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Beyond the Edge: Linking Agricultural Landscapes, Stream Networks, and Best Management Practices

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Abstract

Despite much research and investment into understanding and managing nutrients across agricultural landscapes, nutrient runoff to freshwater ecosystems is still a major concern. We argue there is currently a disconnect between the management of watershed surfaces (agricultural landscape) and river networks (riverine landscape). These landscapes are commonly managed separately, but there is limited cohesiveness between agricultural landscape-focused research and river science, despite similar end goals. Interdisciplinary research into stream networks that drain agricultural landscapes is expanding but is fraught with problems. Conceptual frameworks are useful tools to order phenomena, reveal patterns and processes, and in interdisciplinary river science, enable the joining of multiple areas of understanding into a single conceptual–empirical structure. We present a framework for the interdisciplinary study and management of agricultural and riverine landscapes. The framework includes components of an ecosystems approach to the study of catchment–stream networks, resilience thinking, and strategic adaptive management. Application of the framework is illustrated through a study of the Fox Basin in Wisconsin, USA. To fully realize the goal of nutrient reduction in the basin, we suggest that greater emphasis is needed on where best management practices (BMPs) are used within the spatial context of the combined watershed–stream network system, including BMPs within the river channel. Targeted placement of BMPs throughout the riverine landscape would increase the overall buffering capacity of the system to nutrient runoff and thus its resilience to current and future disturbances.

Core Ideas

- Managing agricultural landscapes should integrate land and river network concerns.
- An ecosystems approach and resilience are combined with strategic adaptive management.
- BMPs should be implemented across the integrated landscape.

Nutrients are important to agricultural and natural ecosystems because they drive the overall productivity of these systems. Highly productive agriculture is required for an increasing human population, and additions of nitrogen (N) and phosphorus (P) are necessary to sustain large crop yields. In natural ecosystems, limiting nutrients can determine community composition and ecosystem functioning. Managing nutrients in agricultural landscapes is a key issue that commonly entails trying to limit nutrient runoff. The cost of agricultural nutrient management and mitigation is estimated at US$3.5 billion annually in the United States (Becker, 2002). The cost associated with increased nutrients in aquatic ecosystems and eutrophication is ~$2.2 billion annually (Dodds et al., 2009). In the Gulf of Mexico, where the world's second largest hypoxic zone forms each summer, the estimated cost of reaching hypoxic zone management targets is $2.7 billion annually (Rabotyagov et al., 2014). Most of this investment in nutrient management occurs in the Upper Mississippi River Basin, with conservation efforts focused on minimizing nutrient loss from agricultural fields. Similarly, in the Laurentian Great Lakes, algal blooms occur more frequently during the summer months in shallow bays receiving P-rich runoff from agricultural areas (Michalak et al., 2013). The estimated cost of a 20-yr plan to reach target P loads in Green Bay (Lake Michigan) is as high as $618.5 million (Vande Hey, 2014), and up to $263 million for Maumee Bay (Lake Erie) (Keitzer et al., 2016).

Research on the nutrient management in agricultural landscapes primarily focuses on reducing nutrient exports (Sharpley and Jarvie, 2012). Programs such as the USDA's Environmental Quality Incentives Program and the USEPA's Great Lakes Restoration Initiative have developed and applied best management practices (BMPs) for nutrients in agricultural landscapes within the United States. These practices have two broad objectives: to improve nutrient management at the source, with a focus on land management activities, and to reduce nutrient and soil loss from agricultural fields into receiving waterways (i.e., transport management) (Sharpley and Jarvie, 2012). Best management practices have had some success in reducing nutrient inputs

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Abbreviations: BMP, best management practice; SAM, strategic adaptive management; SPARROW, Spatially Referenced Regression on Watershed; SWAT, Soil and Water Assessment Tool; TMDL, total maximum daily load; TPC, threshold of potential concern; WWTP, wastewater treatment plant.

to waterways by decreasing the amount of sediment-bound nutrients that are transported during runoff (Sharpley and Jarvie, 2012). However, the considerable investment in BMPs has not substantially alleviated excess runoff in many regions or dealt with legacy effects whereby the negative environmental degradation continues to occur long after the initial appearance of the disturbance. For example, nitrate loading to the Gulf of Mexico has risen by at least 9% since 1980 (Sprague et al., 2011), and the size of the Gulf of Mexico hypoxic zone continues to increase (Turner et al., 2012), even though during that period there has been some reduction in nutrient export associated with a significant investment in BMPs in the basin (USDA, 2012). Despite our increased understanding of the importance of linkages between agricultural landscapes and stream networks, the impact of land-based activities on the latter continues.

There is a disconnect between the management of watershed surfaces—the agricultural landscape—and river networks— the receiving water bodies or riverine landscape—that drain them. Management of these systems is frequently undertaken separately, and there is limited cohesiveness between agricultural landscape-focused research and river science (Gilvear et al., 2016), despite their often having similar end goals in terms of systems understanding (Gore et al., 2016). This limits the effectiveness of BMPs in reducing nutrient loads from agricultural landscapes to receiving stream and river networks.

Many of the challenges of undertaking research at discipline interfaces can be overcome with a greater emphasis on a systems approach (Pickett et al., 1999; Dollar et al., 2007) to resolve issues of scale. Conceptual frameworks are commonly used in interdisciplinary research (Dollar et al., 2007; Delong and Thoms, 2016). They are useful tools for integrating different disciplines and are used widely as a means to organize ideas, understand systems, link cause and effect, and guide decisions about system management (Parsons et al., 2009).

In this paper, we outline a framework for the interdisciplinary study and management of stream networks that drain agricultural landscapes. This framework is designed to increase the efficiency and effectiveness of reducing nutrient loads and eutrophication, thereby promoting greater resilience of both agricultural and natural ecosystems. This conceptual framework integrates an ecosystems approach to the study of catchment–stream networks, resilience thinking, and strategic adaptive management (SAM). It is based on the frameworks developed by Dollar et al. (2007), Parsons et al. (2009), and Delong and Thoms (2016) for the study and management of rivers as coupled social–ecological systems. To properly manage the nutrient biogeochemistry of stream networks in agricultural landscapes, it is necessary to combine these three approaches to overcome potential gaps in each approach. The ecosystems approach emphasizes linkages between catchments and their stream networks, whereas resilience thinking uses a social–ecological approach to understand ecosystems. Strategic adaptive management follows a detailed operational procedure for managing ecosystem resilience, one that has not been applied to agricultural ecosystems (Rogers et al., 2008). Integrating these three key concepts into a single conceptual–empirical structure will provide a new basis for studying and managing agricultural ecosystems.

Underlying Concepts

Ecosystems Approach

An ecosystems approach integrates land, water, and biotic components and, in doing so, facilitates an understanding of the structure and functioning of landscapes and riverscapes as a whole (Likens, 1992). The ecosystems approach acknowledges the influence of disturbance, scale, spatial heterogeneity, and temporal variability on the relationships between the physical environment and biotic community, as well as the role of humans as a keystone species in coupled social–ecological systems (Parsons et al., 2009). Taking an ecosystems approach allows managers to have a broader view of landscapes, the connections between the various components contained within them, and the different spatial and temporal scales at which they operate. Managers are then better able to address complex anthropogenic environmental problems with holistic, integrated solutions that benefit the entire ecosystem (Likens, 1992). Resource managers deal with landscapes that are constantly changing as a result of both natural and anthropogenic disturbances (Lake, 2013). Management decisions may be made that react to the current situation but are not always beneficial to the whole ecosystem. This situation frequently results in complex, often unpredicted ecosystem responses to management decisions that may not be effective in the long term and may come with additional problems.

The structure and functioning of riverine landscapes are governed by a variety of independent and dependent process variables (Fig. 1). The climate, geology, topography, soils, vegetation, and land use (including agriculture) of a watershed are the main variables that determine the key processes of the flow, sediment, and biogeochemical regimes of riverine landscapes. These dependent watershed-process drivers, in turn, directly influence the morphology and dynamics of river channels in terms of their size, shape, slope, and planform, as well as the adjacent floodplain. Traditionally, agricultural science has focused on the nature of the independent variables and the impact of various land uses, with relatively little focus beyond the edge of the stream or river, even though agriculture has a direct effect on stream structure and functioning (Fig. 1). Likewise, river scientists often have difficulty upscaling from individual site and reach-specific studies to larger river corridor and watershed applications (Harvey and Gooseff, 2015).

Applying an ecosystems approach to nutrient management within agricultural landscapes takes into account both the land surface and the stream network through collective management (Fig. 2A). Connections between the terrestrial landscape and the river network play an important role in regulating the flow of materials and information across both landscapes. Likens (2004) suggested that streams are like the bodily fluids of an ecosystem and are an indication of ecosystem health, not only by identifying potential problems but also as a reflection of basic ecosystem functioning.

The transfer of materials from terrestrial landscapes to stream networks varies naturally according to landscape geology and slope conditions, as well as spatial and temporal variations in rainfall (Allan, 2004). Agricultural land managers have traditionally focused on the terrestrial landscape, although the effects of agricultural production do not stop at the edge of the field and stream. Human land-based activities, such as agriculture, significantly

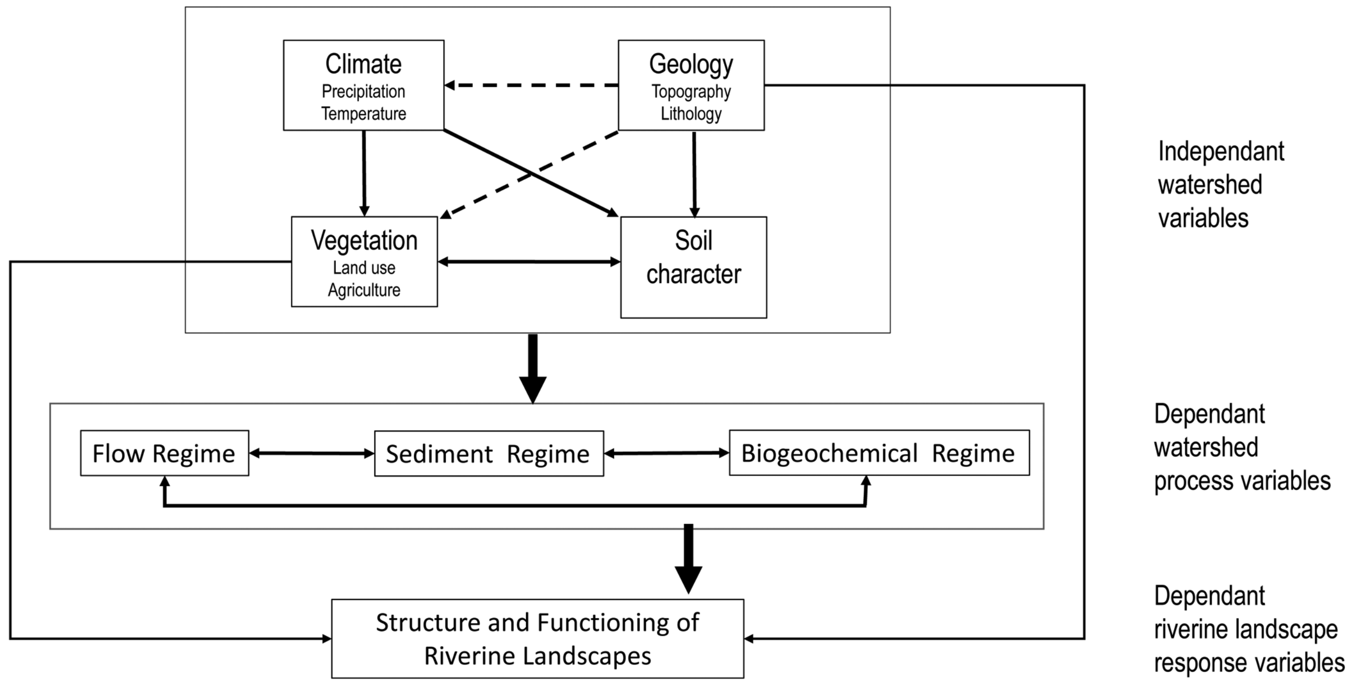

Fig. 1. Watershed and process drivers of the structure and functioning of riverine landscapes.

influence the biophysical character of receiving stream networks via changes in the flux and composition of materials from the terrestrial landscape (Allan, 2004). In some locations, natural erosion of the streambank also contributes a significant portion of the sediment load to the stream network (Fox et al., 2016). Soil and nutrients that end up in the water column as a result of streambank erosion and agricultural land management practices cause problems in the stream network and deplete the terrestrial landscape of essential nutrients. Thus, in the agricultural ecosystem, it is imperative to manage the terrestrial landscape and stream network together for the benefit of both.

The connection between nutrient sources and sinks is a critical factor determining the rate of nutrient delivery, material cycling, and transformation. Agricultural fertilizers and other agrochemicals may be applied some distance from where they are eventually deposited into the river network or water body. These materials are transported along multiple flow paths from the application site to surface or groundwater pools. Movement of N- and P-rich surface water into groundwater pools is a flow path that eventually leads to the enrichment of streams and rivers (Tesoriero et al., 2009). Water removal practices, like tile drainage and ditching, that bypass vegetated buffer zones containing carbon-rich riparian soils increase land–water connectivity and are particularly potent at delivering dissolved N and P to surface waters (Tomer et al., 2008). Although these drainage practices substantially improve agricultural production on hydric soils, they also contribute significantly to flux of nutrients into stream networks (Tomer et al., 2008). Nitrate is more commonly found moving though groundwater flow paths than soluble P due to the propensity of P to bind to soil aluminum and iron. However, in regions where soil cation concentrations are low, groundwater tables are high, or the groundwater has low dissolved oxygen and

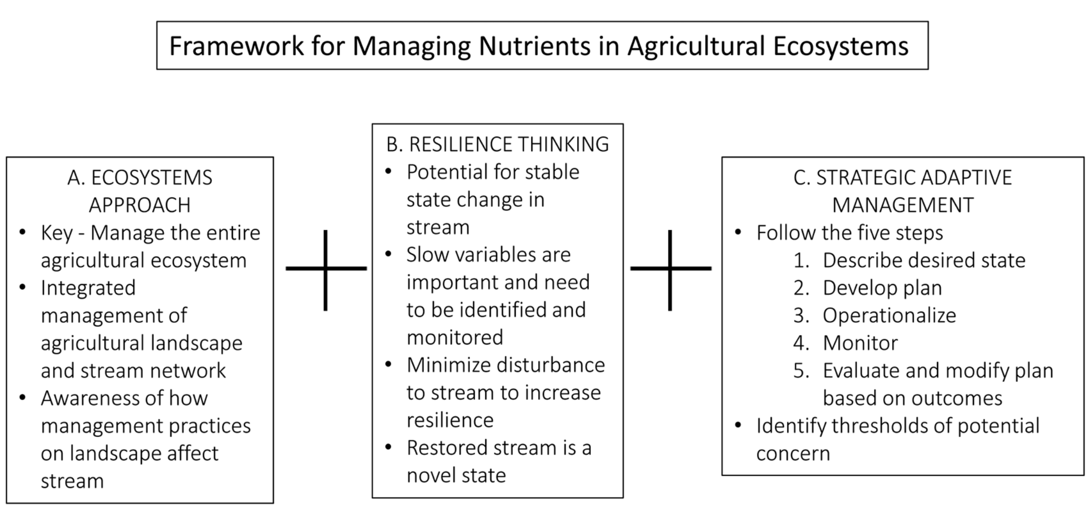

Fig. 2. Framework for managing nutrients in agricultural ecosystems that consists of three components: ecosystems approach, resilience thinking, and strategic adaptive management. Each element has the key tenets listed underneath it.

high sulfate and silica concentrations, groundwater may be a significant source of soluble P (Caraco et al., 1989; Tesoriero et al., 2009). The elevated influx of nutrients to the stream network is a major disturbance.

Although natural disturbances are important drivers that define ecosystem structure and function, anthropogenic disturbances commonly cause additional stress on ecosystem structures and functioning (Lake, 2000). Ecosystems adjust to the magnitude and frequency of natural disturbances, which promotes species diversity and enhances ecosystem processes like nutrient spiraling (Lake, 2013). However, additional stress from anthropogenic disturbances often does not allow aquatic ecosystems time to adjust to natural disturbances (Folke, 2016). For example, repeated application of N and P to agricultural landscapes via fertilization and waste products from animals often leads to nutrient-saturated soils that are at risk of erosion during storms, releasing particulate nutrients to receiving streams and rivers (Withers and Jarvie, 2008; Kröger et al., 2013). Dissolved nutrients are also drained from the soil into tile drains that empty into streams (Blann et al., 2009; King et al., 2015). Typically these are pulse disturbances, as most of the sediments and nutrients are added to the stream during storms, and nutrient concentration decreases after the storm flow subsides. However, if enough dissolved nutrients infiltrate the groundwater, concentrations in groundwater will become elevated and the base flow nutrient concentration in the stream will increase (Tesoriero et al., 2013). Other human disturbances such as straightening of the channel, removal of debris and vegetation, and sedimentation reduce transient storage and hyporheic exchange, undermining the capability of the stream to retain and cycle nutrients (Sheibley et al., 2014).

Anthropogenic disturbances can create legacy effects (Allan, 2004), which is a press disturbance. Nutrients applied to farm fields can have an effect years beyond the targeted growing season. Legacy P "stored" in farm fields and stream beds can be released decades after nutrient inputs to the landscape have ceased (Sharpley and Jarvie, 2012), and dissolved nutrients that have infiltrated the groundwater can appear in streams years after they were applied to fields (Tesoriero et al., 2013). Managing agricultural ecosystems using the ecosystems approach (Fig. 2A) challenges managers to assess how management practices on the agricultural landscape can affect the entire ecosystem from the agricultural landscape to the stream network for many years, not just the current growing season.

Resilience Thinking

Resilience thinking is a rapidly developing concept focused on coupled human and natural systems. It advocates an approach in which ecosystems, economies, and societies are managed as linked social–ecological systems, effectively facilitating knowledge exchange and adoption at the turbulent boundary of science, management, and policy (Thoms et al., 2017). In this context, resilience is viewed as a key property of human (including agricultural) and natural ecosystems for maintaining desired states and long-term sustainability (Gunderson et al., 2010). The Resilience Alliance defines resilience in terms of system change, where "resilience is the amount of change a system can undergo (its capacity to absorb disturbance) and remain within the same regime that essentially retains the same function, structure, and feedbacks" (Walker and Salt, 2006). Although this definition is a foundation of the social–ecological concepts of resilience thinking, there are different ways to consider resilience. From an engineering perspective, resilience emphasizes efficiency, and it depends on constancy and predictability, as these are the factors desired by fail-safe design (Gunderson et al., 2002). Engineering resilience emphasizes the speed of return to a steady state after perturbation, and the focus is on efficiency of function (Gunderson et al., 2002). Ecological resilience recognizes conditions far from a steady state, where instabilities can flip or tip a system into another regime of behavior. In this context, resilience is measured as the magnitude of disturbance that a system can absorb before the system is restructured in another state with different structures, processes, and feedbacks (Gunderson et al., 2002). The focus of ecological resilience is, therefore, on maintaining function (Gunderson et al., 2002). Parsons et al. (2009) note that the views of engineering resilience tend to come from deductive traditions of mathematics and engineering, whereas those of ecological resilience are from traditions of applied mathematics and applied resource ecology (Gunderson et al., 2002). In agricultural ecosystems, resilience is typically discussed in terms of engineering resilience.

Under the ecological definition of resilience, there are also two types of resilience to consider: general and specific resilience. Specified resilience has an emphasis on specific threats to ecosystems (Walker and Salt, 2012). Identifying and managing this type of resilience is important. However, ecosystems have properties that allow them to absorb unforeseen disturbances without changing state, and these properties confer general resilience. In general, properties that confer general resilience are diversity (variety of species, people, and institutions), modularity (linkages between system components), and tightness of feedbacks (how quickly and strongly change in one part of a system is felt in another part of the system) (Walker and Salt, 2006). Despite the importance of general resilience, identifying and managing properties of general resilience such as diversity, cross-scale linkages, and feedbacks are rare in comparison with the management of properties of specific resilience.

The ability of an ecosystem to absorb disturbance is a function of several properties, each of which has been described by several interacting concepts with application in the agricultural, ecological, social, and economic disciplines. These are adaptive loops, thresholds, slow variables, cross-scale interactions, multiple states, regime shifts, adaptability, transformation, and adaptive management. As a heuristic tool, in resilience thinking, Walker and Salt (2006) have divided these concepts into several groups. The first pertains to the concepts of multiple states and regime shifts. These concepts describe the way that systems change from one state to another. Related concepts of capacity loops and cross-scale interactions form the second group. These concepts describe the dynamic nature of systems and our ability to manage coupled systems. Thus resilience thinking provides managers with a different approach for the sustainable management of stream networks that drain agricultural catchments in general, and of determining restoration approaches in the agricultural landscape ecosystem specifically.

The way that systems change state, and the thresholds that sit between different states, have been described in resilience thinking as the ball-in-cup model (Fig. 3; Walker and Salt, 2006). The

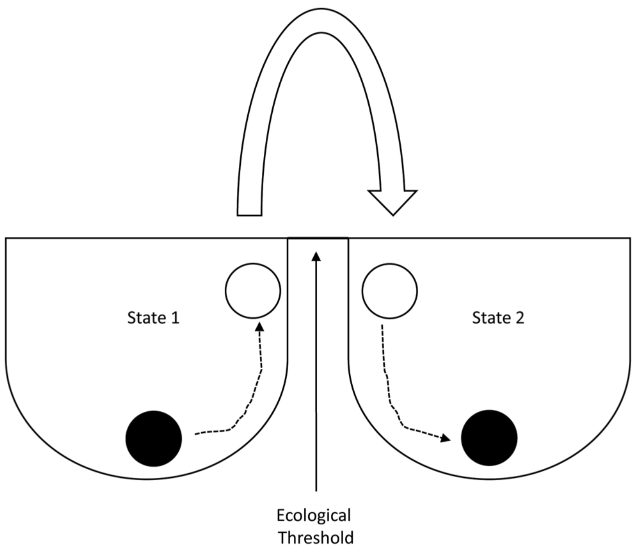

Fig. 3. Ball-in-cup model illustrating a stable state change. The system is illustrated by the ball, and the cup represents the stable state. When the ball surpasses the ecological threshold, it enters a new stable state.

space where the system, or "ball," tends to remain is known as the basin of attraction. The system typically remains in one state until a disturbance tips it over a threshold or boundary into the next state (Leuven and Poudevigne, 2002). In river systems, the basin of attraction is constantly moving, because rivers are continuously affected by disturbances, internal and external forces, stochasticity, and decision making (Walker et al., 2004). The properties around the basin of attraction may change, as systems frequently change from internal and external disturbances, inducing a state change in the system. The shallower a basin becomes as it is subjected to repeated disturbances, the more likely that the system will be pushed into another state (Walker and Salt, 2006). Increased agricultural disturbances have driven many streams to shift into a degraded state (Leuven and Poudevigne, 2002; Allan, 2004).

Regime shifts have been demonstrated in many terrestrial and aquatic ecosystems worldwide, and they can occur at multiple scales (Parsons et al., 2009). Our current understanding of thresholds between regime shifts suggests that, although system dynamics are driven by many variables that operate at different scales, system trajectories are driven by a small set of controlling variables (Folke et al., 2004). These variables are also known as "slow" variables because they determine the boundaries beyond which disturbances could push the system into another state (Scheffer and Carpenter, 2003). Regime shifts into a new state are commonly associated with a reduction in the productivity of natural ecosystems, often determined by ecosystem goods and services (Leuven and Poudevigne, 2002). For example, in agricultural ecosystems, the disturbance is land use change, and the loss in natural goods and services is reflected in the increase in artificial fertilizer. For the associated river network, the slow variable is changes in in-stream nutrient concentration, and the fast variable is algae community composition and abundance. Identifying and managing the slow variables is necessary to maintain the resilience in agricultural ecosystems (Fig. 2B).

Regime shifts may be reversible or irreversible, or the ecosystem may have to undergo hysteresis first, in which there is more than one stable state over a range of conditions that are separated by an unstable equilibrium. Reversing an undesirable regime shift is scale dependent, as regime shifts at larger spatial scales generally take a longer time to restore. Regime shifts are caused by disturbances that are external to the system that change the slow or controlling variables of the system (Walker et al., 2012). As the slow variable approaches a threshold, the fast variable, which is the variable that is of primary concern to ecosystem users, fluctuates more to the point that it may push the system across the ecological threshold to a different state (Walker et al., 2012).

Restoring river ecosystems degraded by human activities can cost billions of dollars annually (Bernhardt et al., 2005). Rivers and streams that drain agricultural landscapes become degraded and, as a consequence, display reduced species diversity and are unable to perform the same ecosystem functions and feedbacks as natural river ecosystems. For example, reduced biodiversity of stream algae has been shown to reduce water quality by limiting nitrate uptake rates (Cardinale, 2011). Restoration attempts to influence the adaptive capacity or resilience capacity of systems. Many current attempts at stream restoration follow channel-design approaches, such as the Natural Channel Design method (Rosgen, 1994). These methods restore the physical structure of the natural channel by focusing on channel morphology; however, restoration of the key chemical and biological processes is ignored (Lake et al., 2007; Bernhardt and Palmer, 2011; Palmer et al., 2014). Structure-focused restoration only improves conditions at the local site or reach (Gilvear et al., 2013; Wohl et al., 2015), and in some cases, the physical process of restoring the stream actually further degrades it (Bernhardt and Palmer, 2011). To restore ecosystem functions and feedbacks throughout the entire stream network, restoration needs to occur not only in the river channel but also on the agricultural landscape (Gilvear et al., 2016).

Restoration that attempts to transition systems back to the "natural" stable state may not be feasible as long as intensive agriculture and other human disturbances continue to occur in the watershed—often referred to as a "pressed" disturbance (Palmer et al., 2014; Wohl et al., 2015). Restoration success may be short-lived, as agricultural disturbances will continually stress the natural functioning of the ecosystem. The system may be past the logistical threshold for conservation (Truitt et al., 2015), because the monetary requirements to restore the stream to a natural state are too great. In some cases, stream restoration may be attempted, but the resulting stream condition is different in the restored stream compared with a natural stream. The restored stream has thus entered a new state where the stream functions differently from streams in both the natural and degraded agricultural stable states (Fig. 4). The restored state, or "novel ecosystem" (Hobbs et al., 2014), does not retain the full ecosystem functions of a natural stream but can still provide important ecosystem services that are lacking in the degraded agricultural state, such as clean drinking water, nutrient processing and retention, and habitat diversity for fish and other organisms. The criteria to determine whether a stream is in the restored state will differ for each stream, depending on the goals and objectives for ecosystem functioning defined by management. A stream can be considered restored if it is able to maintain that state and function without continual management input to resist the agricultural disturbance, and also through providing enhanced ecosystem services. Best management practices can be used in the agricultural ecosystem to help reduce the magnitude of the

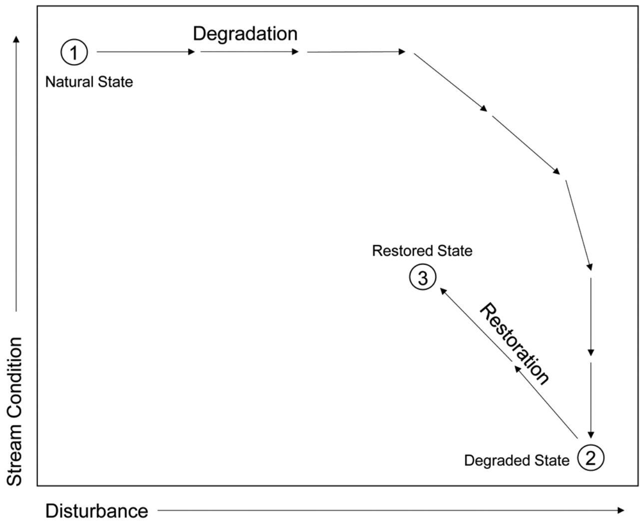

Fig. 4. Three potential stable states for a stream based on the amount of anthropogenic disturbance occurring in the watershed. If a stream is minimally disturbed and the stream condition is excellent, it will be in the natural state. A stream that experiences a lot of anthropogenic disturbance due to agriculture will be in a degraded state where the stream condition is poor. A stream is in the restored state when a considerable amount of the catchment is in agriculture and, historically, the stream experienced much anthropogenic disturbance but is now buffered from many of the negative effects due to agricultural best management practices and other restorative measures. The stream condition of the restored state is improved over the stream condition of the degraded state, but it is not in the same condition as the natural state. (adapted from Lake et al., 2007; Fig. 3).

disturbance experienced in the stream and maintain the stream in the restored state.

Ultimately, resilience of the stream network is linked to the resilience of the agricultural landscape, because they are both components of an integrated agricultural ecosystem (McCluney et al., 2014). For stream restoration to either a natural or a restored state to be successful, we suggest that resilience thinking must be incorporated into the management of the agricultural ecosystem (Fig. 2B). Agricultural resilience is currently defined as helping farmers maintain their productivity and livelihood while dealing with outside stressors (Farming First, 2014). We suggest that for an agricultural ecosystem to be resilient, it also needs to be sustainable for both the land surface and river network. This occurs when agricultural practices meet the needs of humans while being efficient in the use of nutrients, water, and land, and while limiting the negative impacts on biodiversity and downstream water bodies (Bennett et al., 2014). Successful management of agricultural ecosystems acknowledges that anthropogenic disturbances will continue to affect streams. By minimizing both press and pulse disturbances experienced by the stream through the effective use of BMPs and by promoting mechanisms for the stream to buffer against the disturbances, the agricultural ecosystem can become more ecologically resilient.

Strategic Adaptive Management

Strategic adaptive management is a procedure for natural resource management and decision making in environmental, social, and institutional situations characterized by variability, uncertainty, incomplete knowledge, and multiple stakeholders. Three key tenets form the basis for the management and decision-making process in SAM: strategic and value-based planning based on scientific and societal needs and values; a learning-by-doing approach to management planning; and participatory engagement of all stakeholders to serve their needs, access their inputs, and secure their cooperation (Rogers et al., 2008). It presents an ideal approach to manage for resilience in agricultural ecosystems, because it focuses on a future desired condition that balances societal needs with desired ecosystem functioning (McLoughlin and Thoms, 2015). Strategic adaptive management gives managers the flexibility and tools to manage for resilience while adapting to unforeseen problems and disturbances in the ecosystem. One of the main benefits of SAM is that it allows managers to learn by doing through both trial and error and simple experimentation (McLoughlin and Thoms, 2015). For SAM to be effective in agricultural ecosystems, all stakeholders (agricultural land managers, farmers, the general public, and river managers from local, state, and federal agencies who are responsible for fish and wildlife management and enforcing water quality standards) need to be involved in the process. By developing common goals and objectives and by being flexible to learn from the outcomes of management decisions, stakeholders can benefit. Open communication and more collaboration between agricultural land and river managers also results in greater engagement from both sides.

There are five interacting steps in SAM that incorporate ecological and societal needs throughout the process (Fig. 2C; Rogers et al., 2008). First, all stakeholders work toward an agreed desired state, which involves developing a shared vision of the preferred future social–ecological conditions of the system and translating the collective vision into a set of ecosystem objectives and outcomes. Objectives may include managing for a particular riverine community composition while maintaining a target agricultural yield. Once the ecosystem objectives and outcomes are identified, thresholds of potential concern (TPCs) are generated. Thresholds of potential concern represent a set of operational goals that explicitly and quantitatively define the conditions for management. They form the basis of an inductive approach to adaptive management and represent hypotheses of the upper and lower limits of acceptable change in biotic and abiotic indicators of ecosystem structure and function (Rogers and Biggs, 1999). These hypotheses are nested under the vision of the targeted desired state of the ecosystem and the objectives hierarchy, all of which are generated by stakeholder consensus (e.g., scientists, managers, and landholders). Thresholds of potential concern can be adaptively modified as understanding and experience of the system improves. They act as mediators of a structured science–monitoring–management–society relationship.

The second step in SAM is to develop a management plan. This step includes looking at all the potential outcomes of each identified management option and then selecting the preferred option that can achieve or maintain the desired state while taking into account the different needs of all stakeholders. Managers can look at all options to improve water quality or river health in view of the potential loss of agricultural productivity. Agricultural conservation frameworks and spatial tools have been developed, for example, to help identify areas where BMPs can be employed in the terrestrial landscape and in the riverine network to improve soil management and decrease nutrient loss from the fields (Ghebremichael et al., 2013; Tomer et al., 2013, 2015). These tools offer an assortment of management options to reduce nutrient loss to streams while providing the flexibility needed for farmers and land managers to maintain crop productivity and yields.

The third step is operationalization and execution of management options. Examples would be stream restoration and using BMPs on the agricultural landscape and within the river channel. The fourth step is monitoring and auditing, which includes measuring the key indicators of achievements to understand the response of the system to natural and management interventions. Long-term integrated monitoring, research, and modeling are all used to track criteria relative to establishing and refining TPCs, which determine whether management action or recalibration of the TPC is needed. Because SAM is an integrative process, the final step is evaluation and learning from the outcomes and applying that knowledge to modify and improve the management plan.

Each step of the process is reviewed to assess (i) if the knowledge gained informs and improves the understanding of the ecosystem, (ii) ecosystem responses, (iii) how realistic the desired outcomes are, and (iv) how useful the processes used to achieve them are (Rogers et al., 2008). Thus, objectives and management options may frequently change as managers learn from the process (Rogers and Biggs, 1999).

Many governmental agencies around the world are already undergoing SAM by implementing programs that attempt to limit eutrophication of their water bodies (Hilton et al., 2006; Kröger et al., 2013). A common element of many of these SAMs is the use of BMPs to increase the resilience of the entire agricultural ecosystem. This includes using BMPs not only on agricultural surfaces, but also within the river channel. Agricultural BMPs can either prevent the ecosystem from crossing the ecological threshold into a degraded state, or they can increase the structure and functioning of the degraded state. They are implemented to reduce nutrient inputs into the ecosystem and to minimize transport of nutrients and sediments to streams (Kröger et al., 2013; Tomer et al., 2013). Targeted BMP use can provide the most efficient nutrient management strategy with the least cost (Sharpley and Jarvie, 2012). Terrestrial BMPs include such things as conservation tillage, water erosion control, buffer strips, and proper fertilizer and manure management through appropriate application rates, timing, and methods (Sharpley and Jarvie, 2012). Proper manure management in confined animal feed lots is important, as manure has a high P content; otherwise, runoff may amount to several kilograms of P per hectare (Smil, 2000). In tile-drained fields, end-of-tile treatments, such as filter cells, cartridges, and structures installed at the drainage outlet (King et al., 2015) and rerouting tiles to flow through riparian buffers (Jaynes and Isenhart, 2014), can remove nutrients before they enter the stream by increasing nutrient processing. Aquatic BMPs include reconstructing drainage ditches to include a two-stage ditch to intercept nutrients and sediments during storms, which can also reduce nutrient flux to streams (Davis et al., 2015), and installation of riparian buffer zones and sediment traps to target legacy nutrients.

Promotion of in-stream nutrient retention may potentially be another aquatic BMP to reduce anthropogenic disturbances, mitigate high nutrient concentrations and loads, and target legacy nutrients. The historic focus of agricultural stream research on the delivery aspects of nutrient flux from agricultural landscapes partly reflects the mistaken belief that streams are unreactive pipes where little nutrient cycling occurs. This assumption is incorrect, as streams are capable of temporarily removing P through sediment burial and biotic uptake and permanently removing N through denitrification (Reddy et al., 1999; Richardson et al., 2004; Kreiling et al., 2011). Macrophytes and phytoplankton assimilate nutrients throughout the growing season, temporarily removing them from the sediment and water column and then rereleasing them as organic N and P during decomposition (Reddy et al., 1999; Birgand et al., 2007). Although there is no permanent biogeochemical removal pathway for P, sediment burial can be long term (P is essentially biologically unavailable) if P is bound to a calcium salt or is in a resistant organic P form (Reddy et al., 1999). Most nutrient retention processes occur in areas with greater transient storage (Sheibley et al., 2014) and commonly in shallow headwater streams, where the sediment surface to water volume ratio is large (Boyer et al., 2006). As stream size and mean stream depth increase, nutrient removal typically decreases (Alexander et al., 2000); however, larger rivers may be more efficient at removing nutrients per unit stream length (Seitzinger et al., 2002). Thus, in-stream nutrient retention is an important ecosystem service that could be actively managed.

To aid managers in implementing SAM, quantitative models such as Spatially Referenced Regression on Watershed (SPARROW) and the Soil and Water Assessment Tool (SWAT) can be used to locate areas for BMP placement and to assess BMP effectiveness (García et al., 2016; Scavia et al., 2017). In the Lake Erie Basin, SWAT has been used to identify areas for BMP placement to target P reduction (Scavia et al., 2017); SPARROW has been employed in the Upper Mississippi-Ohio River Basin to locate areas for BMP placement to reduce N export (McLellan et al., 2015). The SPARROW model has also been applied in the Upper Mississippi River Basin to assess the effects of conservation practices on N and P transport. Results from SPARROW provided evidence that conservation practices were having more of an effect on N loading than P loading (García et al., 2016).

Previous attempts to manage runoff from agricultural ecosystems can be characterized as a command-and-control paradigm, which focuses on technical solutions to well-defined problems (Gleick, 2003; Pahl-Wostl, 2009). For example, riparian mitigation schemes are built to contain runoff impacts on waterways without assessing their effectiveness. Increasingly, alternative approaches to waterway and stream-network management are emerging that are based more on a learning-by-doing process (Parsons et al., 2016). These new approaches, such as SAM, highlight that learning is a critical requisite for dealing with complexities inherent in the management of agricultural ecosystems (Pahl-Wostl et al., 2011). Stream restoration and BMPs are viewed as experiments in which to test and predict functional responses and feedback mechanisms of various interventions. Because many of these BMPs are implemented on private land, government assistance may be needed to offset costs incurred by landowners. Lessons learned from these management actions are then used to develop and refine future management plans (Parsons et al., 2009). Thus, the steps in SAM are frequently repeated.

Example of Application of Framework: Fox River Basin

The above sections outline the conceptual basis of the framework. In this section, we illustrate how the framework can be applied to an agricultural ecosystem. The framework facilitates the translation of scientific and management concepts through adoption and application. In particular it is a means to organize the "why," "what," and "how" components of landscape restoration or management. The integration of an ecosystems approach, resilience thinking, and SAM is necessary because, even though each is strong on its own, when applied to the problem of managing land and water issues in agriculture watersheds, there are gaps in each approach that can be strengthened by principles from the other approaches. Resilience thinking presents a useful social–ecological approach for understanding resilience in ecosystems, but it does not have a strong operational and implementation procedure. Strategic adaptive management provides an excellent operational procedure for managing resilience in ecosystems, but so far it has only been applied to managing ecosystems where biodiversity conservation is the main goal. An ecosystems approach has a strong conceptual and scientific basis but does not have an operational procedure associated with it within a management context. Thus, integration of the principles from each approach provides a powerful and potentially novel basis for the components of a framework for land and water management.

We illustrate the operationalization of the framework with the Fox River basin, which drains into Lake Michigan in eastern Wisconsin. Management actions in the basin are already occurring and form the impetus for this framework. Although management is not following our framework strictly, we provide details of how they are applying some of the key tenets of the framework and how managers may be able to improve restoration efforts by focusing on the entire agricultural ecosystem.

Some form of water quality management of riverine ecosystems has been occurring in the United States since the early to mid-20th century. In the 1930s, the US government enacted programs to reduce topsoil loss from agricultural fields. With the authorization of the Clean Water Act in 1972, local and federal laws have been passed to lessen nutrient and sediment pollution of navigable waters (Fig. 5A). In the 1970s, eutrophication of the Laurentian Great Lakes was recognized as a problem by the US and Canadian governments (GLWQA, 1987). Phosphorus was identified as the main contributor, so target total P loads were established for all five lakes in 1978 (Fig. 5A; GLWQA, 1987).

Green Bay is a large (4210 km^2) shallow bay on the western shore of Lake Michigan, and it receives elevated loads of sediment and P from the Fox River. In 1987, the USEPA identified the last 11.2 km of the Fox River and the lower section of Green Bay (lower 88 km^2) as an "Area of Concern," where significant environmental degradation has occurred. The Lower Green Bay Area of Concern receives ~85% of the total P load into Lower Green Bay (Dolan and Chapra, 2012). More recently, the USEPA has designated the entire Lower Fox River as a priority watershed, where action needs to be taken to reduce nutrient loads (Cadmus Group, 2012). The USEPA mandated that the State of Wisconsin set a total maximum daily load (TMDL) for P in the Lower Fox River. In 2012, Wisconsin set the TMDL at 0.1 mg P L^{-1} (Cadmus Group 2012). This TMDL requirement puts the burden on local municipalities to limit their P point-source discharges to an average P effluent concentration of 0.2 mg P L^{-1} (Vande Hey, 2014).

Total P load into Lower Green Bay has been consistently above the target TMDL of 100 t yr^{-1} for the Lower Green Bay Area of Concern since yearly measurements began in 1974 (Fig. 5B; Lesht et al., 1991; Dolan and Chapra, 2012). Loads decreased from 1974 until 1990, likely due to a decrease in point sources (Lesht et al., 1991), but have been slowly increasing since 1990 (Dolan and Chapra, 2012). Point sources had historically been the main contributor of P in the Lower Fox River Basin (Sager and Wiersma, 1975), but with improvements in wastewater treatment plants (WWTPs) and the ban on P in detergents, point sources currently only contribute ~35% of the load (Cadmus Group, 2012). Instead, nonpoint pollution from agriculture contributes the highest share of the load, at 46% (Cadmus Group, 2012). Dairy farming is the main agricultural practice in the Lower Fox River. The size of the farms has been increasing dramatically since the 1990s (Fig. 5C; USDA, 2017). Although farm size is growing, available agricultural land is shrinking throughout the basin due to urban sprawl. In dairy farms, manure is typically used as the P additive for agricultural soils. In the Lower Fox River Basin, the reduction in available land to sustainably incorporate the manure (Vande Hey, 2014) has resulted in continued increased P loads to the river network, despite efforts by land managers to combat increased land-based P loading.

To reach the target P TMDL, a new management strategy is being implemented in the basin. Managers have already gone through some of the steps of SAM (Fig. 2C). The desired ecological state has been described (i.e., less eutrophic conditions in Green Bay—Step 1 in SAM) and the key TPC has been set by legislation (i.e., P TMDL). The development of the management plans (Step 2 in SAM) has been under way for more than a decade. To target areas for nutrient load reduction, managers first modeled the loads of P and sediment coming from areas in the basin (Baumgart, 2005). That information was used to target optimal areas for nutrient reduction through various BMPs, mainly on the terrestrial landscape (Table 1). Managers identified subbasins that were the main contributors of the P load and developed nonpoint-source implementation plans to reduce P loading from the basins (OCLCD, 2015). These plans identified many different scenarios to reduce P, including areas for wetland restoration, riparian buffer placement, and farm fields where nutrient management plans and other BMPs could be put into place. Costs were estimated for all the different implementation plans, and grants from the federal and state governments have allowed local governments to begin operationalizing some of these nonpoint-source implementation plans (Step 3 in SAM). Some of the areas where BMPs are in use are being monitored to assess the effects of BMPs on water quality in the receiving water bodies (Step 4 in SAM). This information will be used to develop and implement more nonpoint-source pollution mitigation plans (Step 5 in SAM; Merriman, 2015). Basin-wide execution of these proposed plans is based on funding availability and private landowner participation, which potentially can be two key impediments to implementation (Cadmus Group, 2012).

To fully realize the goal of P reduction in the basin, placement of BMPs should be put into a more spatial context within the ecosystems approach (Fig. 2A). Currently, most of the BMPs in the basin are employed in the terrestrial landscape (Table 1). Vegetative buffer zones and wetlands are being placed in some riparian zones, but to a limited extent, and the only river channel BMPs are structural modifications to reduce streambank erosion (Vande Hey, 2014). With current computer modeling capabilities, optimal BMP placement can occur on the terrestrial landscape, in the riparian zone, and in the river channel to attain desired results while keeping

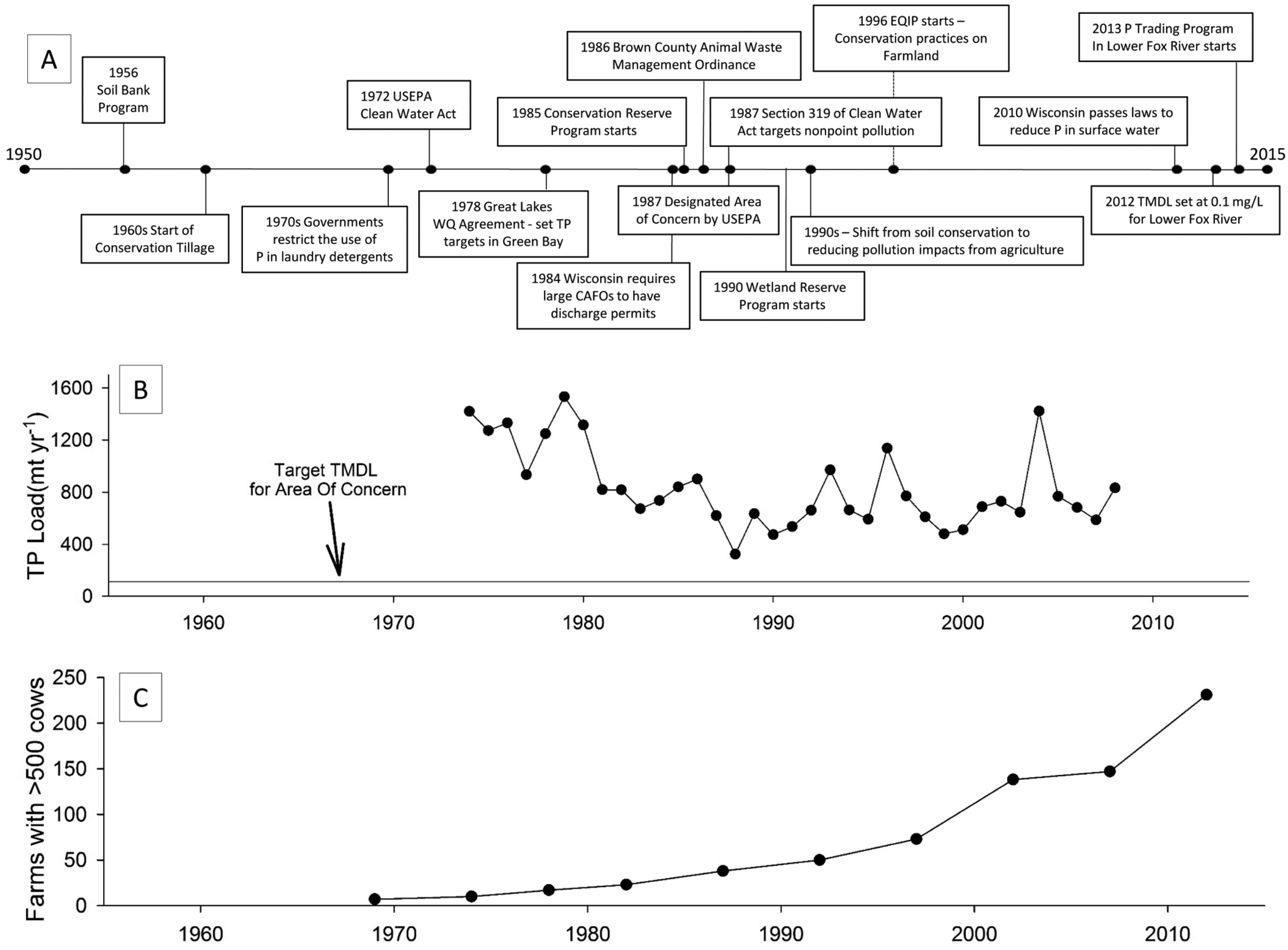

Fig. 5. (a) Timeline of key legislative acts and the implementation of conservation practices in the Fox River Basin from 1950 to 2015. WQ, water quality; CAFO, confined animal feeding operation; EQIP, Environmental Quality Incentives Program. (b) Total phosphorus (TP) load into Lower Green Bay from 1974 to 2008. The arrow is pointing to the target total maximum daily load (TMDL) for the Lower Green Bay Area of Concern. The Lower Green Bay Area of Concern receives ~85% of the TP load entering Lower Green Bay. Total P load data are from Lesht et al. (1991) and Dolan and Chapra (2012). (c) Number of farms in the Fox River Basin that have >500 cows. Data are from the USDA.

costs down (Tomer et al., 2003; Tomer et al., 2014). The targeted placement of various BMPs throughout the ecosystem would also increase resiliency in the basin, because the BMPs act as buffers to reduce the chances of the system from crossing the threshold into a further degraded state (Fig. 2B). Examples of BMPs that could be installed include those in Table 1, as well as end-of-tile treatments, two-stage ditches, and promotion of in-stream nutrient and sediment retention. Once the system is restored, BMPs can be used to enhance the structure and function of the novel ecosystem. Thus BMPs are more than just a preventive strategy to reduce degradation from agricultural disturbances; they are also an enhancing strategy to increase ecosystem functioning. By placing a variety of BMPs throughout the agricultural ecosystem, we suggest that managers can reduce the probability of the ecosystem encountering a negative ecological surprise (Doak et al., 2008), where unintended harmful consequences occur due to management actions.

In one of the Fox subbasins, some of the framework concepts are employed. Point-source polluters in the Fox River basin have been mandated to lower the P concentration in the river below where they discharge. Treatment plants were given three options: upgrade of existing WWTPs; water quality trading, which requires pollution reductions from other sources such as agriculture, urban storm water, or other WWTPs; or adaptive management of the watershed in which the WWTP resides (Vande Hey, 2014). Plants evaluated the potential cost of all these options and decided the best option for each of them. In 2014, the WWTP in the city of Green Bay, Wisconsin chose the adaptive management process and have started a small pilot project in a small stream (Silver Creek) in the Duck Creek basin which is adjacent to the Fox River basin that is on Native American land. The plant is taking an ecosystems approach (Fig. 2A) and is collaborating with indigenous leaders, farmers, land managers, river managers, and researchers to implement and then monitor the effectiveness of BMPs. The goal is to reduce nutrient and soil run-off from farm fields to improve water quality and fish habitat in the small local stream. Currently, most of the BMPs are being installed on the land and not in the stream, but some stream restoration has been undertaken to reduce streambank erosion (Vande Hey, 2014). In the context of resilience thinking (Fig. 2B), stream restoration and placement of BMPs on the landscape are making the agricultural ecosystem more resilient to the disturbances caused by agriculture. Because this is a pilot study, information learned may inform the SAM process in other watersheds.

Table 1. Best management practices (BMPs) currently implemented in the Fox River Basin, which drains into Lake Michigan from eastern Wisconsin.

BMP	Function	Location	Biogeochemical benefit	Biogeochemical risk
Barnyard runoff control	Practices such as animal waste storage and runoff management systems to limit nutrient runoff from barnyards	Barnyard	All N and P forms and sediment trap	
Nutrient management plans	Target fertilizer application to meet plant needs at the best time to reduce nutrient loss	Farm fields	All N and P forms	
Conservation tillage	Soil is covered with crop residue or vegetation to reduce soil erosion and nutrient loss	Farm fields	Particulate N and P and sediment trap	Export of dissolved P and nitrate to groundwater and streams
Cover crops	Crop that is planted after cash crop is removed	Farm fields	Particulate N and P and sediment trap	Export of dissolved P to groundwater and streams
Conservation crop rotation	6 yr rotation of 3 yr of corn (*Zea mays* L.) silage with fertilizer application followed by 3 yr of alfalfa (*Medicago sativa* L.) or hay without fertilizer application	Farm fields	All N and P forms and sediment trap	
Grassed waterway	Protect soil from concentrated flows by having permanent vegetation in channel to reduce gully erosions	Farm fields	Particulate N and P and sediment trap	
Sediment control basins	Traps soils and nutrients before they reach the river	End of storm water diversion ditch	Particulate N and P and sediment trap	Potential legacy P storage area
Forested and grass riparian buffers	Trap soil and nutrients before they reach the river	Along the streambank	Nitrate and particulate N and P	Potential legacy P storage area
Wetland restoration	Reduce sediment and nutrient transport and remove sediments and nutrients from water column	Riparian buffer zones and edge of tile drains	Total N and total P and sediment trap	Export of dissolved P and ammonia to the stream and release of greenhouse gases (NO_x, CO_2, and CH_4)
Streambank stabilization and erosion control	Limits erosion of streambank	Stream	Particulate N and P and sediment trap	

Conclusion

The framework presented here provides a road map for the management of nutrients in agricultural ecosystems. The framework is composed of three parts: an ecosystems approach, resilience thinking, and strategic adaptive management. The ecosystems approach calls for the integrated management of nutrients on the agricultural landscape and in the stream network. It stresses that managers should have an awareness of how management practices on the agricultural landscape affect the stream network. Resilience thinking recognizes that because ecosystems can have more than one stable state, anthropogenic disturbances to streams may need to be maintained at a level lower than the ecological threshold to keep the stream in a particular state. As a result, many restored streams are actually in a novel state that functions differently from either a natural or a degraded state due to continued anthropogenic disturbances. Because both resilience thinking and the ecosystems approach are conceptual in nature, SAM provides the operational procedure for this framework. It consists of five interacting steps that allow managers to learn by doing. Best management practices and stream restoration are viewed as experiments that are tested, and knowledge gained from these actions is used to shape future management plans. Thus, the integration of the ecosystems approach, resilience thinking, and SAM provide a sound framework for nutrient management.

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