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The Latitudes, Attitudes, and Platitudes of Watershed Phosphorus Management in North America

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Abstract

Phosphorus (P) plays a crucial role in agriculture as a primary fertilizer nutrient—and as a cause of the eutrophication of surface waters. Despite decades of efforts to keep P on agricultural fields and reduce losses to waterways, frequent algal blooms persist, triggering not only ecological disruption but also economic, social, and political consequences. We investigate historical and persistent factors affecting agricultural P mitigation in a transect of major watersheds across North America: Lake Winnipeg, Lake Erie, the Chesapeake Bay, and Lake Okeechobee/Everglades. These water bodies span 26 degrees of latitude, from the cold climate of central Canada to the subtropics of the southeastern United States. These water bodies and their associated watersheds have tracked trajectories of P mitigation that manifest remarkable similarities, and all have faced challenges in the application of science to agricultural management that continue to this day. An evolution of knowledge and experience in watershed P mitigation calls into question uniform solutions as well as efforts to transfer strategies from other arenas. As a result, there is a need to admit to shortcomings of past approaches, plotting a future for watershed P mitigation that accepts the sometimes two-sided nature of Hennig Brandt's "Devil's Element."

Core Ideas

- North American P mitigation experiences spanning 26 degrees of latitude are explored.
- Uncertainty and lag times create schisms in mitigation programs.
- One-size-fits-all approaches cannot account for management trade-offs.
- Acknowledging shortcomings in messages and approaches is imperative to moving on.
- Unified P mitigation provides a framework and eliminates solution singularities.

The discovery of phosphorus (P) by Hennig Brandt in 1669 and its subsequent use in fertilizers helped to fuel the 20th century's Green Revolution, overcoming fundamental constraints to crop production and enabling crop yields that are the basis of the inexpensive, reliable food supply of the modern world (Jarvie et al., 2015; Sharpley et al., 2018). However, as P use has expanded and the cost of P has declined, its presence in the environment has fueled eutrophication that is now one of the most ubiquitous and persistent water quality concerns facing humankind (Schindler, 1977; Carpenter et al., 1998; Schindler et al., 2012; Ulen et al., 2012). The obstacles to mitigating eutrophication are manifold, not the least of which are economic, challenging science and its application, pitting agendas in opposition, and tripping up simple solutions.

For agriculture, eutrophication mitigation has become synonymous with addressing diffuse P sources across watersheds (i.e., watershed P mitigation). The principal challenge to watershed P mitigation is the contextual nature of the problem and its solutions. Simply put, one size does not fit all. Although there are many commonalities in the sources of P, processes of mobilization and transport, and management options, the interaction of climatic, biogeochemical, and anthropogenic variables necessitates mitigation strategies that adapt to local conditions. Understanding these variables is key to developing strategies that are applicable to local settings, capable of responding to changing knowledge and conditions, and reasonable in balancing expectations of different stakeholder groups.

The extreme differences in climate, physiography, and management systems found across latitudes highlight the limits to standardized P mitigation approaches. Across latitudes, nonpoint source P transport is generally dominated by storm flow, be it snowmelt processes on seasonally frozen ground in northern regions, saturation excess runoff processes from soils with saturated antecedent conditions, or infiltration of excess runoff following convective storms in the south. Further, cropping, nutrient management, and drainage systems reflect the interaction of this climate with local physiography. Different mitigation strategies are needed to account for these contrasting conditions

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Abbreviations: BMP, best management practice; TMDL, total maximum daily load.

(Ulen et al., 2012; Baulch et al., 2019). Our ability to recommend and employ P mitigation efforts is further challenged by the fact that we are working toward a "moving target," where temperatures are increasing and the hydrologic cycle is intensifying (Ockenden et al., 2017). These changes, which differ across latitudes, may affect when, where, and how P is redistributed within the landscape. Thus, to mitigate P loss, we must consider how to minimize not only current P transport offsite but also future P losses.

Attitudes toward addressing water quality concerns have changed in response to the recognition of the importance of nonpoint sources of P, and these changes have occurred at many levels. In the United States, the 1985 Farm Bill marked a significant change in policy that required compliance with conservation requirements to be eligible for commodity subsidies (Cain and Lovejoy, 2004). Our understanding of watershed P sources has also evolved over the last several decades. In the mid-20th century, as animal production was dispersed, animal manures were viewed as an excellent source of P (plus organic matter and other nutrients) across the landscape. However, concomitant with the evolution of confining sectors of the animal industry to specific regions, rates of manure application exceeded crop removal rates, leading to excessive levels of surplus P in soils. Manure-related P issues are such a problem in some regions, that interested parties sometimes blame manure for water quality issues in other regions where animal populations are insufficient to produce excessive levels of manure P. There are also ongoing discussions related to soil fertility. Current fertility guidance was largely developed in the mid-20th century based on agronomic assessments with ample safety margins built in so that guidance covered large geographic regions. Since P fertilizer was viewed by agronomic professionals as cheap "insurance" against crop yield loss, they recommended that farmers apply P at ample rates for decades. Many of these farmers continue to apply P, even very high rates of P fertilizers, when the current fertility recommendation is for zero application (Smith et al., 2018). Some of these farmers were told many years ago by extension personnel or crop advisors to apply P at rather high rates annually, regardless of soil tests, and continue to do so. This multi-decadal mantra has such endurance that it is now difficult to convince some producers that they can get by with less (or even zero) P.

Platitudes that oversimplify messages concerning the causes of P losses and approaches to mitigating nonpoint sources of P loss can exacerbate efforts to improve water quality. A better embrace of the complexity of the processes of P loss and understanding of trade-offs in conservation strategies regarding P loss is needed to improve the relevance of programs to P mitigation (Kleinman et al., 2015). For example, soil health initiatives must come to terms with the consequence of severe vertical stratification of P in soils that can occur after several years of tillage exclusion (Baker et al., 2017; Smith et al., 2017). "Feed the soil to feed the plant" (Withers et al., 2014) is gradually being recognized as cost effective but inherently inefficient with regard to P use efficiency. Nutrient management strategies are in dire need of updates to recognize the need to balance crop demand with environmental risks. Applications of "pop up" or "starter" P fertilizers to high P soils are another impediment to reducing excessive soil P levels, but researchers have not offered alternatives for replacing the practice other than "stop adding P." Integration of social sciences with biophysical research is needed to better frame P mitigation science and its outcomes.

With a goal of elucidating major factors related to the success and failure of watershed P mitigation efforts, this paper (i) reviews the influence of climatic drivers on P fate and transport across the latitudes of North America; (ii) identifies past and future changes in attitudes to nutrient management that will move us closer to improvements in water quality; and (iii) discusses the importance of moving away from platitudes if we are going to successfully reduce P loads from agricultural watersheds and minimize the ecological, social, and economic costs associated with freshwater eutrophication.

Watershed Phosphorus Concerns across Latitudes

For roughly the past half-century, P-based eutrophication has been a concern to Lake Okeechobee (1890 km^2 lake area; 10,000 km^2 watershed area), the Chesapeake Bay (11,600 km^2; 167,000 km^2), Lake Erie (25,700 km^2; 78,000 km^2), and Lake Winnipeg (24,500 km^2; 983,000 km^2; Fig. 1). Efforts to understand and curb this eutrophication describe the arc of P mitigation in North America, evolving from an initial emphasis on point sources. In both Lakes Winnipeg and Erie, binational water quality agreements have been required because the lakes receive inputs from watersheds in both the United States and Canada, whereas the other water bodies have relied on federal and state policies. During the early stages of P mitigation, the first policies emerged to reduce the use of P in detergents and drive improvements to

Fig. 1. The latitudes, attitudes, and platitudes of P mitigation for key watersheds in North America: Lake Okeechobee, the Chesapeake Bay, Lake Erie, and Lake Winnipeg.

septic systems and wastewater treatment. These point source achievements were followed by efforts to identify and pursue nonpoint sources, including an initial emphasis on soil conservation and land retirement programs for agriculture, followed by comprehensive nutrient management efforts to tackle nonpoint sources of P. As nonpoint source P programs have matured in these watersheds, the scope of nutrient management has broadened from agriculture to include other major land management areas, including managed turf.

Lake Okeechobee, situated in the subtropical region of Florida, averages 2.7 m in depth and has been the subject of eutrophication-related restoration efforts since the 1970s. Agriculture (primarily sugarcane [*Saccharum* spp.], improved pasture, and citrus, as well as dairy), wastewater treatment, and urban runoff are the primary sources of lake P loadings, as are internal cycling of legacy P between lake sediments and the water column. Despite more than four decades of best management practice (BMP) implementation, P loss from agricultural lands remains a major concern. The US Clean Water Act provides the larger regulatory backdrop for Lake Okeechobee, under which there is a total maximum daily load (TMDL), buoyed by Florida's own Lake Okeechobee Protection Plan of 2000 (Zhang et al., 2008). Mitigation activities for agriculture emphasize adoption of BMPs and improved drainage water management, with documentation of all activities critical to efforts to understand and predict outcomes. Dissolved P release from the flat, endo-saturated Spodosols of the region has focused attention on thresholds of soil P sorption saturation as a means of capping fertilizer application rates (Dari et al., 2018).

The Chesapeake Bay watershed, spanning coastal plain and Appalachian mountain regions of the temperate, mid-Atlantic United States, averages 7 m in depth and includes parts of six US states, as well as Washington, DC. The 2010 TMDL to restore Chesapeake Bay health places limits on P, nitrogen (N), and sediment delivered to the bay. As with Lake Okeechobee, agriculture, wastewater treatment plants, and urban runoff are major sources of P, and internal cycling of P has been identified as important, including in the "dead zone" that forms annually within the central channel of the bay. Mitigation of P has kept pace with TMDL goals, with all jurisdictions but Pennsylvania meeting planned implementation of remedial practices. Monitoring across the bay watershed points to mixed success, with troubling increases in dissolved P reported in many agricultural tributaries (Moyer and Blomquist, 2017). Legacy P, critical source areas, artificial drainage, trade-offs in conservation programs, farmland conversion, and manure management all contribute to a complex set of challenges, many of which are difficult to achieve under voluntary programs (Kleinman et al., 2019).

Lake Erie had long been a beacon of P mitigation success, with major reductions in total P loadings in the 1970s and 1980s achieved by a combination of improved wastewater treatment, elimination of phosphates in detergents, and extensive soil conservation efforts. As an international water body, the Great Lakes Water Quality Agreement between Canada and the United States has played a critical role historically both for P mitigation and science (International Joint Commission, 2009; Joosse and Baker, 2011). Since the 1990s, however, agricultural watersheds in the Western Lake Erie basin have emerged as primary sources of dissolved P. Artificially drained croplands are considered the main source of this P, with drainage intensity continuing to increase. Land management reporting for the Sandusky and Maumee watersheds in the United States indicate that cropland under high-residue management increased from <10% in 1985 to 70 to 80% in 2007 (Jarvie et al., 2017). As a result, the interaction of soil conservation programs with P cycling has been a central focus in recent years, raising concerns over the unintended consequences of strategies aimed at controlling sediment-bound P.

The Lake Winnipeg watershed encompasses roughly 1 million km^2 across the provinces of Alberta, Saskatchewan, Manitoba and Ontario in Canada, as well as the US states of North and South Dakota and Montana. As with the other water bodies discussed here, Lake Winnipeg is relatively shallow (7 m in depth) and has seen a gradual shift in its trophic status, marked by greater severity of cyanobacterial blooms. In the immediate area of the lake, flat landscapes dominate, with low gradient drainage a primary concern. Over the past three decades, nutrient loads have increased by approximately 10% (Roy et al., 2007). The Red River, which meanders through fertile farmland in both the United States and Canada, is the principal source of P to Lake Winnipeg; however, urban and wastewater sources also contribute significantly to P loads. Mitigation activities are complicated by Lake Winnipeg's enormous watershed.

The large watersheds examined in this paper are subjected to significant differences in climate drivers, which interact with geomorphology and management to present unique hydrologic conditions favoring P loss in each watershed. At the southern extreme (Lake Okeechobee), rainfall is concentrated in the summer months and winter precipitation is limited (Finkl, 1995). In this region, runoff responses are rainfall induced due to the warmer temperatures, and most runoff and nutrient transport occurs via groundwater (Finkl, 1995; Muñoz-Carpena et al., 2005). The temperate Great Lakes and Chesapeake Bay watersheds have climates that are intermediate between the northern and southern extremes, with precipitation evenly distributed throughout the year, as a mixture of snow (nival) and rainfall (pluvial) events of varying intensity. In comparison, runoff from the frigid Lake Winnipeg watershed occurs primarily during snowmelt over frozen soils in the spring. Given differences in climate drivers, the processes controlling P mobilization and the pathways through which P is transported may differ. Consequently, the adoption of P mitigation strategies that are appropriate to the prevailing climate conditions of a region is critical.

Hydrology, Artificial Drainage, and Phosphorus Loss

It is often noted that variations in hydrology within a landscape can overwhelm variations in P sources (Buda et al., 2009). While nearly all the watersheds featured here include substantial areas where subsurface P transport is a concern, historically, surface runoff has been the primary pathway of concern for P. It is useful to distinguish between saturation excess runoff (Dunne and Black, 1971), in which certain areas of the landscape are waterlogged at the onset of a storm and therefore have negligible capacity to absorb rainfall, and infiltration excess runoff (Horton, 1933), understanding that these processes define different ends of a continuum. Horizontal discontinuities within soils (e.g., hard pans, frost lenses, textural contrasts) can all serve to perch water tables, supporting conditions that favor saturation excess runoff (Kane and Stein, 1983; Zobeck and Ritchie, 1984). The role of saturation excess runoff in P transport has been described in the uplands of the Chesapeake Bay and Lake Erie watersheds (Needelman et al., 2004; Macrae et al., 2010) and in the lowlands of the Lake Okeechobee watershed (Heatwole et al., 1987). It has also been observed in the Lake Winnipeg watershed, although infiltration excess overland flow is more common. Even when soils prone to saturation excess runoff generation are a minority of the watershed, they can dominate as sources of watershed P export. In these areas, there are few management options, short of introducing artificial drainage, to modify this hydrology, which is based on landscape processes. Source management, principally following the "4R" principles for nutrient stewardship (Right source, Right rate, Right time, Right place), provides the most comprehensive P mitigation strategy (Vollmer-Sanders et al., 2016; Bruulsema, 2018). This may contrast with the emphasis of soil conservation programs that may be focused on well-drained soils, where management of infiltration properties is a priority (e.g., building soil organic matter and residue cover).

Snowmelt and runoff over frozen or partially frozen soils are important, if not primary, processes for P transport in the northern watersheds, from the Chesapeake Bay to Lake Winnipeg. Indeed, runoff and P mobilization in the Lake Erie and Lake Winnipeg watersheds primarily occur throughout the nongrowing season (Macrae et al., 2007; Ford et al., 2018; King et al., 2018; Plach et al., 2019) and is most often dominated by spring snowmelt (Macrae et al., 2007), although large-magnitude events can occur at other times of year, including the summer months (Macrae et al., 2010; Van Esbroeck et al., 2017). Mitigation strategies that are effective under rainfall-dominated conditions during the growing season have not been successful during the winter period (Baulch et al., 2019; Kieta et al., 2018). Around Lake Winnipeg, spring snowmelt over frozen ground represents the dominant P transport pathway, with a shallow frost lens impeding infiltration and therefore rendering ineffective strategies to improve soil infiltration, as described above, or trapping strategies that require infiltration (e.g., riparian buffers zones). Management practices to mitigate P loss include storage of snowmelt runoff in on-farm reservoirs, avoiding P application during winter months, and drawing down soil P concentrations (Tiessen et al., 2011; Liu et al., 2018, 2019).

Artificial drainage plays a critical role in agricultural production in all the watersheds discussed here, particularly in flat coastal and lake plain areas where regional water tables may saturate soils, preventing field access and crop growth. There is little doubt that intensifying drainage can improve productivity when soil moisture excess is a concern, but debate persists over the trade-offs of introducing drainage on P loss (King et al., 2015). On one hand, artificial drainage may serve to reduce periods of saturation excess surface runoff. On the other, some of the largest losses of P within all four of the featured watersheds are associated with export from tile drains and open ditches (Campbell et al., 1995; Kleinman et al., 2007; King et al., 2015). Artificial drainage introduces quickflow, which reduces residence time and interactions between flow with soil P sorption–desorption controls. Indeed, for the transport of P, tile drainage is often considered to carry the concentration–discharge signature of surface runoff (Radcliffe et al., 2015; Smith et al., 2015a). Where open ditches are the principal conveyance, P leaching processes are a concern and P sorption saturation a controlling factor, as documented in the coastal plain of the Chesapeake and Lake Okeechobee watersheds. When fields are internally drained (e.g., low areas within fields where surface water would normally collect), artificial drainage introduces hydrologic connectivity (Smith et al., 2008). Mitigation strategies include the use of reactors to sorb P (in-line with drainage or intercepting flows before they are drained), control of flows (drainage water management), and lessening bypass flows such as with "blind inlets" (Buda et al., 2012; Smith and Livingston, 2013; Williams et al., 2015).

Manure and Fertilizer Management

All watersheds reviewed here include animal production in varying concentrations, including poultry (Chesapeake Bay, Lake Winnipeg), dairy (Lake Okeechobee, Chesapeake Bay, Lake Erie, Lake Winnipeg), and swine (Chesapeake Bay, Lake Erie, Lake Winnipeg). High concentrations of animal production are a product of the efficiencies associated with "economies of scale" that come with locating farms near processing facilities to minimize transportation costs, among other things. These concentrations can change rapidly: new poultry production has been established in the Pennsylvania portion of the Chesapeake Bay, and new growth of swine production is ongoing in the Michigan and Ohio portion of the Western Lake Erie basin. In regions such as the DelMarVa Peninsula, Lancaster County and Shenandoah Valley of the Chesapeake Bay, and the Grand Lake St. Marys region of the Western Lake Erie basin, high concentrations of animal production are a primary concern for water quality impairment. In the early 20th century, when most farms were diversified with a mixture of crops and animals, the P cycle was tightly coupled on the landscape. However, as we have decoupled crop production with animal feeding operations, we have transitioned to a broken P cycle with arches that span vast distances across North America (Fig. 2) or even over multiple continents, as some of the P fertilizer we use comes from Africa and much of the food we produce is consumed internationally. Most notable to this discussion, high concentrations of animals are tied to legacy P in soils (expanded on below), which is one of the most insidious problems facing P mitigation programs.

Although manure is undoubtedly a fertilizer resource, it is an understatement to note that managing manure is different than managing fertilizer. Unlike mineral fertilizer, manure possesses liabilities that limit its transport, reduce its shelf life, and therefore reduce the value of its associated fertilizer nutrients. Manure is odorous, contains pathogens, and because of low nutrient concentrations, is expensive to store, handle, and land apply. Nutrient concentrations within manures are somewhat static, and thus custom blends cannot be adjusted to N and P concentrations independent of each other, as can be done with commercial fertilizers. Further, manure management is inextricably connected to many aspects of a farming operation, from animal feeding to cropping systems to farmstead infrastructure. As a result, P management associated with manure is much more complicated than fertilizer management. Aspects of this dynamic are connected to grazing operations, too, a major concern of the permanent pastures of the Lake Okeechobee watershed.

Fertilizer management is a driving concern in all watersheds, although the focus is currently most acute in Lake Erie. At one level, there is concern that established standards for fertilizer use are met. The 4R certification programs that have been pioneered

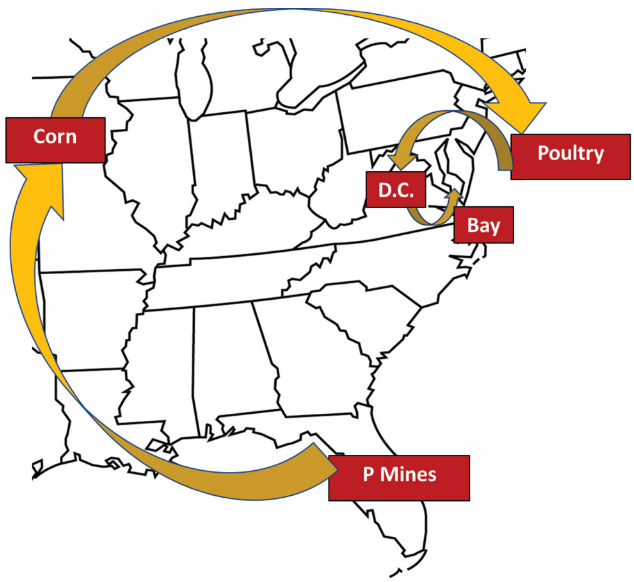

Fig. 2. Once animal production became highly specialized and regionally isolated, the on-farm P cycle was broken. Today's continental-scale P cycle resembles a series of arches (phosphorescent arches), where P is transported from mines in Florida (or other countries such as Morocco) to the grain-producing areas, such as the midwestern United States. Phosphorus is then transferred in the grain to where the animal production facilities are, such as the poultry production facilities on the DelMarVa Peninsula. Finally, poultry facility cleanouts and litter application to nearby croplands transfer P to soils, where it accumulates in great excess, and then a portion of P is transferred to the Chesapeake Bay via dissolved and particulate P loss in runoff via an artificial ditch and tile drainage network that connects sources to sensitive water bodies. Poultry meat, and the P contained therein, produced on the DelMarVa Peninsula is consumed in large metropolitan areas, such as Washington, DC. Some of the consumed P will wind up in the municipal wastewater treatment plants, and a small fraction of that amount will be delivered to the rivers and ultimately the Chesapeake Bay.

in the Lake Erie region, a voluntary program developed by the fertilizer industry, arose as a response to concerns that local land grant recommendations were not being followed. High rates of application and marketing of unproven additives were justified on the suspicion that land grant recommendations, developed over the past 50 years, were inadequate for modern yield expectations, or by personal observations that suggested P deficiency at some point in the growing cycle (even if they were not borne out by harvest yields). The 4R program has emerged as a significant international umbrella under which P fertilizer mitigation can be implemented.

The Critical Role of Peak Flows in Phosphorus Transport

Phosphorus mobilization in the landscape is largely episodic, associated with storm and/or melt events, and the quantities of P flushed increase with event size (Macrae et al., 2007). Numerous studies, undertaken across latitudes and regions, have shown that peak flow conditions dominate annual nutrient loads to surface water bodies (Robertson, 1997; VĘĘnĘnen et al., 2006; Macrae et al., 2007; Richards et al., 2008). Although the role of peak flows in annual nutrient budgets is important across latitudes, the scale of peak flows and the drivers of such conditions vary. For example, in the more northern regions (i.e., the Canadian portion of the Great Lakes and Lake Winnipeg regions), the accumulation of snow during the dormant winter months, followed by snowmelt, creates "peak" flow conditions in springtime (Macrae et al., 2007; Corriveau et al., 2011), and the magnitude of such peak conditions can differ with antecedent moisture, snowpack development, and the occurrence of rainfall during melt. Spring rain storms on wet antecedent soil conditions can also produce significant P loads (Macrae et al., 2010; Kokulan et al., 2019). Further, regions such as the Lake Winnipeg watershed and the Great Lakes are prone to periodic large-magnitude summer and autumn events that can generate substantial runoff volumes when drainage systems become overwhelmed (Macrae et al., 2007; Shook and Pomeroy, 2012; Rattan et al., 2017). In contrast, although snowmelt runoff is less important in the Chesapeake Bay watershed and absent in the Okeechobee watershed, these watersheds are prone to peak flow conditions following significant summer and autumn rainfall produced by warm, moist air masses that often originate over the Atlantic Ocean. In the coastal and southern regions, such storms (Correll et al., 1999; Zhang et al., 2013) and the periodic occurrence of hurricanes (Miller et al., 2006; Jin et al., 2011; Hirsch, 2012) can substantially exacerbate water quality issues through the connection of large P sources such as critical source areas in the watersheds or tributaries (Gburek and Sharpley, 1998; Zhang et al., 2013) or the resuspension of sediments and other organic material in adjacent wetlands or tidal areas (Axelrad et al., 1976; Newman and Pietro, 2001).

Given that the specific processes governing peak flows vary across latitudes, appropriate mitigation strategies may also differ. However, it is critical that P mitigation practices are able to address P loss during peak periods, as P mitigation strategies are undermined when they are unable to address peak flows. Comprehensive P management must also address P loss under lower flows, particularly in intensively drained systems. Thus, effective P mitigation strategies must be able to manage both large and small flow conditions. Such strategies are unlikely to include a one-size-fits-all strategy but rather will be a suite of practices that may be operational under different conditions. One example can be found in the treatment of runoff to remove P. The treatment of runoff under peak flow conditions is impractical because such systems are often overwhelmed under peak flows and thus not functioning when it is most important to do so; however, under lower flows, treatment systems have the potential to be more effective.

Climate Uncertainty and the Challenge of Moving Targets

Mitigation efforts are challenged by a nonstationary climate that is creating a "moving target" in which eutrophication will be minimized. Even as we improve land management and the supply of P in the landscape declines (Powers et al., 2016), the hydrologic transport mechanisms are changing. These changes are anticipated to significantly increase the movement of P in the landscape (Ockenden et al., 2017). Climate change is creating warmer temperatures, particularly in winter, lessening the extent and duration of frost and decreasing the ratio of precipitation that falls in the form of snow in colder regions (Dumanski et al., 2015; Verma et al., 2015). Warmer temperatures may decrease the extent and duration of frozen soil, whereas the lack of snowpack accumulation may increase the incidence of frozen ground. Such changes are likely to influence the pathways through which runoff and P travel (i.e., surface or subsurface), thereby affecting both P loads and the timing of these loads.

Climate change is also expected to lead to an intensification of the hydrologic cycle, with greater rainfall volumes and intensities, but also greater periods of drought between rainfall events (Najjar et al., 2010; Verma et al., 2015; Dumanski et al., 2015; Wang et al., 2018). Changes in the timing, form, and intensity of precipitation are already being noted in some regions (Shook and Pomeroy, 2012; Wang et al., 2018; Baulch et al., 2019). Thus, larger contrasts between wet and dry conditions and greater potential for peak flow conditions following intense rainfall are expected. Collectively, these can affect the way water is stored, used, and transported through the landscape and may affect the appropriateness of mitigation strategies. Thus, scientists, managers, and farmers, when selecting mitigation strategies and making management decisions, must consider both present and future conditions.

Not only is climate change influencing P loads from contributing areas, but it is also affecting the responses of surface water bodies to nutrient loads. Michalak et al. (2013) highlighted the coupling of elevated P loads from the Lake Erie watershed with ideal climate conditions for algal productivity to create a substantial bloom in Lake Erie, including warmer water temperatures and a longer growing season. Such warm and mild conditions may occur more frequently under a changing climate, making surface water bodies more vulnerable to eutrophication, particularly in northern latitudes. In other words, even if P loads from watersheds are reduced due to mitigation strategies, the benefits of these reduced loads may be offset by ecological responses that are climate driven. Drought also reduces river flows and velocities, leading to greater in-channel water retention time, higher water temperatures, and increased light levels, which can increase algal blooms in rivers and lakes (Whitehead et al., 2009). In the coastal areas of the United States, the Chesapeake Bay and Everglades are increasingly under threat by rising sea levels that may readily connect P sources to waterways, exacerbating water quality issues (Najjar et al., 2010; Saha et al., 2011). These areas are also prone to increased risk of extreme weather and hurricane activity (Najjar et al., 2010), which can lead to greater incidences of peak flow conditions.

A Patchwork System for Action

Across these watersheds, water quality is regulated both at the national level and within individual provinces and territories or states. At the national scale, examples include the US Clean Water Act and the Canadian Environmental Protection Act. At smaller scales, water quality legislation is also in place. For example, the Canadian provinces of Ontario, Manitoba, and Quebec all have nutrient management legislation, as water quality governance in Canada is shared by federal and provincial bodies. Although water quality regulations exist across North America, they do not target P specifically. This approach differs from the more targeted approaches adopted by the European Union, such as the Water Framework Directive, which requires that EU member states determine type-specific nutrient criteria to support "good ecological status" of surface waters (Poikane et al., 2019).

Indeed, the regulation of P specifically has not occurred as of yet, although significant efforts have been made to mitigate P through a combination of voluntary programs and legislation. For example, within the state of Ohio, two recent pieces of legislation were signed into law to address fertilizer management. The first was Ohio Senate Bill 150, which requires anyone who applies fertilizer to more than 20 ha be certified every 3 yr. This law addresses only commercial fertilizer sources and does not address manure fertilizer sources. However, since large volumes of manure fertilizer originate from confined animal feeding operations in Ohio, it is already regulated. The second piece of legislation (Ohio Senate Bill 1) restricts the application of manure and commercial fertilizer on frozen, snow-covered, or saturated ground. Within the province of Ontario, the application of manure during winter and/or on frozen ground is also strongly discouraged as the entry of manure into surface water is an offense under the Environmental Protection Act, the Ontario Water Resources Act, and the Fisheries Act.

While erosion standards were originally the major tie to control P, the emergence of a joint agreement by the USEPA and USDA, followed by the promulgation of Conservation Practice 590 (Nutrient Management) standard, served as the first national rubric aimed at improving P management, resulting in widespread development of the P index by states (Lemunyon and Gilbert, 1993; Sharpley et al., 2003). The P index provides a means of assessing fields for risk of P loss across a farm, but there have been concerns over differences in its development and deployment across state lines and its efficacy in promoting management change (Osmond et al., 2012; Sharpley et al., 2012). Furthermore, it is ineffective in regions in which there is regional P excess as there are few options to divert manure applications out of the impacted area. Despite these challenges, P indexes serve as risk assessment tools that under best conditions, educate managers and provide ways to link nutrient management programs to farm activities. In Canada, various federal–provincial–territorial policy frameworks for agriculture since 2005 have provided cost share funding for agri-environmental programs to farmers through provincially developed programs to meet the needs of farmers in each jurisdiction. Several provinces have, in turn, adopted versions of the P index and other priority setting tools (e.g., Environmental Farm Plan, Farmland Health Check-up) to target mitigation activities. In addition, there are binational agreements (e.g., Great Lakes Water Quality Agreement, Boundary Waters Treaty) in place to guide water quality mitigation efforts.

Ohio agribusiness recently came together with the fertilizer industry to develop the 4R Nutrient Stewardship and Certification program, which is focused on the right source, rate, time and placement of fertilizer. This is a voluntary program that to date has 41 certified retailers within the Lake Erie watershed that manage fertilizer on approximately 770,000 ha of cropland. The success of this program in Ohio among other locations was great enough that 4R Nutrient Stewardship was promoted through the 2018 Farm Bill. Ontario has also adopted a 4R Nutrient Stewardship Certification program for retailers modeled after the Ohio example.

Voluntary BMPs are at the center of most P mitigation programs for agriculture in the United States and Canada. In the United States, the 1985 Farm Bill tied farm income support to conservation adoption by producers (Cain and Lovejoy, 2004). Since then, conservation titles in the Farm Bill have expanded conservation technical and financial support. In the Chesapeake Bay watershed alone, it was estimated that nearly US$2 billion was spent between 2011 and 2017 in support of restoration activities by federal ($1.41 billion) and state ($0.57 billion) governments (US Office of Management and Budget, 2017). In Canada, the Canadian Agricultural Partnership was recently established. Within the Lake Erie watershed, the Great Lakes Agricultural Stewardship Initiative (GLASI) and Lake Erie Agriculture Demonstrating Sustainability (LEADS) are examples of provincial programs that were both established under federal–provincial agricultural policy frameworks in Canada to assist farmers with conservation adoption. To help fund BMP adoption by agriculture, nutrient credit trading (point–nonpoint source trading) has been proposed in the Lake Winnipeg watershed and implemented in several states within the Chesapeake Bay watershed, a means of funding agricultural BMPs in the Chesapeake Bay, although very few trades have occurred (Voora et al., 2009; Branosky et al., 2011).

Although a large number of programs have been put in place with the goal of mitigating P loss, programs across North America are fragmented as they are done at binational, national, and provincial/state scales, and they do not target P specifically. It is unclear whether a more holistic approach to P management would be more effective and what the implications of such a program would be for other aspects of agricultural production systems.

Understanding and Addressing Attitudes

The last 50 years have seen considerable evolution in our environmental, nutritional, and agronomic understanding of what constitutes sustainable P management (Jarvie et al., 2017; Sharpley et al., 2018). Despite this enlightenment, it is clear that because of the contextual nature of watershed P processes and management options, we do not have all the answers. Nor, one could argue, do we even know all the questions to ask. In most cases, ready solutions to P mitigation come at great cost, especially to landowners who typically front installation costs, yield losses, and maintenance expenses (Sharpley et al., 2016). Thus, we must consider how the costs of these society-level problems will be addressed and expenses covered (Shortle and Horan, 2017). For instance, in light of changing consumer attitudes in favor of promoting environmental stewardship, food processors are increasingly becoming more interested in the environmental impact through the entire life cycle of the products they take to market (Sharpley, 2018).

Tensions between Producers and Environmental Groups

Given uncertainty, delays between implementation of practices and water quality outcomes, limited mitigation practice adoption, and verification of adoption—all aspects of the fundamental difficulty of implementing watershed mitigation programs (Meals et al., 2010; Jarvie et al., 2013a; Kleinman et al., 2015)—it is perhaps unsurprising that there are continual tensions between stakeholders involved in watershed P mitigation over the frameworks, inducements, and incentives that are needed to affect water quality improvement. Compared with other regions (e.g., Europe) and nations (e.g., New Zealand), P management in North America is spread across a patchwork of legislation and regulation. For instance, some European countries are forming P-related regulations around a circular economy, to codify P recovery and recycling (Schoumans et al., 2015; Smit et al., 2015; van Dijk et al., 2016). Canadian and US legislation and regulations related to P largely focus on the relationship between land use and water quality (Ross and Omelon, 2018). North American producers tend to be averse to what they might consider heavy-handed regulation. Thus, there would no doubt be strong pushback to European-style regulatory frameworks.

Under the divisive conditions that sometimes emerge in watershed P mitigation, decision support systems play a critical role in providing consistent assessment and guidance for P mitigation at multiple scales (e.g., field, farm, and watershed). These systems are key to supporting adaptive management strategies that enable improvement in mitigation based on past experience (Drohan et al., 2019). At their best, risk assessment tools offer a level playing field that agricultural and environmental communities can occupy with trust and confidence. Although there is a plethora of risk assessment tools produced in the research arena, very few translate to the programmatic and practical conditions that are required for effective P mitigation. Those tools that are adopted can be highly effective and play pivotal roles in P mitigation. Examples from the watersheds examined here include the Chesapeake Bay model, Lake Okeechobee Agricultural Decision Support System, and the P index (used in all watersheds). While these systems have all proven their utility, and to some degree, their efficacy, they are by no means universally appreciated and are frequently critiqued for their inferences and for the demands they place on programs. Even so, the advancement of risk assessment tools that can connect science with management, policy, with practice, and water quality outcomes with mitigation practice options, holds the key to satisfying different stakeholder interests.

Although fewer than 30% of US producers participate in government-sponsored programs and practices, substantial evidence in multiple contexts indicates that agricultural producers do in fact adopt conservation practices without government assistance. More than 90% of producers have a positive attitude about taking steps to aid in solving P-related eutrophication issues (Wilson et al., 2014). Reimer et al. (2012a) found that conservation adoption is higher among producers who are concerned with stewardship and understand the myriad of complex issues around the environmental aspects of production compared with those who only considered profit margins on their farm. However, for the rate of conservation practice adoption among producers to truly improve, the financial benefits to the farm and compatibility with current operations must be stressed, in addition to the environmental benefits (Reimer et al., 2012b).

A significant proportion of farmers have experience with government-sponsored conservation programs; however, many do not participate in these programs because of the complexity involved with enrolling in them (Reimer and Prokopy, 2014). Crop advisors are known to be one of the most trusted sources of information for producers, and they could provide a valuable linkage to conservation programs to producers who do not currently participate, thereby expanding the adoption rate of conservation practices (Eanes et al., 2019). A significant fraction of cropland is rented in the United States and Canada. Given the short-term nature of rental agreements and aversion to risk, producers are less prone to conservation adoption on rented land compared with that which they own (Ranjan et al., 2019). As many as 85% of producers use soil testing to guide fertilizer recommendations, although adoption of other nutrient-focused conservation practices, such as variable rate applications or altering the timing of applications, are much less likely to be adopted (Wilson et al., 2014; Ulrich-Schad et al., 2017;). In the Western Lake Erie basin, for example, only 35% of producers delay fertilizer applications to avoid potential wash-off of nutrients by rain storms (Zhang et al., 2016). Educating producers about the on-farm economic impact of nutrients lost from fields to receiving water could be one of the most cost effective methods for improving their willingness to change the rate or timing of fertilizer applications (Wilson et al., 2014, 2019).

It is widely agreed that voluntary solutions are preferred, and, indeed, they are universally the first step in the watersheds reviewed here. The introduction of regulation and litigation can fuel divisiveness between agriculturalists and environmentalists (Sharpley, 2018). As Sharpley (2018) pointed out, water quality litigation related to agriculture began in the 1990s, but the 2010s has been quite litigious, with at least seven lawsuits filed against farmers or groups of farmers. The cost of litigation by both plaintiff and defendant may exceed the cost of conservation adoption within these regions. Thus, logic and economics would suggest that cooperative efforts by members of the public and the farming community could result in conservation practice adoption by producers in watersheds of concern for less cost and divisiveness than proceeding with lawsuits. Further, litigation may delay action as the proceedings play out in court for years or even decades. As P mitigation is highly site specific, regulatory frameworks that would require specific actions to be taken could prove both unfruitful in terms of decreasing P losses and onerous to producers who would have to adopt such practices. Each watershed, farm, and field is different, and thus risk assessments at each of these scales should be conducted to inform the suite of practices that would best optimize agricultural production, P loss mitigation and other resource concerns (e.g., N leaching or erosion).

Currently, private food processing corporations are capitalizing on the public demand for sustainably grown food and encouraging the farmers they work with to adopt conservation practices to limit their off-site impacts so that they can document this during marketing to the retail customer (Sharpley, 2018). For example, General Mills is “engaging with farmers to reduce the environmental impact of agriculture,” and they are “documenting continuous improvement over time” at the farm and field scales (General Mills, 2019).

Enhancing and Expediting Soil Drainage

In recent years, one of the primary infrastructure improvements producers have invested in is improved drainage in their fields. Further, producers have placed tile drains at shallower depths or at narrower spacings (higher drainage density; Kladivko et al., 2004). Unfortunately, this can increase P loss from fields (King et al., 2015). Although drainage is expanding in the northern Great Plains, most drainage is currently in the form of drainage ditches with surface drains in adjacent fields that rapidly drain water from the flat, clay-rich surface.

Although drainage control and water retention have been the subjects of much discussion, their efficacy in attenuating P loads remains unclear (Ross et al., 2016). Some environmental advocates have promoted regulating P discharges from tile. While drainage tile conveys water from distant fields to surface waters, it is undeniable that drainage greatly improves crop yields in humid regions with fine-textured soils, thereby providing great public benefit (King et al., 2015). Further, one cannot definitively predict that the removal of drainage tile would eliminate P loss, as the hydraulic gradient the tile provides vertically through the soil profile does hold the potential to greatly decrease erosion and the associated sediment-bound P.

Phosphorus Fertility and Soil Testing

The agronomic science community has long been a proponent of soil testing and ensuring that crops are provided ample P to minimize yield loss. In our fervor to advise producers to maximize yield potentials, we have lost sight of the trade-offs associated with “too much of a good thing.” As Withers et al. (2014) emphasized, our current system is inherently leaky and inefficient at allowing the crop to capture the P we apply. This can result in P in excess of crop needs, as well as a higher risk of P transfer from the field to receiving waters.

As a consequence of overapplication of P, and the subsequent transport of P from fields to drainage networks, we now have significant build-up of P in soil and sediment within the stream network, or P taken up by biota (i.e., aquatic microbial communities or plants within the stream network). In Iowa, as much as 1 m of sediment was deposited in former floodplains of streams during the late 19th and early 20th centuries. This sediment now disconnects the stream from the floodplain, in effect channelizing the streams, and serves as a legacy P source to the streams a century or more after deposition.

In addition to soil legacy sources of P from overapplication of manure, there has also been overapplication of fertilizer P. Phosphorus is applied to fields for crop production at levels that are based on soil fertility guidelines. However, guidelines vary across regions and are seldom updated and calibrated, and soil P levels are not regulated, leading to highly elevated P levels in some regions. Calibration of soil fertility tests for P was generally accomplished during the 1950s through the 1980s. Few regions have been able to support continued thorough investigations of P fertility. Iowa is one of the rare exceptions that has been able to continually research P fertility and update P guidance (Iowa State University, 2013). In many regions, the original P fertility datasets have been lost, or the regional extension service determined it was more expedient to use the guidance developed by a neighboring region. In recent years, crop consultants in the Western Lake Erie basin region of the United States have discovered that in the vast majority of circumstances, P guidance advises greater-than-necessary P application. These consultants are now recommending lower rates of P fertilizer, such that surveys of producers in Indiana and Ohio indicate that most fields have P applications lower than the current fertility guidance (Fig. 3; Smith et al., 2018). Surveys of soil P levels have indicated a reduction in the relative number of high soil test P fields between 2001 and 2015 throughout the region (Bruulsema, 2016). On the DelMarVa Peninsula, grain producers historically maintained that they required supplemental P applications (typically using poultry litter) even when soil tests indicated excessive P levels. They insisted that they needed P fertilizer even when the fertilizer index value was up to 1000 (i.e., 1000 mg kg^{-1} Mehlich 3 P). However, after some of the producers had discussions about the situation with practitioners from the

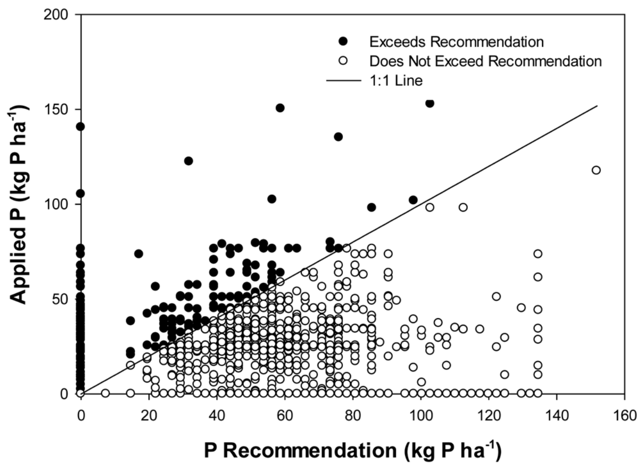

Fig. 3. Survey results from 2232 farm fields in Ohio regarding P application rate versus the recommended P application rate given current fertility guidance. Attitudes of farmers regarding P management are shifting in this region, as more than 90% of these fields apply P at or below recommended rates for their crop rotation. (Adapted from Smith et al., 2018.)

Lake Erie region, they realized that this practice may be excessive and agreed to reduce their insistence on P to soils with a fertilizer index value up to 500.

In regions with insufficient manure P to meet crop demands, the primary P sources are supplied by commercial inorganic fertilizers (Maguire et al., 2007). While it is often easier to control the precise rate of application of these types of products, the bioavailability of these products is generally designed to be greater than that of manure. It is typically estimated that 50% of P in manure is bioavailable, whereas the most common P fertilizers sold to producers (monoammonium phosphate, also known as MAP, and diammonium phosphate, also known as DAP) contain more than 90% of the P as soluble P (Smith et al., 2016). Thus, if fertilizer application is mistimed, a runoff generating rainfall event can result in bioavailable P losses to receiving waters.

Although P application rates have been excessive historically, and still are in some regions, agriculture has progressed tremendously since the P fertility guidance was developed. Farmers till less than they did 30-plus years ago, planted cultivars have changed drastically, and yield potential and goals have steadily increased. The advent of precision agriculture not only allows producers to understand the yield variability in their fields but also allows for variable rate application of inputs (seed, fertilizer, pesticides, etc.) within fields. This permits producers and resource managers to utilize available information on soils, soil test P levels, drainage patterns, potential connectivity to streams, and yields from the previous year to tailor P fertilizer prescriptions within each field to optimize productivity, environmental protection, and profitability. Continued advancement of these efforts coupled with improved calibration of soil fertility guidance and increased mechanisms for routine soil testing is needed.

Attitudes and Misunderstandings: Phosphorus Mitigation Does Not Always Reduce Eutrophication

In terms of attitudes, we face a major challenge that although considerable effort has been invested in P-based mitigation through implementation of both point and nonpoint P management controls over the last couple of decades, these efforts have not always delivered the expected, lasting ecological improvements within many freshwater bodies suffering from eutrophication (Jarvie et al., 2013b).

Adoption of nonpoint P management controls has been widely effective at reducing edge-of-field P losses (Sharpley et al., 2009), although there has been less success in demonstrating water quality improvements at the watershed scale (Sharpley et al., 2009; Reckhow et al., 2011). These disconnects between the field and watershed scales may arise from inadequate intensity and insufficient targeting of BMPs to address the critical source areas of P loss across watersheds (Sharpley et al., 2009). Conservation practice adoption has been common to all four impaired watersheds discussed here. However, it may be that mitigation efforts are not being applied at a sufficient intensity to make a significant difference. For example, in the Lake Okeechobee watershed, producers are required to manage and document P applications (Ehmke, 2014), and although P losses to receiving waters have declined in the region, lake and wetland ecology has continued to indicate persistent P enrichment (Childers et al., 2003; Canfield et al., 2018; Khare et al., 2019).

The apparent lack of improvement in P loads and concentrations at the watershed scale may also reflect lags associated with the continued chronic release of "legacy P" from past land management, in soil and in sedimentary stores between the field and the watershed outlet. Overapplication of P, either in the form of fertilizer or manure, has led to the widespread accumulation of legacy P in soils and its subsequent distribution across sedimentary stores within the wider landscape. Legacy P can buffer against rapid water quality improvements and can continue to impair water quality over timescales from years to decades and longer (Jarvie et al., 2013a,b; Sharpley et al., 2013). This can mean a long wait for downstream water quality improvements in response to mitigation measures. Although each of the four regions considered here has a unique history and legacy of P application, legacy P plays a particularly important role in areas such as the Chesapeake Bay and Lake Okeechobee watersheds, where large numbers of livestock operations are located (Maguire et al., 2007), promoting a buildup of legacy P in the soil and long-term accumulation in sedimentary stores, such as floodplains, wetlands, and water-body sediments. A recent study in the Lake Winnipeg watershed showed that a drawdown in soil P can reduce P loads in runoff (Liu et al., 2019); however, achieving a drawdown across watersheds, particularly in zones with substantial P buildup, will require more time. Indeed, the timescales of legacy P retention and recycling across different landscape pools can be highly variable, leading to large differences in water quality lag times (Jarvie et al., 2013a).

The converging and cumulative effects of BMPs can also result in unexpected and unintended responses to mitigation measures over the longer term. For example, in the Lake Erie watershed, agricultural conservation efforts since the mid-1980s have focused on reduced tillage (Great Lakes Water Quality Agreement) and transforming highly erodible land into permanent cover (i.e., the Conservation Reserve Program in the United States). Assessments of conservation efforts suggest that the current level of conservation adoption has reduced P losses from cultivated land to the Great Lakes by 39% (USDA-NRCS, 2011b) and by 41% in the Chesapeake Bay (USDA-NRCS, 2011a). Conservation tillage is widely recognized as being effective at reducing sediment and total P loads in Lake Erie (Richards et al., 2008, 2009). However, since the early 2000s, there has been an increase in soluble P losses in the Western Lake Erie basin, linked to a "re-eutrophication" of western Lake Erie and increases in the magnitude and frequency of nuisance and harmful algal blooms (Jarvie et al., 2017). While the exact reasons for this are still under investigation, there is likely a myriad of contributing factors (Smith et al., 2015b), including soil stratification, which, with continued broadcast application of P fertilizer, has resulted in accumulation of labile P at the soil surface (Baker et al., 2017), increases in hydrological connectivity via tile drainage, and reductions in suspended particulate matter available to sorb soluble P. It is increasingly clear that solutions to P-related water quality problems from the 1980s and 1990s may have unintentionally contributed to a completely unexpected and different kind of P-related water quality impairment (Jarvie et al., 2017). Indeed, mitigation efforts that are spatially precise are needed to effectively reduce P loads, and adaptive management is critical to success.

Even where reductions in P loadings have been achieved, there have often not been the expected ecological improvements, and ecosystem recovery does not always conform to smooth reversible trajectories (Jarvie et al., 2013b). Although P loads and concentrations may have been reduced, if they remain above limiting threshold P concentrations even substantial decreases in P loadings are unlikely to reduce algal growth (Bowes et al., 2012). Further P reductions, to below limiting thresholds, may be needed to have a measurable impact on reducing nuisance algal growth (Jarvie et al., 2018), and despite the vital role that P can play in nuisance and harmful algal growth, a variety of wider factors and multiple stressors (physical, chemical and biological) can influence algal biomass accrual and decouple the ecological response to reductions in P concentrations or loads. For example, algal growth can also be limited by N and by water temperature or light availability (Bowes et al., 2011). Flow velocities and flow regime also influence phytoplankton production in rivers, for example, if water retention times are lower than phytoplankton regeneration times. Nitrogen colimitation of phytoplankton growth and the "top-down" effects of invertebrate and fish grazers can exert a dominant control on algal biomass accrual. It is increasingly recognized that to achieve lasting water quality improvement and ensure more resilient ecosystem functioning, we need more holistic approaches to eutrophication management that, in addition to P control, consider other nutrients, pollution controls, land-use legacies, and restoration to address physical habitat and functional food-web interactions (Jarvie et al., 2013b). Changing attitudes may be needed, with a recognition that relying solely on P-based mitigation may not always be the most effective strategy to control freshwater eutrophication.

Avoiding Platitudes

Following the detection of microcystin in the drinking water of Toledo, OH, in 2014, Smith et al. (2015b) discussed some of the potential causes of the harmful algal blooms in Lake Erie. The "causes" they listed were the summary of several years of meetings and discussions with various stakeholder groups within the watershed. A common theme to those meetings was an interest group stating unequivocally that the problem was one specific component (e.g., manure, tile drainage, P stratification) or that one solution (e.g., cover crops or soil health) would solve the vexing issues of the day. These platitudinous pontifications, which are not unique to discussions surrounding the Lake Erie watershed, are not constructive, as they discount other factors that contribute to the problem or may preclude solutions that prove useful within the landscape. Indeed, there is no single solution or "silver bullet" for the successful mitigation of P.

Silver Bullets and Solution Singularities

When water quality issues are primarily affected by a "simple" problem like P, the easiest solution for policymakers, regulators, and conservationists to adopt would be a single solution across the entire landscape. However, P-related water quality problems are not as simplistic as they may seem, especially given the myriad of water quality pressures facing some watersheds. There is a need to address the complexity of P loss across multiple scales: within the field, between the field edge and surface water, within streams and tributaries, cumulative watershed effects, and also within-lake effects. Moreover, as noted previously, drivers may differ both within and among watersheds and regions. Environmental conditions interacting with decades of management and social adjustments have led to current conditions (Smith et al., 2015b). As has been highlighted in this paper, environmental drivers, runoff, and pathways and sources of P in the environment differ across watersheds. Thus, a single "silver bullet" is unlikely to solve P issues across these watersheds, nor will such a solution solve issues within these watersheds. In fact, the Lake Erie watershed provides an important example whereby the widespread application of a single management strategy (e.g., conservation tillage), along with continued broadcast fertilizer applications, may have unintentionally resulted in soil stratification and elevated soluble P loads transmitted via tile drains in the Maumee and Sandusky watersheds (Jarvie et al., 2017). Although this solution, which had widespread adoption, helped reduce sediment-associated P losses, it may have exacerbated soluble P losses, which in turn optimized the nutrient balance in Lake Erie for *Microcystis* blooms.

Use of cover crops is one example of a conservation practice that policymakers want to assign to the entire landscape to solve the P loss problem. However, it is highly unlikely that this practice unto itself will improve water quality. While cover crops perform exceptionally well at minimizing nitrate leaching and keeping the ground covered to minimize erosion (and associated sediment-bound P; Angle et al., 1984; Zhu et al., 1989; Dabney et al., 2001), evidence in the literature indicates that this practice may exacerbate soluble P loss under some winter conditions (Bechmann et al., 2005; Tiessen et al., 2010; Cober et al., 2018). It is also possible that cover crops could exacerbate P stratification, which is one complicating factor in solving the soluble P conundrum (Baker et al., 2017). Further, reliance on a single practice such as cover crops does not directly guide producers to use P fertilizers judiciously, nor does it address legacy P that exists either in the field (i.e., high soil test P) or the P deposited beyond the edge of field that is in various stages of transit to the receiving water (i.e., sediments deposited in the floodplain or P spiraling through the stream network). By no means does this imply that cover crops should be prohibited in P-impacted watersheds, but rather, it demonstrates that cover crops should be assigned to address specific resource concerns as part of a holistic solution across the entire landscape.

Soil Health

Soil health, the continued capacity of soil to function as a vital living ecosystem that sustains plants, animals, and humans, is a concept that inspires academics, agricultural advisors, and producers alike. There is no disagreement that a living soil in equilibrium with its environment is an optimal state for balancing and sustaining agronomic and environmental services. However, current approaches to achieving soil health are far too static and insensitive to the various initial states of poor soil health and the consequences of applying a single prescribed management strategy (no-till and cover crops) to achieve improvement. Too often, advice to producers draws simplistic correlations between soil health and improved water quality (USDA NRCS, 2011c; Soil Health Institute, 2016), but thus far, scientific studies have found no direct links between the metrics associated with soil health and water quality, and they conclude that overall relationships are complex (Smith, 2015; Roper et al., 2017). Soil health measurement metrics are still being debated, and the USDA-NRCS Soil Health Division's list of candidate metrics do not include any species of P (USDA-NRCS, 2018). If soil health is to be a part of the solution to multifaceted agronomic and environmental societal problems, then it would seem the indicators of soil health should at the very least include those that provide insight into the quantity, intensity, and environmental availability of major contributors to both types of problems.

In the Western Lake Erie basin, reduced tillage and increased drainage intensity increased nonpoint source soluble reactive P losses (Baker et al., 2014; Maccoux et al., 2016; Jarvie et al., 2017). The mechanisms driving these increased soluble reactive P losses are related to the vertical stratification and buildup of soil test P near the surface of the soil profile and enhanced vertical hydrology between the surface and tile drains, brought about by improved soil structure (Baker et al., 2017). Recently in Ontario, in the northwestern portion of the Lake Erie watershed, elevated P loss from tile drains were not observed from rotational conservation tillage and subsurface placement of P (Lam et al., 2016). Indeed, periodic inversion to disrupt pore connectivity from surface soil to tile drains and subsurface P placement have been proposed as possible solutions to the P conundrum in Lake Erie (Kalcic et al., 2016; Baker et al., 2017); however, inversion tillage is anathema to soil health proponents, and subsurface P placement is costly and time consuming compared to surface application. These examples, within the Lake Erie watershed alone, further illustrate the complexities associated with solving the P problem. Improvements are needed in our ability to clearly demonstrate which metrics of soil health (and conservation practices used to achieve these conditions) are coupled with improved water quality, and which practices may represent a trade-off toward water quality. Soil health experts should own this problem and develop a unified strategy and message for addressing the issue.

The Path Forward

Society requires food security, and food must also be affordable. However, society also requires clean, secure water sources that will support many uses. The agronomic "moon-shot" is to provide water and food security to a growing population. Over the last 350 years, society has greatly benefited from the discovery of P (Sharpley et al., 2018). Yet it has only been within the last 20 to 30 years that the feast-or-famine of P distributions within regions has resulted in an obesity equivalence in our water bodies (i.e., hypereutrophic lakes and hypoxia in coastal zones). We are the generation that must find the solutions to balance soil, agronomic, and aquatic health to ensure our P supplies last another 350 years while supporting that which society not only demands but deserves.

It is time that the agricultural and conservation communities recognize that the water quality problems facing society today are the result of a myriad of issues, some of which we have contributed to or exacerbated. It is our community that has advocated P fertilizers as cheap insurance against yield loss for decades without considering the potential environmental outcomes that would result. We are the ones that recommended that conservation tillage would solve the off-site P transport problem without realizing that leaving P fertilizers on the soil surface would result in P stratification and promote the most bioavailable (e.g., soluble) forms of P to leave the field. Society requires that we take ownership of the problem and provide solutions that will solve the vexing water quality problems. For too long, we have allowed society to blame the farmers in these watersheds, without acknowledging that for the most part, the advice they are following is ours (Smith et al., 2018).

Curbing the off-site impacts of P to our society will take a multipronged approach. As discussed in Jarvie et al. (2019), transformation of our linear P economy to a circular bioeconomy will be a vital component of reducing the P wasted in our current agricultural systems, as well as recovering P and adding value to waste streams. Further, we must undertake massive effort to revise P fertility recommendations (Smith et al., 2018). Where possible, this should include bringing original calibration datasets to the table but must also incorporate new fertility studies that include modern crop cultivars and current management practices (e.g., fertilizer sources, timing, placement). Future fertility guidance should be able to incorporate off-site risk assessments and yield potentials at a subfield scale. This may require data mining coupled with crop and water quality modeling to provide background and context to such recommendations. Additionally, it is highly advisable that holistic nutrient management is considered in long-term research efforts, such as for the "aspirational" treatments imbedded within projects such as the Long-Term Agroecosystem Research (LTAR) network (Kleinman et al., 2018).

The challenges we continue to face in managing eutrophication across North America, from Lake Winnipeg to Lake Okeechobee, demonstrate that it is imperative that attitudes change and that we find and adopt solutions to address these large-scale water quality issues more holistically. While we are prescribing these solutions, we must consider the complex interactions of biotic (e.g., crop response, soil health, cover crop, aquatic biological responses) and abiotic (e.g., altered hydrology, more intense hydrologic cycle, sources of nutrients) factors that ultimately affect P fate, transport, and aquatic ecological impacts across latitudes. Just as the recommendation of conservation tillage contributed to the increase in soluble P losses to Lake Erie, we must consider the "law of unintended consequences," as we must be able to foresee how our management recommendations today will potentially perturb the system in 10 to 30 years.

We must learn lessons from our neighboring watersheds so that we do not repeat the same mistakes. Continued dialogue and increased opportunities for the transfer of knowledge within and among watersheds is necessary, both within North America and globally. To solve the P conundrum, we must ensure that we are not blinded by our own platitudes. Collectively, we must find solutions to address P source and transport throughout the landscape, recognizing that solutions may vary both within and across watersheds.

Conflict of Interest

The authors declare no conflict of interest.

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