P) loading to surface waters. However, the contribution of streambank erosion to surface water P loads remains unclear and is typically unaccounted for in many nutrient loading assessments and policies. As a result, agricultural contributions of P are overestimated, and a potentially manageable nonpoint source of P is missed in strategies to reduce loads. In this perspective, we review and synthesize the results of a special symposium at the 2022 ASA-CSSA-SSSA annual meeting in Baltimore, MD, that focused on streambank erosion and its contributions to P loading of surface waters. Based on discussions among researchers and policy experts, we overview the knowns and unknowns, propose next steps to understand streambank erosion contribution to P export budgets, and discuss implications of the science of streambank erosion for policy and nutrient loss reduction strategies.

Abbreviations: BSTEM, Bank Stability and Toe Erosion Model; LiDAR, light detection and ranging; UAV, unmanned aerial vehicle.

1 INTRODUCTION

Core Ideas

- Streambank erosion is a commonly overlooked source of nonpoint phosphorus loads to surface waters.
- Not accounting for streambank erosion will overestimate agricultural contributions to phosphorus loads.
- Knowns and unknowns of streambank erosion contributions to phosphorus export are reviewed.

Globally, hypoxic zones of coastal and freshwater resources have increased exponentially since the 1960s (Diaz & Rosenberg, 2008; Jenny et al., 2016). To monitor and meet water quality objectives aimed at decreasing hypoxic zones from local to regional scales, it is necessary to understand the contribution of sources to total phosphorus (P) loading, which can be broadly categorized as point and nonpoint sources. Point sources are operationally defined as discrete, identifiable origins of P loads that can be monitored (e.g., wastewater treatment plants). In contrast, nonpoint sources are diffuse and generally more difficult to monitor (USEPA, 1998) due to their diffuse nature (Loehr, 1974) and temporal variability (e.g., event-based “flashiness” and interannual variation), largely driven by precipitation events (Goolsby et al., 2000; Sinha & Michalak, 2016). Within the nonpoint sources, agriculture has received relatively more attention and remedial action, including increasing pressure to reduce hypoxic zones such as in the Gulf of Mexico (Rabalais & Turner, 2019) or the Baltic Sea (Carstensen et al., 2014), and the creation of policies by legislation like the Clean Water Act in the US (Uttormark et al., 1974). Agricultural P losses originate from inputs such as fertilizer or manure as dissolved reactive P via horizontal (e.g., runoff) or vertical (e.g., tile drainage) losses (Wang et al., 2020), as well as from soil via erosion or organic matter mineralization (Barrows & Kilmer, 1963; Dinnes et al., 2002). While the relative importance and absolute magnitudes of P losses from agricultural fields vary by biophysical context (Hansen et al., 2002), particulate P loss by erosion from fields is the dominant pathway by which P transfers occur due to agricultural activities (Gentry et al., 2007; Sharpley et al., 1993).

However, there are contributions to nonpoint source P loads to surface waters that can be difficult to estimate (Loehr, 1974). A key and often overlooked nonpoint P source is the erosion of streambanks. As the name implies, streambank erosion is the loss of soil and associated material (e.g., stones and vegetation) from the bank adjacent to the stream (Fox et al., 2016). Streambank erosion is a naturally occurring process that can be exacerbated by human (especially agricultural) practices. Streambank erosion is "natural" in that it has always taken place, and is responsible for the formation of hills, valleys, marshes, islands, and other landforms around the world at geological timescales. The Grand Canyon, for example, is in part the result of streambank erosion on a massive scale, and the extensive rolling hills of southern Iowa were created through glacial-driven erosion processes to the north that delivered sediment downstream. Further downstream of Iowa, the coastal marshes and barrier islands on the southern US coast of the Gulf of Mexico are similarly the products of the glacially driven erosion process in the Upper Mississippi River Basin that sent enormous quantities of sediment downstream. Streambank erosion can also be affected or even driven by human activities, notably urbanization and stormwater management. Streambank erosion was recognized as a nationwide issue for water resources in 1974, when the US Congress enacted the Streambank Erosion Control Evaluation and Demonstration Act (USACE, 1981).

Since soils contain relatively large reserves of native P compared to fertilizer inputs (McDowell et al., 2023), regardless of agricultural management practices streambank erosion can entail an appreciable terrestrial-aquatic transfer of P via eroded sediments (Fox et al., 2016; Zhou et al., 2022). As a result, streambank erosion and the P-rich sediment loads it generates is a critical piece of the water quality challenge faced by many watersheds of varying scales. Quantifying streambank erosion—where and how much—is necessary to distinguish nonpoint source P responsiveness to best management practices, which can be used to identify and prioritize cost-efficient mitigation strategies. However, many nutrient loss reduction monitoring efforts and strategies do not distinguish between agricultural and nonagricultural sources P within the nonpoint sector. Without this, streambank erosion often is inaccurately assumed to be wholly an agricultural-derived P loss, which can result in a significant overestimation of agricultural contributions. This is not true for agricultural input-derived P losses (e.g., fertilizer and manure), though indirectly agricultural practices such as tile drainage can increase streambank erosion rates by increasing stream power following precipitation events (Miller & Lyon, 2021a). In some contexts, however, tile drainage can decrease stream power following precipitation (Boland-Brien et al., 2014), and thus likely streambank erosion, pointing to context dependency of hydrologically mediated agricultural impacts on streambank erosion. The counterfactual of a non-tile drained field is also important to consider, because an undrained field may experience more soil erosion and thus overland erosion contributions to sediment loading. A related unknown requiring comprehensive assessment is the relative contribution of the multiple pathways of water and concurrent sediment and nutrient export. For example, modeling studies have found tile drainage to increase peak streamflow by 14% but reduce surface runoff from fields by 7%–29% (Valayamkunnath et al., 2022). Depending on streambank and field characteristics, this could entail net decreases or increases in sediment loading via erosion of streambanks and via erosion of surface soils in fields—two distinct sources of sediment and nutrients impacted by tile drainage.

As an example of the importance of quantifying streambank erosion contributions to P loading of surface waters, an estimated 31% of total annual P losses in the state of Iowa, situated in the heart of the US Mississippi River Basin, are due to streambank erosion (Schilling et al., 2022) even without accounting for first or second order stream contributions that are likely to be net sources of sediment via bank erosion. Though generally smaller streams have lower bank heights than larger streams, in aggregate the bank erosion of these smaller streams can be a major, if not the dominant, contributor to sediment and thus P loading in the watershed (Dharamdial & Khanbilvardi, 1988; Laubel et al., 2003). In addition to stream size, contributions of streambank erosion can vary seasonally. For example, in a New Zealand catchment with dairy cattle, streambanks contributed 21% to 100% of stream P loads in winter and spring, respectively (McDowell & Wilcock, 2007). In addition to ensuring the accuracy of nonpoint source P budgets, quantifying how much and how long it will take the resulting sediment P to enter surface waters is needed to establish realistic timelines for achieving reductions in P loads. Finally, monitoring efforts to reduce nonpoint P loads will not be as accurate or effective as anticipated (or needed) if the resources for these activities are not directed to one of the most important nonpoint sources of P. Though this review focuses on streambank (i.e., creek, stream, river) erosion, it should be noted that many of these principles apply for shorebank erosion of ponds, lakes, and reservoirs.

This perspective is based on a special symposium held at the 2022 ASA-CSSA-SSSA meeting in Baltimore, MD, entitled "Streambank erosion and its contributions to P loading of surface waters." A group of researchers and policy experts convened to overview the science of streambank erosion and P loading to surface waters from the perspectives of hydrology, biogeochemistry, agricultural engineering, and nutrient regulations, with a focus on the US Mississippi River Basin. Topics discussed via invited presentations and a panel of experts are synthesized here to provide researchers and policymakers with a broad overview of the knowns, unknowns, and future directions on streambank erosion contribution to P loss budgets at varying spatiotemporal scales. Finally, we present implications of the science of streambank erosion and P loading for policy and nutrient loss reduction strategies.

2 | KNOWNS AND UNKNOWNS

2.1 | How streambank erosion is measured

Streambank erosion is difficult to quantify, and various techniques can be used that have distinct advantages and disadvantages: erosion pins (Palmer et al., 2014; Papanicolaou et al., 2017), total station surveying (Myers et al., 2019), terrestrial or airborne laser scanning (Thoma et al., 2005), unmanned aerial vehicles (UAVs) (Hamshaw et al., 2019), aerial imagery analysis (Miller et al., 2014; Ross et al., 2019), and modeling (Bressan et al., 2014).

1. Erosion pins are narrow metal rods, typically 400–800 mm length, inserted perpendicularly into the bank face. By measuring the newly exposed pin length, the bank retreat or recession rate over a given time interval can be estimated (Hooke, 1979; Ross et al., 2019). Erosion pins are inexpensive, easy to install and suitable for a wide range of fluvial environments (Myers et al., 2019). However, erosion pins can artificially inflate erosion estimates due to bank destabilization during installation or flow turbulence caused by pins (Zaimes et al., 2006). On the other hand, erosion pins may strengthen the bank against erosion, similar to rebar in reinforced concrete. Erosion pins may not capture the fine-scale losses that are cumulatively important over a large bank surface area, or the specific stretches that account for the majority of erosion (i.e., hotspots). Employing erosion pins to capture spatial variability of streambank erosion at high spatial resolution is generally cost-prohibitive due to high material and labor costs. Loss of erosion pins during mass failure of the bank (Palmer et al., 2014; Sass & Keane, 2012) leads to uncertainty of the magnitude of mass failure events in which the majority of streambank erosion loading of P occurs (i.e., hot moments).
2. Total station surveying uses mobile electronic survey to delineate the shape of streambank change over time from erosion by integrating horizontal and vertical angle and distance measurements. However, overhung banks with undercuts can challenge the feasibility of total station survey, deployment of which may disturb the bank and cause artificially higher erosion rates and thus P loads (Myers et al., 2019). Additionally, the weight of surveying stations may limit spatial coverage, especially in streambanks that are difficult to access (e.g., heavily vegetated riparian corridors and tall banks).
3. Terrestrial or airborne laser scanning uses light detection and ranging (LiDAR) technology to create high-resolution point clouds of a surface with three-dimensional (3D) topography by combining laser-based distance with precise orientation measurements. The main advantage of

laser scanning is that it can detect minute changes in surface position and shape along the bank with up to millimeter resolution, enabling accurate quantification of streambank P load (Thoma et al., 2005). However, LiDAR measurements can face optical interference from water reflection and physical interference by vegetation obscuring banks (Myers et al., 2019). Overhung banks with undercuts can also compromise the accuracy of aerial LiDAR quantification.

4. UAVs, also known as drones, can be outfitted with photogrammetric systems to collect photos of the same streambank from various angles to yield a 3D model of bank topography, which can be further integrated with digital elevation models to estimate the sediment mass loaded by erosion (Meinen & Robinson, 2020). UAV-based photogrammetry is cost-effective for high-resolution surveys of large areas of streambanks (Hamshaw et al., 2019). However, UAV flight survey of streambank eroding conditions is often limited by high density of vegetation on streambanks or unfavorable weather conditions (Hamshaw et al., 2019).
5. Imagery analysis derives the streambank lines or channel centerlines over time from rectified high-resolution aerial photography (e.g., National Agriculture Imagery Program images). Zones of erosion and deposition driven by channel migration can be identified by comparing the location of the channel between two time periods. Then, bank retreat rate and volume of soil loaded via erosion can be calculated based on the distances of the bank lines or channel centerlines (Williams et al., 2020). Aerial imagery analysis allows for rapid evaluation of bank erosion over large spatial and temporal ranges (e.g., decadal) but has less precision compared to in situ methods such as terrestrial or airborne laser scanning (Purvis & Fox, 2016).
6. Process-based models developed over the past decades can predict streambank erosion at varying scales. For example, the Bank Stability and Toe Erosion Model (BSTEM) (Midgley et al., 2012) and Conservation and Channel Evolution and Pollutant Transport System (Langendoen & Simon, 2008) are designed to predict streambank retreat and mass failure at the "field" or stream stretch scale, whereas the Soil and Water Assessment Tool (SWAT) (Narasimhan et al., 2017) and River Erosion Model (Lammers & Bledsoe, 2019) can simulate streambank erosion at the watershed scale. Recently, BSTEM has been integrated into HEC-RAS 1D (USACE, 2023). Similar to imagery analysis, these modeling tools can predict streambank erosion at fine temporal resolutions and can cover extensive areas, but also carry uncertainties that scale with the variability of streambank properties (e.g., erodibility and seasonality of erosion).

2.2 | Magnitudes and spatiotemporal variability of streambank erosion to P loading

How much streambank erosion contributes to total riverine P loading is challenging to determine. Absolute and relative contributions of streambank erosion to P loading vary by spatial scale, and perhaps most importantly for its quantification, by time (Peacher et al., 2018). Though in some cases streambank erosion is dismissed as a minor contributor to watershed P loads (e.g., Gentry et al., 2007), studies at broader spatiotemporal scales have revealed potentially substantial contributions. For example, in the US Mississippi River Basin states, contributions of streambank erosion to watershed P export loads have been found to be 30%–44% in Iowa (Beck et al., 2018; Schilling et al., 2022),10%–47% in Oklahoma (Miller et al., 2014; Mittelstet et al., 2017; Purvis & Fox, 2016), 7%–67% in Minnesota (Belmont et al., 2011; Kessler et al., 2012; Thoma et al., 2005), and 10%–67% in Missouri (Jordan et al., 2019; Wilson et al., 2022). Globally, the relative contribution of streambank erosion to P is estimated to range from 6% to 93% (Fox et al., 2016). Consequently, the relative contributions of streambank erosion to watershed P export entail large magnitudes of absolute P loads. For example, Peacher et al. (2018) reported 1.04 kg P ha^{-1} $year^{-1}$ P from streambank erosion for a 560 km^2 watershed in Missouri over a 4-year period, averaging a total of 58 Mg P $year^{-1}$. In Blue Earth County, Minnesota, Kessler et al. (2012) estimated 0.34 kg P ha^{-1} eroded annually from streambanks, totaling 66.5 Mg P $year^{-1}$. In Iowa, Schilling et al. (2022) reported an 18-year average loading rate of 0.53 kg P ha^{-1} $year^{-1}$ entailing an estimated 7700 Mg P $year^{-1}$.

2.3 | Context matters: Landscape position, channel evolution, and adjacent land use

Geographical setting and environmental context, such as landscape position, channel evolution stage, and adjacent and watershed land use and management, significantly influence streambank erosion and P loading rates (Fox et al., 2016). According to the channel evolution model (Simon & Rinaldi, 2000), incised streams undergo a six-stage sequence of bank evolution: pre-modification, channelization, degradation, degradation and widening, aggregation and widening, and quasi-equilibrium. Headwaters (i.e., source zone) and middle streams (i.e., transfer zone) (Gellis et al., 2016) are at the early to middle stages of channel evolution when the stream bed is actively (vertically) degrading and (laterally) widening leading to decreased bank stability and thus enhanced bank erosion and P loading (Fox et al., 2016). Assessment of P loading via streambank erosion, and potential erosion control measures, should therefore consider tributary headwaters (small streams) and middle zones of a watershed.

A key knowledge gap for future research identified in the session is the effect of riparian buffers on streambank erosion. Commonly used by farmers and land managers to protect streambanks from erosion, riparian vegetative buffers can deliver additional ecosystem services (e.g., habitat for native pollinators) (Naiman et al., 1993). Riparian vegetation such as grasses, shrubs and trees growing along streams can increase the cohesive strength of streambanks and simultaneously reduce the shear stress experienced by banks (Langendoen et al., 2009). Compared to streambanks without vegetation buffers with immediately adjacent annual cropping or pasture land use, streambanks with established riparian forest buffers have been found to have up to threefold lower bank recession rates (Daly et al., 2015; Zaimes et al., 2008), 72% less streambank soil loss (Zaimes et al., 2004), and one order of magnitude lower P loading rates (Zaimes et al., 2004). However, vegetation and wetlands in riparian buffers can lead to increased streambank P concentrations by trapping P from runoff and sediments from upslope areas (Hoffmann et al., 2009), posing a risk of "flash P loads" in subsequent bank erosion events. Another practical consideration of riparian buffers is the cost of installation and maintenance. For example, the annual costs of installing approximately 46-m width grass riparian buffer strips on stream edges bordering agricultural fields in the Harpeth River watershed (224,552 ha) in Tennessee, entailing 4955 ha of riparian grass buffers, totaled \$1.3 million—a cost of \$262 ha^{-1} buffer across the watershed (Roberts et al., 2009). Expenditures for riparian buffer installation could be offset by emerging carbon sequestration incentives or by water quality credit trading programs.

3 | CHALLENGES TO QUANTIFYING STREAMBANK EROSION-DRIVEN P LOADS

3.1 | A spatiotemporally heterogeneous process

P loading to streams from streambank erosion operates in nearly all watersheds but is seldom quantified largely due to the inherent variability in when and how much streambank erosion occurs. Streambank erosion is a cyclical process of interrelated— and simultaneously operating—steps of subaerial erosion, fluvial erosion, and mass wasting (Couper & Maddock, 2001; Fox et al., 2016). Subaerial erosion is caused by weather-driven freeze-thaw cycles which can reduce soil erodibility by forming a crust layer on the bank face and eventually leads to soil erosion or increased risk of fluvial entrainment and mass failure (Wynn et al., 2008; Yumoto et al., 2006). Fluvial erosion depends on the applied shear stress (i.e., flow discharge and sinuosity) and the resistance of bank soil to the fluvial force (i.e., soil erodibility) (Mittelstet et al., 2017). Factors controlling shear stress include stream discharge (e.g., precipitation events), velocity, and channel sinuosity (Peacher et al., 2018). For example, streambank erosion rates were highest in winter (December to March) and strongly correlated with annual stream discharge (Peacher et al., 2018; Wilson et al., 2022). Streambank recession rates during high discharge years can be up to 50-fold greater than in drier years of lower discharge (Palmer et al., 2014). Soil erodibility also varies regionally and seasonally depending on soil properties in conjunction with soil moisture and freeze-thaw cycles (Wilson et al., 2022). Mass wasting is caused by gravitational force and—particularly after the internal strength of the upper bank is undermined by saturation—undercutting or foundation deterioration from seepage (Fox & Wilson, 2010). Mass wasting is episodic and thus a dominant hot moment of streambank P transfer. As pointed out in the session, these three processes are causally linked, because by initially weakening a bank, subaerial erosion facilitates fluvial erosion which then undercuts the bank or scours the bed leading to increased bank instability and mass wasting (Fox et al., 2016). The episodic nature of mass failure, in which the majority of sediment and thus P loading from streambank erosion occurs (Purvis & Fox, 2016), entails acute variability in time. Overall, estimates of streambank erosion contributions to P loading are generally improved with increasing timescale of measurement, because the punctuated moments in which the majority of terrestrial-aquatic P transfers via erosion occur are more likely to be captured.

3.2 | Streambank soil P concentrations and forms vary—and matter

Streambank P load is a function of eroded soil mass (i.e., sediment) and soil P concentration, which means that streambanks with similar erosion rates could have different P loading rates due to variability of P concentration across sites and regions. Vice versa, streambanks with different erosion rates could have the same P loading rate. For example, streambank soil total P concentrations across Iowa varied by more than one order of magnitude, from 75 to 1600 mg kg^{-1} (Schilling et al., 2022). Variability in P concentrations of eroded soils is in part explained by soil texture because total P content tends to increase with fine particle content (Day et al., 1987). For example, streambanks with coarse-textured soils in Iowa had total P concentrations averaging <120 mg kg^{-1} (Moustakidis et al., 2019) whereas fine-textured soils averaged nearly 600 mg kg^{-1} (Schilling et al., 2009). Phosphorus concentrations in streambank soils are controlled by parent material, weathering, and adjacent land use (He et al., 2021). As a result, streambanks with slightly weathered soils (e.g., entisols and inceptisols) tend to have higher P concentration than more weathered banks (e.g., mollisols) (Zhou et al., 2022). Likewise, streambanks under agricultural land use tend to have higher soil P concentrations in surface horizons than streambanks under forest and grassland (Zhou et al., 2022), though this varies dramatically depending on agricultural practices. The potentially high variability of soil P concentrations poses challenges to quantify streambank P contributions across the watershed, since point observations may not fully capture soil P spatial variability and because maps of soil P concentrations at fine-scale resolution are not available (Yang et al., 2013). Since total P does not provide information on the various forms or pools of P that differ in fate upon mobilization into streams, understanding the type of P is hypothesized to be useful for understanding the impacts of streambank erosion on downstream water quality at varying timescales (Zhou et al., 2022).

3.3 | Streambank soils: Shifts from P source (release) to sink (sorption)

Though streambanks are commonly recognized as a source of P loads to streams via erosion, in some cases streambanks can serve as P sinks through sorption of dissolved P in the stream (Fox et al., 2016; Hongthanat et al., 2016). The degree of P saturation, soil P storage capacity, and equilibrium P concentration (EPC) indices can be used to evaluate the risk of P loss from soil to water, and are useful for understanding short-term P release following bank erosion (Rahutomo et al., 2018; Zhou et al., 2022). In particular, EPC depends on soil particle size and flood events, enabling prediction of the P sink–source relationships (McDowell et al., 2019). To address this in future research, session participants pointed to the need for evaluating P sorption and desorption dynamics in streambanks with variable hydrological and stream chemistry conditions (e.g., anoxic–aerobic continuum).

4 | CHALLENGES TO QUANTIFYING STREAMBANK EROSION IMPACTS ON WATER QUALITY VIA P LOADING

4.1 | True "backgrounds" or reference measurements

Since streambank erosion is a natural fluvial process, the input of P with bank material may be regarded as background P losses under certain conditions (Kronvang et al., 2012). For example, the 60% of nonpoint source P loading to surface waters in Denmark due to streambank erosion is treated as a background source and cannot be managed by agriculturalists (Andersen & Heckrath, 2020). Understanding background rates and magnitudes—how fast and how much—of biogeochemical processes is in general useful to contextualize and develop reasonable expectations of nutrient loss reduction strategies. Specifically, quantifying these background loads within the nonpoint source sector is important to avoid overestimation of the human-driven and thus manageable nutrient losses. For streambank erosion P loading, however, it is difficult to derive these measurements given that many streams have been straightened (i.e., channelized) from their original trajectories (Gregory, 2006; Simon & Rinaldi, 2006) and that finding sites that are representative of nonagricultural land uses is difficult if not impossible in many watersheds. Here, the challenge is less about agricultural impacts on soil P stocks, but on bank recession rates—which in many circumstances can be influenced by agriculture at the watershed scale.

Hydrological modification of agricultural landscapes, in particular increased field drainage and hydrological connectivity, are expected to impact stream flow and power, but predicting net effects on streambank erosion and thus P loading is complex. By improving drainage, tile drainage can reduce surface runoff volume and thus stream flow and bank erosion. However, tile drainage may increase the flashiness of the watershed, which could increase stream flow and thus bank erosion. Here, the counterfactual of a non-tile drained field is yet again important to consider, because an undrained field may experience more soil erosion and thus overland erosion contributions to sediment loading. For example, modeling studies have found tile drainage to increase peak streamflow by 14% but reduce surface runoff from fields by 7%–29% (Valayamkunnath et al., 2022). On the other hand, tile drainage may not increase and could even ameliorate downstream flashiness. For example, tile drainage in Iowa was found to buffer streamflow across low to high-flow conditions (Boland-Brien et al., 2014), whereas in Ohio tile drainage increased the flashiness of streamflow (Miller & Lyon, 2021b).

A related unknown is the comparison of different pathways of water and concurrent sediment and nutrient export. Depending on streambank and field characteristics, this could entail net decreases or increases in sediment loading via erosion of streambanks and/or of surface soils in agricultural fields—two distinct sources of sediment and thus P. A third critical question relevant to understanding tile drainage net impacts on sediment and P loading is: how much of drainage effluent is precipitation versus groundwater? Again, depending on context, tile drainage effluent can be dominantly groundwater derived (e.g., up to 65%) (Williams et al., 2022), but this can change to precipitation-dominated tile effluent in wetter years—even for the same field (Miller & Lyon, 2021b).

Obtaining a true background or baseline of streambank erosion rates and thus encumbered P loads is challenging as a result of indirect effects of land use, in particular at the watershed scale, and agricultural engineering of hydrology at the field to watershed scales. Pairwise comparisons of watersheds with "less" versus "more" agricultural land use or urbanization offer an imperfect but best available approach to estimate how land use change has impacted streambank erosion rates. Longitudinal studies in space (i.e., increasing drainage area with monitoring down the stream) and time (e.g., long-term monitoring) can also help derive insights into how land use and interannual weather variation contribute to observed streambank erosion rates (e.g., Boudreault et al., 2019). Other methods include developing a relationship between land use intensity and watershed load and using this relationship to backcast to where land use intensity is "low" (e.g., <5% agricultural land). In the freshwater sciences, relationships are more common than pairwise comparisons as finding representative watersheds with low agricultural land uses is difficult (McDowell et al., 2013).

Observed differences in the influence of tile and other forms of agricultural drainage on streambank erosion could very well reflect how different regions with distinct soils, landscapes, land and stream use histories, and climates interact today with agricultural and other drainage systems. Regionally different characteristics could lead to disparate streambank erosion responses to drainage practices and precipitation events. We do not yet know if understanding streambank erosion in one region helps in understanding how it operates in another. Extrapolating known properties and reactions in one region to another must be confirmed; otherwise, doing so may lead to ineffective or adverse policies and outcomes.

4.2 | Ultimate fate of P eroded to streams

Quantifying long-term (e.g., historical) streambank erosion can help gauge the magnitude and lag time of sediment legacy P deposited into channels. To improve estimates of how streambank erosion contributes to water quality degradation, it is necessary to quantify how much of the eroded soil P is bioavailable. As for any soil, a small fraction of total P is immediately available as soluble phosphate-P. This is important both as a direct driver of eutrophication (Holtan et al., 1988) and because only this soluble P will be measured as dissolved reactive P (DRP) loads (Figure 1). The different types of P in streambank soils means that the resulting sediment may release P at varying timescales from weeks to decades or even centuries, depending on how long the sediment is in contact with the water column. For example, organic P can be transformed (i.e., mineralization) to orthophosphate downstream by microbial activity (Dodd & Sharpley, 2015; Dodd et al., 2018), iron-bound P can be dissolved after prolonged submersion in water via reductive dissolution (House et al., 1998; Norton et al., 2008; Rahutomo et al., 2018), and calcium phosphate—abundant in many parent materials (Porder & Ramanchan, 2013)—will slowly dissolve and release P over decades (Emelko et al., 2016). The loess parent material that dominates the upper Mississippi River Basin contains P predominantly as calcium phosphate, leading to calcium phosphate-rich soils at subsurface depths (Sun et al., 2022). Thus, erosion of lower portions of streambanks (e.g., undercutting) in which P is mostly in this largely insoluble P form may not translate to immediate pulses of dissolved reactive P downstream. Measuring P species in streambank soils enables estimating the future release of DRP from sediments eroded from banks (Zhou et al., 2022).

5 | IMPLICATIONS OF SCIENCE ON STREAMBANK EROSION OF P LOADING FOR NUTRIENT LOSS REDUCTION STRATEGIES

5.1 | Missing piece of P budgets

Quantifying P loading as a result of streambank erosion is necessary to accurately distinguish between agricultural and nonagricultural sources within the nonpoint source sector. Nonpoint source contributions to P loads are generally calculated indirectly, by subtracting more readily quantifiable point source loads—many of which are regulated or monitored (e.g., total maximum daily loads) (Rissman & Carpenter, 2015)—from total P loads. As a result, nonpoint sources cannot be simply equated with agricultural contributions via fertilizer or overland erosion from fields. Correcting this default approach is important because to the extent that nonpoint source P is not derived from agricultural contributions of P that can be managed (e.g., conservation practices and 4Rs [Right source, Right rate, Right time, and Right place] of fertilizer management), resources and regulations targeting agricultural contributions will not be effective in reducing nonpoint P loading.

Examples of blind spots in nutrient budgets and watershed plans are exemplified by the US Mississippi River Basin states. Of the 12 out of 31 states in the Mississippi River Basin that have nutrient loss reduction plans or strategies (Christianson et al., 2018), streambank erosion as a nonpoint P source is acknowledged by only six states (Illinois, Arkansas, Iowa, Minnesota, Missouri, and Wisconsin) and discussed as a potential target for reductions in the original states' nutrient loss reduction strategy only by five states (Illinois, Iowa, Minnesota, Missouri, and Wisconsin) (Table 1). Three states provide an estimate of the relative magnitude of streambank erosion (Table 1) and five discuss potential mitigation approaches (Table 2). Iowa has been the only state thus far to estimate state-wide magnitudes of streambank erosion

TABLE 1 Recognition of streambank erosion as a phosphorus (P) source in the nutrient loss reduction strategies of 12 states in the US Mississippi River Basin.

State	Streambank erosion recognized as a nonpoint P source?	Description from the strategy	Measures taken to reduce P load from streambank erosion	Reference
Illinois	Yes	• Addressed under "urban nonpoint sources". • approximately 40% of NPS P loads are estimated to be derived from overland erosion, dissolved reactive P losses, and streambank erosion. • Severely eroding streambanks estimated to contribute approximately up to 30%–50% of total sediment entering surface waters in IL.	• The Illinois Streambank Stabilization and Restoration Program funds low-cost stabilization of eroding streambanks. • In 2004–2012, 93 km of eroding streambanks was stabilized, reducing loads by an estimated 25.9 Mg P.	(IEPA, 2015)
Iowa	Yes	• Streambank erosion is a relatively high proportion of P loading to Iowa streams. • Accurately accounting for streambank P sources is challenging due to limited methods for measuring beyond a local scale.	• Riparian buffers and streambank stabilization proposed.	(IDALS, 2012)
Minnesota	Yes	• Streambank erosion is described as a major source of P to surface waters and target for reduction effort. • approximately 20% of the total NPS P load from Minnesota to Mississippi River basin likely comes from streambank erosion. • Streambank erosion is the main source of P under wet conditions, but it is not significant during dry periods.	• Implementing watershed BMPs that promotes the retention or detention of surface runoff and tile drainage will aid in managing downstream flows, consequently reducing streambank erosion.	(MPCA, 2014)
Missouri	Yes	• Streambank erosion in Missouri is a significant part of P loading to surface waters.	• Missouri Soil and Water Conservation Program funds streambank stabilization and grazing management to reduce streambank erosion.	(MDNR, 20114)
Wisconsin	Yes	• Streambank erosion is a major nutrient loading source to lakes, streams, and groundwater.	• 0.3 m tillage setback from the top of a channel should be maintained to maintain streambank integrity. • Streambank and shoreline protection are identified as BMPs to manage sediment and nutrient loading and recommended to use.	(WDNR & UWE, 2013)

(Continues)

TABLE 1 (Continued)

State	Streambank erosion recognized as a nonpoint P source?	Description from the strategy	Measures taken to reduce P load from streambank erosion	Reference
Arkansas	No			(NRD, 2014)
Indiana	No			(ISDA, 2008)
Kentucky	No			(KDW, 2014)
Louisiana	No			(CPRA et al., 2014)
Mississippi	No			(MDEQ, 2012)
Ohio	No			(OEWA & OEPA, 2014)
Tennessee	No			(TDEC, 2015)

Abbreviations: BMP, best management practice; NPS, nonpoint source.

TABLE 2 Recognition of streambank erosion in biennial updates of the nutrient loss reduction strategies of Mississippi River Basin states.

State	Update year	Update mention of streambank erosion	Reference
Arkansas	2018–2023, 2022	• Streambank and shoreline erosion are common sources of NPS pollution. • Approximately 500 teams are engaged in streambank stabilization projects and monitoring stream water quality.	(ADA, 2022)
Illinois	2015–2017, 2019, 2021	• Streambank stabilization is practiced in IL under Streambank Stabilization and Restoration program. • Illinois streambank stabilization projects receive cost-share assistance, on-site technical assistance from Illinois Buffer Partnership. From 2018–2021, 5.9 km of eroding streambanks were stabilized, reducing an estimated 2729 Mg sediment and 1200 kg P loads. • These practices are not currently recommended nor tracked by the state NLRS.	(IEPA, 2015, 2019, 2021)
Indiana	2021	• Streambank and shoreline erosion recognized as nonpoint sources of nutrients in sixth version (2021) of state NRS.	(ISDA & IDEM, 2021)
Iowa	2018–2019	• Release of legacy nutrients to surface waters by bank erosion and groundwater movement is identified as a challenge for measuring state nutrient export.	(ISU et al., 2019)
Minnesota	2020	• River flow increases can escalate streambank erosion. Bank erosion is described as the largest sediment source in many rivers in Minnesota.	(MPCA, 2020)
Missouri	2020	• No mention.	(MDNR, 2020)
Ohio	2020	• No mention in the state NRS, though the state NPS plan identifies streambank erosion as a major target for P loss reductions.	(OEPA, 2020; OEWA & OEPA, 2014)
Tennessee	2021	• Streambank erosion resources will be available through online webinars.	(TDEC, 2021)
Wisconsin	2017–2019	• Streambank stabilization is a frequently used BMP to reduce nonpoint source pollution in Wisconsin. Examples include streambank stabilization in Green Bay and Fox River Area of Concern. • In Wisconsin, a nearly 1-km streambank crossing is installed, and approximately 7.5 km is covered by streambank and shoreline protection practices.	(WDNR, 2020)
Kentucky	2022	• No mention.	(KDW, 2022)
Louisiana	2021	• No mention.	(LDEQ, 2021)
Mississippi	2012	• No mention.	(MDEQ, 2023)

Abbreviations: NLRS, nutrient loss reduction strategy; NPS, nonpoint source; NRS, nutrient reduction strategy; P, phosphorus.

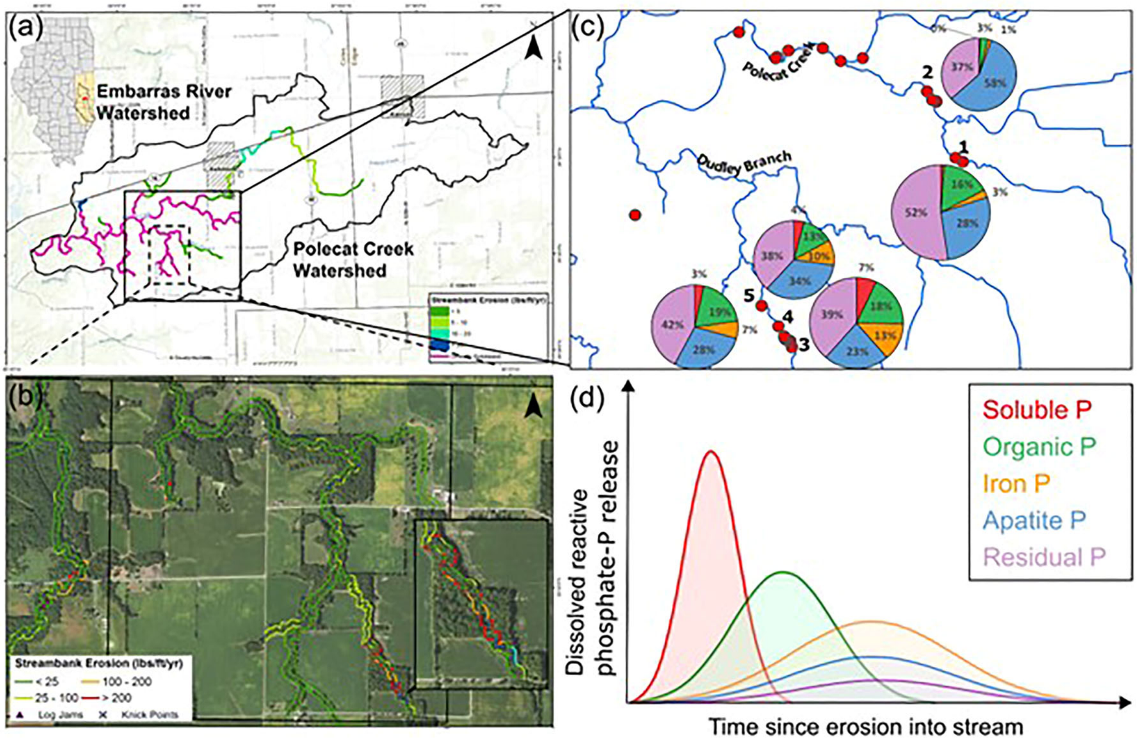

FIGURE 1 Illustration of how different forms of phosphorus (P) in streambank soils will exhibit varying release rates over time as dissolved reactive P following entry into streams via erosion. In (a) Polecat Creek of the Embarras River in central Illinois in the US Mississippi River Basin, (b) field surveys identified varying degrees of erosion severity, and (c) streambanks exhibited varying amounts and speciation of P (size of pie charts is proportional to total P concentrations). Different soil P species are expected to entail (d) varying time lags in the downstream release of P from sediments eroded into streams via bank erosion, from weeks to decades.

(Wolter et al., 2021) and to estimate streambank contribution to state-scale P loads (Schilling et al., 2022).

5.2 | Uncertainty in nutrient budgets

Estimates of the contribution of streambanks to stream P loads have uncertainties that result from necessary assumptions and inherent variability. For example, in the case of nitrate-N export, power tests of precipitation-driven variability in loads exported from Iowa revealed that interannual precipitation mutes the detectability of the magnitude of nitrate-N loss reductions at current timescales sought by the Iowa nutrient loss reduction strategy and the US EPA milestone targets (Danalatos et al., 2022). In the case of streambank erosion and P loading, uncertainties are derived from (1) data collection, (2) upscaling and modeling, (3) weather-driven variability, and (4) historical unknowns that challenge reference points for current nutrient loss reduction strategies. Without sufficient data, the present approach is largely to lump stream bank erosion into P loads attributable to land use (i.e., agriculture). This misses an opportunity to account for and remediate a potentially significant P loss source.

5.3 | Lag times

Sediment P loads to streams and rivers engendered by bank erosion are sometimes referred to as legacy P, as sediments can take a long time to migrate downstream. Despite the varying connotations and intended meanings of this term in soil science, agronomy, and hydrology, streambank erosion is one of the mechanisms by which legacy P is accumulated, specifically in stream channels, which constitutes a large but relatively unquantified P reservoir (Jarvie et al., 2005; Walling et al., 2008). For example, in an Iowa watershed, the amount of P stored in stream channels was estimated to be approximately equal to watershed export load (Beck et al., 2022). Due to gradual release of dissolved P from sediments (see section 4.2) and the relatively slow downstream migration of sediment particles, legacy sediment can entail a chronic release of P that impacts downstream water quality over time scales of decades to centuries (Jarvie et al., 2013). In other words, sediments eroded from streambanks are likely to continue to impact water quality for years after erosion occurred, complicating attribution of P losses and delaying discernable decreases in P loading following mitigation efforts (Sharpley et al., 2013). Quantifying the magnitude and kinetics of streambed sediments engendered by past erosion is challenging but would provide insights to the extent of such lag times. For example, evaluation of sediments in streams that feed Lake Mendota in Wisconsin linked the majority of dissolved reactive P loading to the lake to sediments deposited by erosion—both streambank and overland (e.g., agricultural fields)—in the late 1800s (Bortleson & Lee, 1972; DCP, 2023).

6 | RECOMMENDATIONS ON SCIENCE AND POLICY

6.1 | Science needs and directions

As streambanks are an interface of terrestrial and aquatic systems, understanding streambank erosion requires integration of hydrology and soil sciences (Zhou et al., 2022). The general siloing of hydrology, sedimentology, pedology, and biogeochemistry is a key barrier to an integrated understanding of streambank erosion. For example, while many studies focus on streambank recession rates and erosion rates, there is little to no empirical measurement of soil P stocks to full bank depth, let alone P speciation (Zhou et al., 2022). Measuring P stocks of streambanks is similar to that of upland soils, requiring bulk density measurements paired with soil samples analyzed for total P concentrations (e.g., Zhou & Margenot, 2023). In other cases, watershed-scale P loading from streambank erosion is gauged by mass balance. There is a missed opportunity to link these watershed P balances to drivers such as high flow events or spatially explicit soil P variation along the river corridor (e.g., Noe et al., 2022) to explain the "why" of streambank erosion rates.

Even when soil P and recession rates are explicitly evaluated to increase accuracy of streambank P loading estimates (e.g., Schilling et al., 2022), directly linking streambank erosion with downstream P loads is challenged by sediment P release and thus lag times. Causal linkages of streambank P loading with water quality remain poorly defined and understood, and to establish these linkages will require integration of multiple disciplines. Given the high spatiotemporal variability of streambank erosion at sub-decadal timescales, capturing the flashiness of streambank erosion that entails the majority of P loads is important for accurate assessments but is challenged by the high resource cost needed to achieve this. Research support that recognizes the need for interdisciplinary approaches and provides sufficient spatial and temporal resolution (e.g., project funding >5 years) is needed. For example, establishing long-term streambank erosion programs via erosion pin networks or remote sensing would pay dividends at supradecadal timescales. A third scientific priority is understanding indirect impacts of land use, notably agricultural water management such as ditching and tile drainage, on streambank erosion.

In the context of state nutrient reduction strategies, efforts to better quantify and characterize P loads stemming from streambank erosion are important as they help to differentiate P load sources among agricultural nonpoint sources, point sources, and in-stream nonpoint sources. Improved understanding of the magnitude of P load being generated across these three P source categories will help to inform practical expectations for the amount of P load reductions that can be achieved. Finally, increasing the awareness of those in the water quality and conservation communities on streambank erosion provides a basis for seeding bottom-up and top-down approaches to monitor this inherently variable process.

6.2 | Acting on a developing science: Implications for policy

Policy can play a key role in directing research and management to mitigate nutrient losses. In the case of streambank erosion, a priority for policy is the acknowledgment and accounting for streambank erosion within the nonpoint sector of P losses. In many cases, limited or no data on the magnitude of streambank erosion contributes to nonpoint P loading to surface waters should require caution. At the very least, nonpoint sources should not be necessarily equated with agricultural contributions. In light of the uncertainties in the scientific community's understanding of streambank erosion, policy should consider ways to address these knowledge gaps. Soil and water conservation plans should include monitoring of streambank erosion and consider potential mitigation strategies.

As an illustrative case study, stream bank erosion has been identified as a significant source of sediment and P loads to streams in New Zealand, where more than 90% of agricultural land use consists of livestock grazing pastures year-round (McDowell & Wilcock, 2007). Approximately 80% of P loads were from small streams, often unfenced, in pastures (McDowell et al., 2017). However, extension efforts advocating the use of fencing are thought to be responsible for gradual reductions in P loads over time (McDowell et al., 2019). To increase the rate of water quality improvement, the New Zealand parliament passed a law requiring livestock to be excluded from many streams (Reddy, 2023), which was further reinforced by mandatory farm plans as of 2024 (Stokes et al., 2021). These required farm plans identify remedial actions such as livestock exclusion and streambank stability linked to specific watersheds objectives, and place a legal requirement on landowners and managers to manage agricultural operations to improve water quality.

The extent to which streambank erosion can and should be managed is uncertain, and thus should be approached carefully in recommendations, P loss reduction strategies and legislation. To the extent that streambank erosion is a "natural" or "background" process, mitigation efforts may not be a wise investment of resources. Streambank erosion can be mitigated by direct shielding or armoring of streambanks with riprap (i.e., large stones or concrete blocks), concrete slabs, or vegetation, and indirectly by modifying water flow (Allen & Leech, 1997). Beyond the high costs of streambank protection, which are often prohibitive, there remain unknowns on how to optimize protection strategies and potential trade-offs for stream ecosystem health (Allen & Leech, 1997; Reid & Church, 2015; Shields et al., 2000). Prioritizing streambanks with high erosion and/or P loading rates can support cost-effective management, but requires watershed-scale understanding of drivers of streambank erosion. In some cases, conservation and nutrient loss mitigation dollars may be better invested in wetland buffers that mitigate high flow volumes following precipitation upstream of an erodible stretch of stream.

AUTHOR CONTRIBUTIONS

Andrew J. Margenot: Conceptualization; funding acquisition; project administration; resources; supervision; visualization; writing—original draft; writing—review and editing. **Shengnan Zhou**: Supervision; writing—original draft; writing—review and editing. **Richard McDowell**: Conceptualization; writing—review and editing. **Thomas Hebert**: Conceptualization; methodology; resources; writing—review and editing. **Garey Fox**: Writing—review and editing. **Keith Schilling**: Writing—review and editing. **Shawn Richmond**: Writing—review and editing. **John L. Kovar**: Writing—review and editing. **Dean Lemke**: Writing—review and editing. **Kathy Boomer**: Writing—review and editing. **Shani Golovay**: Writing—review and editing. **Niranga Wickramarathne**: Writing—original draft; writing—review and editing.

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CONFLICT OF INTEREST STATEMENT

The authors declare no conflicts of interest.

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