Published August 15, 2019

Journal of Environmental Quality SPECIAL SECTION

CELEBRATING THE 350TH ANNIVERSARY OF DISCOVERING PHOSPHORUS—FOR BETTER OR WORSE

Increasing the Effectiveness and Adoption of Agricultural Phosphorus Management Strategies to Minimize Water Quality Impairment

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Abstract

Phosphorus (P) is essential for optimum agricultural production, but it also causes water quality degradation when lost through erosion (sediment-attached P), runoff (soluble reactive P; SRP), or leaching (sediment-attached P or SRP). Implementation of conservation practices (CP) affects P at the source (avoiding), during transport (controlling), or at the water resource edge (trapping). Trade-offs often occur with CP implementation. For instance, multiple researchers have shown that conservation tillage reduces total P by over 50%, while increasing SRP by upward of 40%. Conservation tillage may increase water quality degradation as SRP is more bioavailable than is particulate P. Conservation practices must be implemented as a system of practices to increase redundancy and to address all loss pathways, such as P management with conservation tillage and a riparian buffer. Further, planning and adoption must be at a watershed scale to ensure practices are placed in critical source areas, thereby providing the most treatment for the least price. Farmers must be involved in watershed planning, which should include financial backstopping and educational outreach. It is imperative that CPs be used more effectively to reduce and retard off-site P losses. New and innovative CPs are needed to improve control of P leaching, address legacy stores of soil test P, and mitigate increased P losses expected with climate change. Without immediate changes to CP implementation, P losses will increase due to climate change, with a concomitant degradation of water quality. These changes must be made at a watershed scale and in an intentional and transparent manner.

Core Ideas

- Phosphorus-reducing conservation practices must control all P pathways.
- Phosphorus-reducing conservation practices must be utilized as systems.
- New and innovative conservation practices are needed to improve control of P.
- Farmer decision-making must be considered when implementing conservation practices.
- Watershed planning and conservation practice implementation must be intentional.

Phosphorus (P) is essential for life as it forms the backbone of RNA, DNA, and energy transfers. Many perennial plants have symbiotic relationships with mycorrhiza and other soil microorganisms to secure limited natural P (Margalef et al., 2017; Spohn et al., 2018). Phosphorus is also essential for agricultural production, as insufficient amounts often limit optimal crop production (Syers et al., 2008; van de Wiel et al., 2016). Applications of P have optimized crop production, with first applications coming from organic sources such as animal or human waste, bones, and guano, and later via mineral fertilizers mined from rock phosphate deposits. While crop production benefits are many, P is akin to a double-edged sword, where additions have increased yields but P loss to water resources has hastened water quality degradation via increased algal production (Stow et al., 2015; USEPA, 2019). Water resources across the globe—including Lake Erie, Chesapeake Bay, inland and coastal waters of Florida, Baltic Sea, and Lake Taihu in China—are impaired due to increased algae blooms that negatively affect drinking water quality, fishing, recreation, and other uses (Conley et al., 2009; Dale et al., 2010; Federal Leadership Committee of the Chesapeake Bay, 2009; Reddy et al., 2011; Scavia et al., 2014; Qin et al., 2010). Many of these water resources derive much of their nutrients from agricultural activities, which have been implicated in nutrient losses affecting water quality for decades (Brink, 1975; Sharpley et al., 1994; Webb, 1962).

In an attempt to protect soil resources and reduce nutrient losses, conservation practices (CPs) have been prescribed in the United States since the USDA Soil Conservation Service, now the Natural Resources Conservation Service (USDA-NRCS) began in 1935 (USDA-NRCS, 2019a). In the 1970s, the USDA-NRCS focused on reducing nonpoint source pollutants, including P. Conservation practices targeting P have evolved over time but until recently focused primarily on reducing sediment losses (erosion) and P attached to sediment (Sharpley, 1996; Sharpley et al., 2003). Researchers have more recently elucidated other

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Abbreviations: CEAP, Conservation Effects Assessment Project; CP, conservation practice; DPS, degree of phosphorus saturation; NIFA, National Institute of Food and Agriculture; PSM, phosphorus sorbing material; SRP, soluble reactive phosphorus; STP, soil test phosphorus.

J. Environ. Qual. 48:1204–1217 (2019)
doi:10.2134/jeq2019.03.0114
Received 12 Mar. 2019.
Accepted 25 Apr. 2019.
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significant soluble reactive P (SRP) loss pathways associated with surface runoff and leaching.

Because it is critical that we maintain optimum crop production to feed the current population of over 7.5 billion (United Nations, 2019), we need to find ways to use P to increase crop yields while minimizing losses to water resources; it is also imperative to reduce and retard off-site P losses by using current CPs more effectively and to develop new, more effective CPs. The purpose of this paper is to reflect on recent advances in understanding P loss pathways into water and how researchers can use that information to better balance profits and clean water.

Phosphorus Loss Pathways

Agricultural P losses occur where there is an interaction of hydrology with field-scale characteristics and processes, soil P, and applied P as fertilizer, animal waste, and/or other P sources. Sediment-attached P is lost during erosion events, which either redeposit P-rich soil particles within the field or moves them off the field, where they have the potential to affect water resources. The amount of P lost is a function of the amount of soil eroded, the soil P content, and soil texture. Researchers have known for years that smaller particles (particularly clay-sized particles) that are more susceptible to erosion losses contain more P than do sand particles (Cox, 1994; Sharpley et al., 1985).

Runoff P losses occur when water flows over the soil surface, resulting in transport of SRP from fields (Dunne, 1983; Dunne and Black, 1970; Horton, 1933). Runoff P losses are a function of many factors, including soil texture, antecedent soil moisture, soil P content, soil pH, applied P, and vegetation (Cox and Hendricks, 2000; Heathwaite and Dils, 2000; McDowell et al., 2001). While it is beyond the scope of this paper to detail how each of these factors affects runoff P losses, we provide the following examples. Sharpley (1997) found that the magnitude of runoff SRP losses from soils amended with poultry litter was related to the availability of soil P sorption sites (i.e., soil P source), mainly Fe- and Al-oxide minerals in acid soils. Yet, Buda et al. (2009) demonstrated the critical importance of hydrology; they reported higher SRP loads in runoff due to the greater runoff volumes generated in low soil test P (STP) soils that were located at the downslope landscape position nearest the stream. These losses were greater compared with higher STP soils located in upslope positions where little to no runoff was generated (Buda et al., 2009).

Phosphorus leaching, although identified as early as the 1940s (Sims et al., 1998), is the loss pathway that remains the least acknowledged, understood, and researched. Phosphorus leaching has been demonstrated in sandy texture soils, organic soils, and structured soils that develop preferential flow pathways (Heathwaite and Dils, 2000; King et al., 2015a; McDowell et al., 2001; Sims et al., 1998). For example, Kleinman et al. (2015) and Toor and Sims (2015) reported higher P leaching losses through intact soil cores collected from the Atlantic Coastal Plain when preferential flow dominated compared with soils controlled by matrix flow due to greater subsoil contact and capacity for P sorption.

Artificial drainage systems (e.g., tile drains and ditches) were identified as a significant source of P leaching not only in the United States (Sims et al., 1998) but also in European settings (Gelbrecht et al., 2005). King et al. (2015a) determined that ~50% of the SRP and 40% of the total P in the Lake Erie watershed leaches through the soil to drain tiles. Often, leaching losses are more prevalent as STP concentrations increase, especially as P moves downward into the subsoil (Kleinman et al., 2015).

The amount of P leached is also significantly affected by nutrient management practices (Toor and Sims, 2015; Kleinman et al., 2015)—source, placement, timing, and rates. Ultimately, the amount of P leached is a function of soil texture, potential preferential flow paths, drainage intensity of tiles and ditches (if installed) including depth and spacing, ditch design and maintenance (if used), drainage water management, hydrology, climate, cropping systems including tillage type, STP, and nutrient management (King et al., 2015b). Reducing leaching losses is most difficult, partly because of the prevalence of artificial drainage but also because it is challenging to predict how or when leaching losses occur.

Implementing Conservation Practices That Reduce Agricultural P Losses

A variety of CPs are available to reduce P from agricultural fields (Dodd and Sharpley, 2016; Osmond et al., 2012c,d; Sharpley et al., 2000). Multiple control strategies are necessary at the source, during transport, and at or in the receiving water resources to control agricultural nutrient loading to surface waters (Osmond et al., 2012c,d). Scientists and practitioners have discussed the necessity of practice redundancy or systems of CPs for decades. The USDA-NRCS refers to this concept as "avoid, control, and trap" (USDA-NRCS, 2019b). Ideally, the practice(s) implemented should address one or more of the P loss pathways by reducing erosion and sediment-attached P, runoff P, or leached P depending on site concerns.

Practices That Avoid Agricultural P Losses

Because STP and P applications are highly determinant of P losses, *avoiding* is important—keeping STP as low as possible and only applying P as needed and either mixing or injecting (Bergström et al., 2015; Kleinman et al., 2015; Withers et al., 2014). An important first-line strategy to avoid P losses is to maintain agronomically adequate but environmentally low STP levels. Unfortunately, in many areas, STP is high and/or is increasing. For example, Lu and Tian (2017) demonstrated that P application worldwide has increased threefold since 1960, with most of the increase in Asia (specifically China) and Brazil. Soil test P is not necessarily the driving factor in P loss since hydrology and field conditions play a critical role in P transport; yet all things being equal, the greater the STP, the larger the losses (Sharpley et al., 1985).

Kleinman et al. (2002) demonstrated a 3- to 10-fold increase in P runoff concentrations between low STP (12–26 mg kg^{-1} as Mehlich-3) and high STP (396–415 mg kg^{-1} as Mehlich-3) soils of the same soil series. Furthermore, researchers have identified an interaction between STP, P source, and P losses (by all three pathways—erosion, runoff, and leaching); thus, the importance of maintaining agronomically sufficient but low STP cannot be overstated. Unfortunately, many US states have high STP in particular areas due to natural conditions (e.g., Kentucky and Florida), concentrated animal operations (e.g., Delaware, Pennsylvania, and North Carolina), or cropping systems (e.g., Florida and Virginia). The International Plant Nutrition Institute compiled percentage soils testing below critical levels by state, which demonstrates that many states, on average, have high or very high STP (Fig. 1).

Fiorellino et al. (2017) and Kamprath (1999) demonstrated that at very high STP (>100 mg kg^{-1}; as Mehlich-3 P), crop yields were not affected for over 15 yr when P applications ceased. Extrapolating STP drawdown, these researchers suggested it would take more than 30 yr to decrease STP below the agronomic critical level depending on the soil texture and initial soil test level. The STP of Lynchburg soil (fine-loamy, siliceous, semiactive, thermic Aeric Paleaquult) was three times greater than the agronomic critical level (50 mg kg^{-1}), the level at which a yield response would be expected (Kamprath, 1999). Elevated STP was more than seven times greater than the agronomic critical level (also 50 mg kg^{-1} for Maryland) in Mattapex silt loam (fine-silty, mixed active, mesic Aquic Hapludult) (Fiorellino et al., 2017).

McCollum (1991) demonstrated on a mineral–organic coastal plain soil (Typic Umbraquult) that drawdown of P is not linear due to reversion of P into nonextractable forms, which suggests that drawdown will occur more quickly in early years and then slow down. Sharpley et al. (2013) calculated drawdown times ranging from 4 to 27 yr on a range of soils from the United States, Canada, and Europe. Ultimately, many high-STP soils can be farmed with no or limited P additions without affecting yields for decades, but during P drawdown, these soils will continue to lose P to water resources (Qin and Shober, 2018; Johnston and Poulton, 2019).

Conservation strategies that avoid will be viewed most favorably if producers think of them as ways to acquire needed P at lower cost, rather than as a conservation technology that just costs money (Garnache et al., 2016; Hoag et al., 2012b). Nutrient management (i.e., planning the source, rate, timing, and method P application), for example, is regarded as the main practice to avoid agricultural P losses and will be seen by most farmers as a valuable tool to help them apply P most profitably. Nutrient management is not discussed in detail in this paper as this topic is covered separately in this special section (Bruulsema et al., 2019; Grant and Flaten, 2019); we instead focus here on how applied P becomes vulnerable to all loss pathways depending on the source, rate, placement, and timing.

Practices That Control Agricultural P Losses

Practices that *control* sediment-attached P include conservation tillage, cover crops, terraces and grassed waterways, water-control structures, and soil amendments; these practices may address one or multiple P loss pathways by reducing rainfall impact, increasing infiltration, controlling erosion, or reducing leaching. A recent meta-data analysis of particulate P losses due to conservation tillage demonstrated on average 45% (concentration) and 55% (load) reductions relative to conventional tillage (Daryanto et al., 2017). These average reductions, however, do not reflect differences in climate, topography, cropping system, or length of tillage; disaggregating the data indicated that the effectiveness of conservation tillage was lower on more sloping

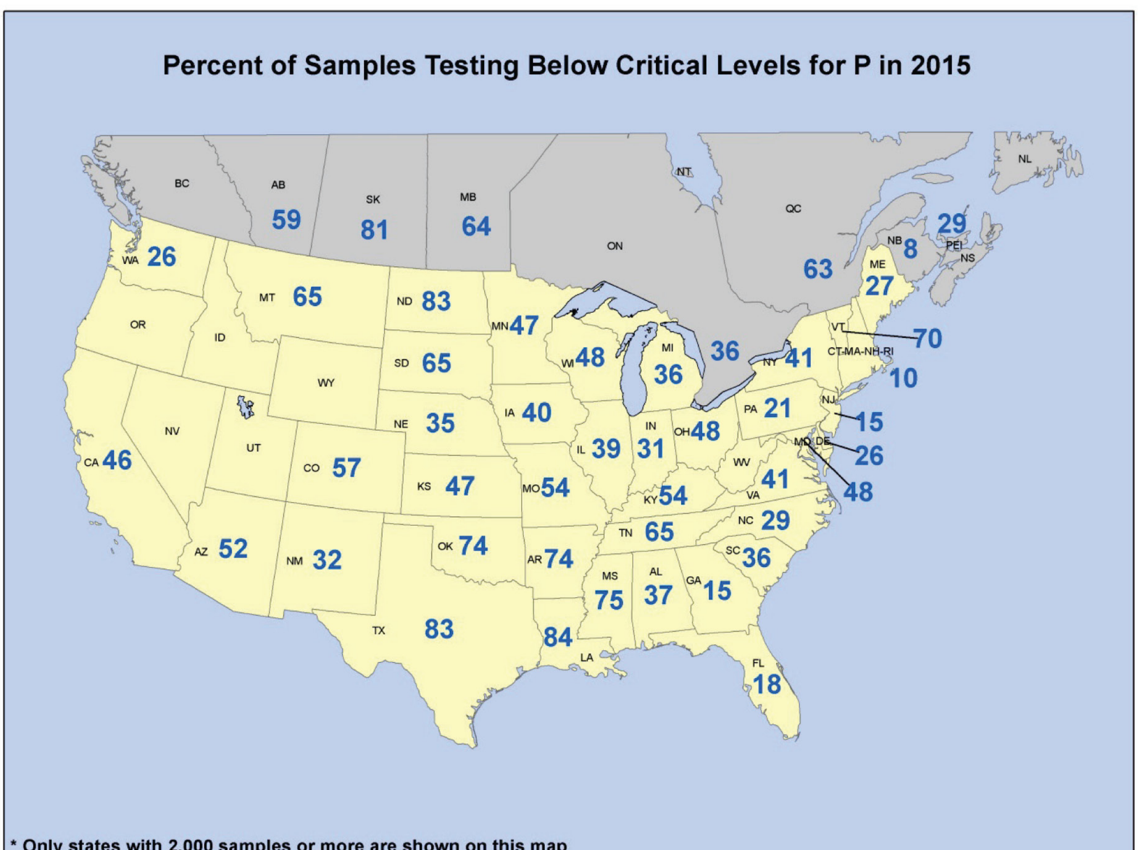

Fig. 1. Percentage soils testing below critical levels by state (2015). Provided with permission of the International Plant Nutrition Institute.

fields (>4%) or with wetter antecedent conditions. Dodd and Sharpley (2016) noted a range for particulate P reduction due to conservation tillage from −33 to 96% compared with 45 to 89% for grassed waterways. Following 9 yr of research on a clay soil in Finland, Uusitalo et al. (2018) demonstrated that particulate P loads and concentrations were 27 and 55% lower, respectively, with no-till than with conventional till.

Research has indicated that P sorbing materials (PSMs) offer some potential in reducing SRP in runoff when land applied as a soil amendment. For example, King et al. (2016) determined that field-applied gypsum, if applied at high enough rates, could reduce surface losses of P by approximately 40%. Similarly, several researchers documented reductions in runoff SRP losses when other PSMs (fly ash, water treatment residuals, steel slag, etc.) were land applied to high P soils (Bryant et al., 2012; Dayton and Basta, 2005; Stout et al., 1998).

Water-control structures can reduce SRP loss from fields by affecting hydrology (Evans et al., 1991; Penn et al., 2017; Zhang et al., 2017). Evans et al. (1991) found a net reduction in edge-of-field total P losses of 35% under controlled drainage. Zhang et al. (2017) reported that controlled drainage decreased surface runoff SRP concentrations by 19% and by 23% when cover crops were used with controlled drainage.

In a recent review article on cover crop effectiveness, Blanco-Canqui (2018) found 13 articles that measured sediment losses and reductions, which ranged between 0 and 100% compared with no cover crop; one site demonstrated no difference due to cover crops. Likewise, planting of cover crops at most locations did not affect total P losses when compared to fields that used no cover crops. Cover crops planted between olive trees (*Olea europaea* L.) in Spain were found to reduce both sediment and dissolved P in 1 of 2 yr (Gómez et al., 2009).

Practices That Trap Agricultural P Losses

Practices that *trap* P assimilate edge-of-field P losses and prevent them from entering sensitive water bodies. Examples of trapping practices include vegetative buffers, wetlands, and stormwater P filters. Various researchers documented a 20 to 90% reduction of total P by riparian buffers through sediment deposition (Cooper and Gilliam, 1987; Dillaha et al., 1989; Dillaha and Inamdar, 1997; Daniels and Gilliam, 1996; Magette et al., 1989; Peterjohn and Correll, 1984; Van Vooren et al., 2017); three recent reviews confirmed these reduction percentages (Dodd and Sharpley, 2016; Hoffmann et al., 2009; Roberts et al., 2012). These studies demonstrate that wider buffers increase removal effectiveness, but the amount of reduction diminishes as buffer width increases. Additionally, slope, soil type, hydrology, and many other factors determine the overall effectiveness for riparian buffers. Finally, resuspension of sediment and transport out of the riparian buffer can occur depending on the buffer type and the storm event, thus reducing overall effectiveness of the buffer over time (Dodd and Sharpley, 2016; Gelbrecht et al., 2005).

Wetlands can also be effective at removing sediment P, where efficacy is related to P source, loading of the wetlands with sediment P, stormwater residence time, and a host of other factors. For example, Bergström et al. (2015) demonstrated an approximately 35% reduction in P by wetlands in Sweden, mostly as particulate P. In a review of the literature, Dodd and Sharpley (2016) reported ranges for particulate P reduction of 47 to 74%.

Phosphorus sorbing materials offer the potential to trap edge-of-field P losses when used in off-site stormwater treatment structures. Off-site structures that contained PSMs were found by multiple researchers (Penn et al., 2017; Qin and Shober, 2018) to be effective at removing soluble P (and to some extent particulate P) from runoff. Additionally, several researchers have demonstrated that PSMs in drain tiles may be useful in reducing leached P (King et al., 2016; McDowell et al., 2008).

Many factors affect P reduction by off-site structures, including structure material, resident time, and influent concentration. In their review, Penn et al. (2017) determined that Fe-containing PSMs performed better (~35%) at reducing P concentrations in agricultural runoff than did nonslag and slag materials containing Al (~25%). However, to be cost-efficient, these structures must carefully balance the P adsorptive capacity of the material, as well as the flow dynamics of the system (i.e., containment structure or buffered filter). In many cases, the major portion of P loss occurs during high storm flows, when there is a larger potential for the containment structure to be breached, allowing P-rich water to bypass and not chemically interact sufficiently with the by-product (Penn, 2014). In a review of the literature, Qin and Shober (2018) referenced multiple articles showing different degrees of PSM effectiveness; ultimately, there is concern that these materials may only temporarily reduce SRP runoff when land applied or used in off-site structures.

Considering Trade-Offs

Conservation practices may have contradictory effects on nutrient loss. Sometimes different nutrients are affected, as is the case with terraces, which are effective in reducing sediment-attached P but may increase nitrogen (N)-leaching losses (Gale et al., 1993; Meals et al., 2012c; Osmond et al., 2012d). Sometimes practices affect different forms of the same nutrient; conservation tillage, for example, decreases sediment-attached P while often increasing SRP in runoff.

Geochemical cycling of P from field to water resources associated with different CPs occurs, as demonstrated with conservation tillage, cover crops, and buffers, which can have unintended consequences of increasing P losses (Dodd and Sharpley, 2016; Osmond et al., 2012d). These sinks can become P sources over time. The net benefit of CP systems will depend on the relative reductions in different P amounts and forms from field to water resource.

Typically, practices that are effective in reducing particulate P, such as conservation tillage, wetlands, buffers, and cover crops, do not appear to be effective in retarding SRP losses in runoff or leachate. A meta-analysis comparing P losses of conservation versus conventional tillage demonstrated a 40% increase in both SRP concentration and load due to conservation tillage, although the amount varied based on rainfall (Daryanto et al., 2017). Other studies have demonstrated SRP increases from land under conservation tillage (Dodd and Sharpley, 2016; Jarvie et al., 2017; Renwick et al., 2018; Uusitalo et al., 2018). The increase in SRP with conservation tillage was relative to the reduction in sediment-attached P, which may actually increase degradation of water quality since SRP is more bioavailable. Researchers in the northern Great Plains of Canada determined that sediment-attached P was reduced when conservation tillage was implemented but that total P load increased by 12% mainly because the proportion of total P load as SRP increased from 67 to 85%, most of which occurred during snowmelt (Tiessen et al., 2010).

As soil P increases in buffer strips, runoff water often contains more dissolved P when it exits the buffer than when it enters (Cooper and Gilliam, 1987; Dodd and Sharpley, 2016; Parsons et al., 1994). Additionally, buffer vegetation may contribute to SRP losses during senescence, as suggested by numerous scientists (Dodd and Sharpley, 2016; Elliott, 2013; Hoffmann et al., 2009). Biogeochemical cycling of P in buffer zones, along with different hydrologic pathways, is highly complex and depends on many factors, including buffer type, soils, and sediment-attached P received. In reviews, Hoffmann et al. (2009) and Roberts et al. (2012) both discussed remobilization of P into more soluble forms through interaction with the soil P pools, microbiological cycling, and/or vegetation, thus increasing SRP losses when saturated or flooded. Jenkins and Sims (2012) recommended measuring the degree of P saturation (DPS) based on the Mehlich-3 concentrations of P, Fe, and Al in buffer strip soils, as an indicator of the potentials for buffers to act as a source (DPS > 0.15) or a sink (DPS < 0.15) for SRP losses through the buffer. The researchers also recommended deep tillage (>45 cm) of P-saturated buffer soils to regenerate P sorption capacity and improve SRP removal by existing buffers.

Wetlands, while somewhat effective at reducing particulate P, vary in their ability to reduce soluble P. Richardson (1985) demonstrated that the capacity for wetland P sorption was a function of soil Al content, with more soluble P sorbed as soil Al content increased. Yet Kovacic et al. (2000) reported that constructed wetlands collecting tile drainage in the US Midwest were virtually ineffective in reducing P, primarily because the wetlands received mainly SRP that was initially sequestered by wetland plants but then released when vegetation died.

Similarly, cover crop research has shown mixed results relative to P losses on water quality depending on cover crop species, stage of growth, soil chemical properties, climate, soil texture, cropping systems, and other factors (Sharpley and Smith, 1991). Some researchers have demonstrated the potential for SRP losses from cover crops is a function of cover crop species, climate, and soil P concentrations, among other factors, which affect the magnitude of the losses (Cober et al., 2018; Lozier et al., 2017). Ulén (1997), however, found no increase in P losses from cover crops or stubble in Sweden. In colder climates, freeze–thaw cycles may increase SRP losses by releasing P from vegetation during hydrologic events, such as runoff from snowmelt (Lozier and Macrae, 2017; Lozier et al., 2017); SRP losses from freeze-intolerant plants are more prevalent (Cober et al., 2019). Nevertheless, Cober et al. (2019) found the contribution of SRP from soil was always greater than that from cover crops, regardless of cover crop type. When cover crops are used with manure applications, SRP losses from the manure can overwhelm any cover crop contribution (Kleinman et al., 2005).

Practices that are effective at reducing particulate P losses often result in higher P leaching losses. For example, an assessment of 50 yr of published drainage P losses indicated a threefold increase in P leaching from no-till fields relative to conventional tillage due to the formation of preferential flow channels (Christianson et al., 2016). Because conservation tillage has been shown to increase P leaching in well-structured soils (Kleinman et al., 2009, 2015), light tillage may temporarily lessen P leaching losses by disrupting preferential flow through macropores, but mixing can also reduce P stratification arising through no-till and thereby reduce overland SRP losses (Sharpley, 2003).

Bergström et al. (2015) found no reduction in leachate SRP from drained fields with cover crops but did see a relationship between STP and leachate P losses. However, a study by Williams et al. (2018) did not indicate differential preferential pathway leaching losses based on tillage, but fertilizer placement (injected or mixed rather than broadcast) did reduce losses. Studies from Canada did not identify leaching (tile drains) or runoff differences based on tillage (disk till vs. ridge till) on a sandy loam, but loss was associated with seasonality; the greatest surface losses were during snowmelt (Lam et al., 2016).

Just as trade-offs between soluble and particulate P losses exist, researchers have also demonstrated trade-offs between surface and subsurface loss, as well as seasonal variations in these reductions based on climate, soil type, and management (Evans et al., 1991; Zhang et al., 2017). These trade-offs must be addressed in conservation strategies focused on P reductions from agricultural fields to water resources. For example, practices that enhance infiltration (e.g., controlled drainage and conservation tillage) can result in higher P losses via subsurface flow (Evans et al., 1991).

Implementing Conservation Practice Systems

The selection of appropriate CPs to reduce impacts on water quality depends on many factors, including the source and magnitude of the potential P losses. Often, addressing P losses at the field scale will require implementation of multiple CPs that address different P loss pathways. Nummer et al. (2018) used a meta-analysis approach to evaluate the effects of CPs (namely, conservation tillage, buffers, grassed waterways, and terraces), implemented alone or in combination, on P losses using data from the Measured Annual Nutrient loads from AGricultural Environments (MANAGE) database. Initial analysis revealed that more fertilizer was applied to fields in the database with CPs (e.g., cropland vs. pasture), thus confounding results due to the opposite effects of fertilizer application and CPs. Once this difference was accommodated, the researchers determined that particulate P was reduced by 58% and total P by 76% when CPs were used either individually or in combination. Daryanto et al. (2017) demonstrated the importance of cropping system to P loss, as did Nummer et al. (2018).

Suites of CPs do not always lead to water quality improvement, as demonstrated by Baker et al. (2018). Failure of CP suites to address P losses may be due to the lack of redundancy in the system, inability of practices to reduce P, transformation of P, or a multitude of issues surrounding CP implementation and water quality monitoring. In a review of conservation measures, Ward et al. (2018) detailed and discussed the benefits and limitations of many USDA-NRCS CP standards. Their conclusion was that understanding these practices is not enough; implementation must be part of a more thorough watershed program.

Scale of Implementation and Adoption

While many CPs are implemented at the farm field or edge-of-field scale, successful conservation strategies must be implemented at a watershed scale (Gale et al., 1993; Osmond et al., 2012c,d). Watershed-scale conservation is more difficult than farm-scale implementation, mainly because it requires planning and implementation at a significantly larger scale, involves more time and resources, and is necessarily more complex due to the involvement of diverse groups of stakeholders with varied vested interests.

Successful watershed-scale planning also requires enough water quality data to understand and define the water quality problem, which includes recognizing the pollutant(s) of concern and the hydrologic pathways of the pollutant(s). It also requires matching the CPs to the pollutant(s) of concern as a system of practices. Failure to properly identify the water quality problem can result in unsuccessful efforts to improve water quality, as well as wasted time and resources. Osmond et al. (2012d) illustrated this concept by analyzing 13 watershed-scale conservation studies. The researchers noted that often the CPs did not match the pollutant of concern. For example, when conservation tillage was used in a flat, drained landscape to control N (pollutant of concern), this practice likely increased subsurface loading losses of both N and P to the drinking water reservoir. Neither the performance of the practice nor the hydrologic pathways were considered when selecting conservation tillage. Overall, systems of practices were rarely considered in any of the 13 studied watersheds unless the practices were automatically tied together (i.e., terraces and grassed waterways, manure and barn management systems and a nutrient management plan).

Ideally, the critical source area(s) of the watershed (i.e., the areas that generate the predominant load; Osmond et al., 2012c,d) should be determined and practices placed in these areas to maximize effectiveness and minimize cost. Since CP implementation is often voluntary, farmer buy in and willingness to implement the recommended practices in the critical source areas are necessary. Working with farmers is extremely complex, and adoption by producers is a multifaceted decision process. At a minimum, it takes financial resources to help fund practices, local agency personnel to work with farmers over the long-term, and time (Hoag et al., 2012b; Jennings et al., 2012). Often the CP(s) utilized did not match the needed practice(s), nor were they implemented in the critical source area(s).

Time of Recovery

Assuming that the watershed-scale conservation plan is well developed and executed, and the water quality monitoring design is sufficient to detect change, the time between implementation of CPs and measured water quality improvement is often significant (Meals et al., 2010). The concept of lag time assumes that given enough time, CPs will reduce pollutant loading. The length of this lag time depends on the pollutant, size of the watershed, storage within the watershed, climate, establishment of the specific CPs, and many other factors. It can take decades to draw down P stored in fields or within buffers or wetlands (as discussed above). Phosphorus accumulated in water resources (lakes, streams, etc.) will take decades if not centuries to be removed (Meals et al., 2010; Sharpley et al., 2013).

Even if the appropriate systems of CPs were identified and implemented within the critical source areas, it cannot be assumed that these practices will be maintained. Conservation practice implementation is not static over time (Jackson-Smith et al., 2010; Jackson-Smith and McEvoy, 2011). Often producers switch practices due to land ownership changes, accommodation of new CPs or programs, changes in cost-share programs, or agronomic reasons (e.g., compaction, weeds, costs). Additionally, historic long-term applications of P, particularly from animal manure, transforms P dynamics by changing soil sorption–desorption equilibrium. For example, high STP from manured soils shows increased P desorption relative to commercially fertilized fields at the same STP (Jiao et al., 2007). Many watersheds in the United States and Europe have had long-term animal waste application resulting in large P stores. Even without additional P inputs, legacy P stores will cause P losses to water resources for decades if not centuries.

Discerning whether lack of water quality progress is due to lag time or other factors is extremely difficult. In a review of eight Swedish watersheds of different sizes, Bergström et al. (2015) suggested that there were no clear water quality trends for P; P loads were lower in some watersheds and higher in others. Many authors acknowledge the difficulty of measuring and detecting P changes due to CP adoption (Bergström et al., 2015; Meals et al., 2010; Osmond et al., 2012c,d).

Conservation by Experiential Learning: Watershed Assessments

Numerous federally sponsored watershed-scale projects have tried to document the effects of CP implementation on water quality change. Early efforts included the Black Creek Project in northeastern Indiana and the Model Implementation Program (Dressing et al., 1983). The Rural Clean Water Program was jointly funded by USDA and the USEPA as a follow-on project to the Model Implementation Program. The intent of the Rural Clean Water Program was similar but not identical to the Model Implementation Program; the expectation was to relate CP implementation at a watershed scale to water quality change (Gale et al., 1993). Many of these early projects focused on experimental design, CP implementation, project management, and documentation of water quality change; due to the diversity of projects, researchers were only able to synthesize some of the lessons learned.

The next iteration of watershed-scale projects was the USEPA Section 319 National Nonpoint Source Monitoring program (Spooner et al., 2011), which utilized guidance developed from the earlier projects and instituted the six-step watershed planning guidance (USEPA, 2013). Some of the Section 319 watershed project managers were able to document effectiveness of grazing management, nutrient management, and stream restoration at the watershed scale.

More recently, the National of Food and Agriculture Competitive (NIFA) Grant Watershed Studies as part of the USDA Conservation Effects Assessment Project (CEAP) tried to document the relationship between CP implementation and water quality change in 13 watersheds across the United States (Osmond et al., 2012c,d). Conservation practices to reduce P (and sediment) were implemented in 9 of the 13 watersheds (Arkansas, Georgia, Indiana, Kansas, Missouri, New York, Ohio, Pennsylvania, and Utah) (Fig. 2). Demonstrated P reductions occurred in two of the watersheds: New York and Ohio (Meals et al., 2012c; Osmond et al., 2012b).

The New York project included paired watersheds—a dairy farm and a forested area (Gale et al., 1993; Osmond et al., 2012b). Many P-reducing CPs were implemented in the treatment watershed (dairy farm), including waste storage, a nutrient management plan, and water management around the barnyard, yet water quality did not improve. Only after additional pollutant sources were determined and more CPs were implemented, including streambank crossings and restoration and precision feeding, did water quality improve. The New York watershed example demonstrates the difficulty in assessing and then implementing necessary practices at the watershed scale, even when the watershed is very small and, in this case, was defined as the farm.

Long-term monitoring (>20 yr) in Rock Creek, Ohio, which feeds Lake Erie, indicated a reduction in sediment-attached P, with much of the success attributed to conservation tillage adoption and reduced P fertilization (Meals et al., 2012c; Richards et al., 2008). Subsequently, SRP increased dramatically due to tillage changes, increased drain tile installation, changes in rainfall intensity and duration, and surface and fall application of P fertilizers with conservation tillage (Jarvie et al., 2017; Meals et al., 2012c; Smith et al., 2018). The Rock Creek example demonstrates, as eloquently stated by Jarvie et al. (2017, p. 123), the "unintended, cumulative, and converging impacts" of changes in CPs and rainfall that increased SRP loads into Lake Erie, triggering harmful algal blooms.

Seven NIFA-CEAP projects could not demonstrate P reduction in the water resources of concern being monitored (Arkansas, Georgia, Indiana, Kansas, Missouri, Pennsylvania, and Utah; Osmond et al., 2012c). For example, the Arkansas watershed had significant and rapid land use change during the 12-yr sampling period (1992–2004), as pastures were transformed into suburbs (Hoag et al., 2012a). Minimal nutrient loads in the Georgia watershed were associated with low-intensity agriculture; agricultural fields comprised approximately 40% of the watershed and were interspersed with natural forested areas and adjacent riparian buffers (Meals et al., 2012d). Thus, it was difficult to measure water quality change as nutrient concentrations were inherently very low. Part of the Indiana project was to determine pollutant sources even as CPs were being implemented. Most of the P was determined to be delivered from urban areas and wastewater treatment plants rather than agriculture (Osmond et al., 2012f). Project personnel in Kansas believed that lack of sediment and P reduction in Cheney Lake was due to a low-intensity water quality monitoring design, Conservation Reserve Program implementation before monitoring began, and conservation tillage primarily being adopted toward the end of the project (Osmond et al., 2012e). Research in the Missouri project was at multiple scales (plot, field, and watershed), and monitoring occurred from 1992 to 2004 (Arabi et al., 2012a). Despite implementation of multiple CPs, no water quality improvement was demonstrated, in part because project personnel demonstrated that most of the CPs were not located in the critical source area. Although the Pennsylvania paired-watershed project demonstrated sediment reductions resulting from narrow, vegetated riparian buffers, stream crossings, fencing, and stream bank stabilization on grazed lands, there was no concomitant reduction in P (Osmond et al., 2012a). The water quality monitoring data in the Utah project were insufficient to provide a historical record. Further, many of the CPs implemented in the original Utah project were no longer in use (Meals et al., 2012a).

As noted above, inability to demonstrate P reductions from CP implementation may be due to multiple reasons, including the following:

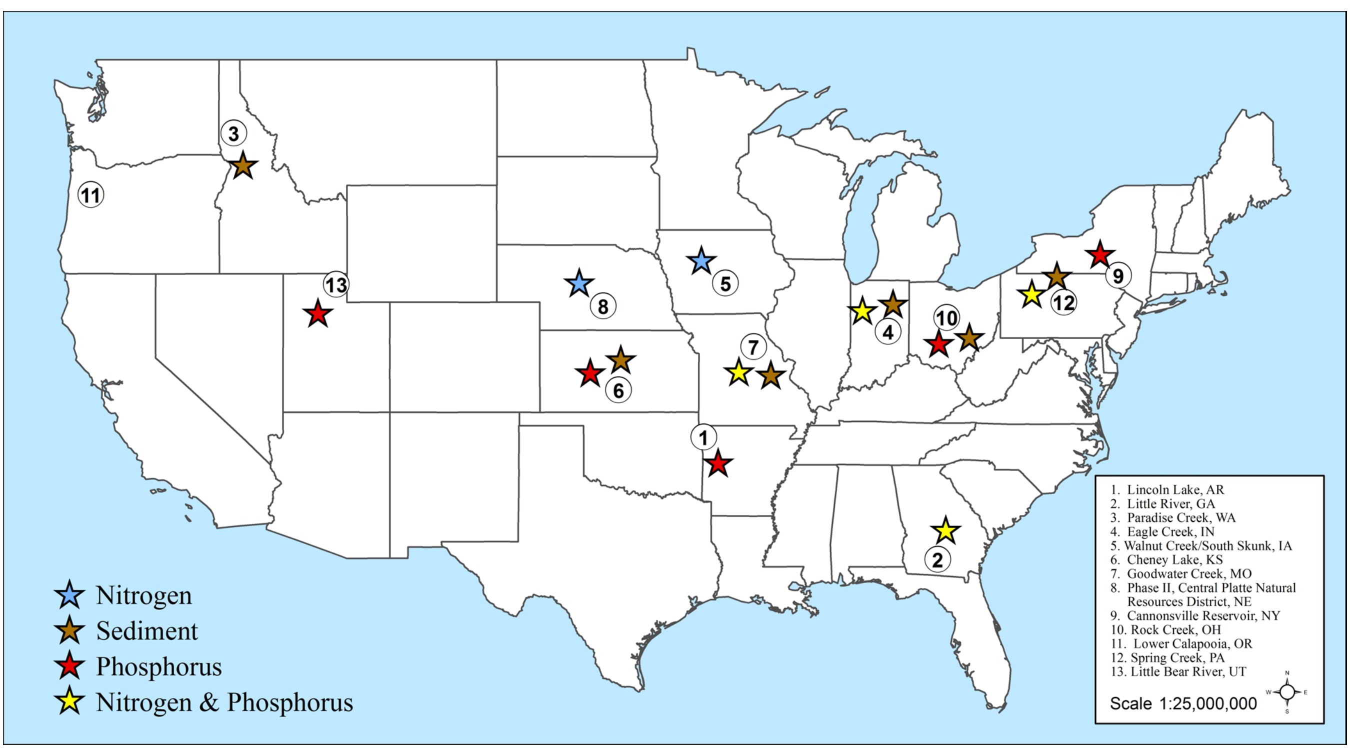

Fig. 2. Locations and pollutants of concern for the National Institute of Food and Agriculture (NIFA) Conservation Effects Assessment Project (CEAP) Watershed Studies Project.

- inappropriate water quality monitoring (Meals et al., 2012b),
- misinterpretation of pollutant hydrologic transport pathways (Arabi et al., 2012b),
- installation of the incorrect practice(s), insufficient implementation or use of practices, and/or practices inappropriately placed (Osmond et al., 2012c,d),
- farmer reluctance to adopt practices for a multitude of reasons or maintain them over time (Hoag et al., 2012b; O'Connell and Osmond, 2018), and/or
- lag times inherent in the land–water interface (Meals et al., 2010; Sharpley et al., 2013).

Economic Considerations

Farmer implementation of CPs that avoid, control, and/or trap P is essential to protect water quality. Therefore, understanding producer decision-making around adoption is critical. The avoidance decision will be heavily biased to economic returns, that is, on the cost of P—the amount used and its price (Hoag et al., 2012b; Ribaudo, 2015). Studies have shown that P fertilizer is price inelastic (Garnache et al., 2016), meaning that use does not vary much with price. Therefore, there is little reason to expect that raising the cost of P fertilizers will reduce use. For example, a 100% fertilizer tax had only a negligible effect on simulated P runoff in southern Michigan (Egbendewe-Mondzozo et al., 2013); even a 900% tax on P fertilizer could not reduce demand by a targeted 40% in the Minnesota River valley (Westra, 2001). Similarly, the price of manure is typically not high enough to justify hauling it very far from its source (Adhikari et al., 2005). Any value of manure P is often overshadowed by a producer's need to dispose of it at least cost (Hoag and Roka, 1995). Therefore, for most producers, it will be more effective to show them how to increase fertilizer or manure use efficiency so they can use less P, rather than rely on increased P prices to shift behavior.

Controlling and trapping will be viewed less favorably than avoiding by farmers because these practices primarily, though not exclusively, cost money but do not generate any revenues (Hoag et al., 2012b; Ribaudo, 2015). A number of public and private conservation programs are aimed at increasing the profitability of conservation systems, or a producer's interest in stewardship, to motivate adoption of control and trap systems. The NRCS, land-grant extension, nongovernmental organizations, and many others offer educational programs and technical assistance related to controlling and trapping CPs. In addition, many federal conservation programs, such as the Environmental Quality Incentives Program, and state and county programs offer financial assistance in the form of subsidies, tax breaks, and cost-sharing to farmers who adopt control or trap CPs. Although education, technical assistance, and financial aid have all helped with CP adoption, success is limited as many methods to control and trap P are not profitable at the farm scale. For details, the literature on program adoption is extensive and widely available (Baumgart-Getz et al., 2012; Hoag et al., 2012b; Liu et al., 2018). In summary of that literature, while education and technical assistance are important, financial incentives and a strong understanding of the local agriculture have repeatedly been found to be the most important contributors to CP adoption (Baumgart-Getz et al., 2012; Hoag et al., 2012b).

There is also potential in control and trap whereby society can effectively buy P abatement. From a benefit standpoint, society can find value to justify spending tax dollars on P abatement. For example, agriculture contributes over 70% of total N and P to hypoxic zones in the Gulf of Mexico (Alexander et al., 2008) and about a third of the total P load in the Baltic (Turner, 2001). Turner (2001) estimated that the total economic benefit of reducing eutrophication levels in the Baltic Sea is about $10 billion. Dodds et al. (2009) estimated that the annual cost of freshwater eutrophication in the United States from N and P exceeds $2 billion. Likewise, the USEPA (2015) documented through multiple studies many of the ways that nutrient pollution in water costs society. For example, one USEPA study showed tourism losses of $37 to $47 million led to business closures in Grand Lake St. Marys, Ohio (Davenport and Drake, 2011). In Maine, pollution cost shellfish fishers $3 million per year in lost catch.

The costs of trap and control CPs also must be considered (Liu et al., 2018; Ribaudo, 2015). To date, a host of education programs, technical assistance, and subsidies have been somewhat effective at increasing CP adoption, but these efforts lack overall coverage (Hoag et al., 2012b; Ribaudo, 2015). Part of the coverage problem lies in the technical challenges discussed above, but another part rests in finding out what motivates farmers. Adoption decisions are complex and have been examined in thousands of studies (e.g., Baumgart-Getz et al., 2012; Liu et al., 2018). However, it largely comes down to how much compensation farmers are offered (Hoag et al., 2012b). As adoption of CPs has been slow (Ribaudo, 2015), incentive programs will likely have to increase payments to producers to spur acceptance. One study even found that farmers might demand a conservation premium for adoption that doubles the cost of implementation (Motallebi et al., 2016). Most producers interviewed by Motallebi et al. (2016) indicated that they would not adopt CPs if incentive programs were priced for farmers to "break even" with the cost of adoption.

Outside forces, such as regulations, taxes, subsidies, education, or technical assistance, as well as more elaborate pollution trading programs or auctions, can be used to bring about behavior change and promote CP adoption. Palm-Forster et al. (2017) used experimental auctions to test farmer preferences for conservation incentives to promote voluntary abatement of P. The researchers found that farmers favored programs that provided financial support through subsidies or tax savings, compared with programs that offered crop insurance for any reduced yields attributed to CPs or a commodity price premium tied to stewardship certification. These results confirm that paying for pollution reduction is an effective tool for increasing CP adoption, especially when paired with education and technical assistance.

Numerous economic and social nuances can influence CP adoption. Technical solutions that help farmers make more money by using less P will help, as will finding more effective ways to educate or encourage adoption (Baumgart-Getz et al., 2012). However, the macro-factors pose greater challenges. Farmers are not sensitive to price. Therefore, the best hope of reducing P losses lies in increasing the efficiency of P manure and fertilizer use and in paying farmers to adopt CPs to control and trap P. Paying incentives could be an expensive proposition, as multiple conservation measures may be needed to reduce P losses at a large scale. Alternatively, regulating farmers, rather than relying solely on education and subsidies to increase voluntary adoption of CPs as has been largely done in the past, would push change. Some change would occur at low cost through regulation, but pushing too hard could threaten farm livelihoods, particularly small-holdings, which could in turn contribute to a destabilization of rural economies and infrastructure. Finding an appropriate balance between regulation and incentives will be difficult, but there are better solutions with reasonable costs.

The Future of Conservation Practices and Phosphorus Reduction

Phosphorus delivery is dependent on hydrology. Altered rainfall patterns are expected due to climate change, which will greatly affect hydrology and, therefore, P delivery to sensitive waterbodies. Lehmann et al. (2018) used a statistical model to demonstrate that heavy rainfall has increased in the central and eastern United States, northern Europe, and Russia but decreased in central Africa. It is predicted that rainfall and runoff patterns will become even more extreme. Yin et al. (2018) showed that runoff rates are increasing faster than precipitation, which will affect the extent, magnitude, and duration of P transport.

Multiple researchers have shown the effects of climate change on increased rainfall, runoff, and P loading. Jarvie et al. (2017) demonstrated that 35% of the increase in agricultural SRP losses in the Western Lake Erie watershed was the result of increased runoff due to changes in rainfall pattern associated with climate change. Ockenden et al. (2016) conducted an integrated assessment with a large group in England; the group determined that the majority (90%) of the total P load is lost during the highest discharge storms from multiple watersheds. As winter rainfall is predicted to increase, it is expected that P transfers will escalate by as much as 30% in some English watersheds (Ockenden et al., 2017). Line and colleagues (D.E. Line, personal communication, 2019) demonstrated that high exports of total P from three North Carolina watersheds were two- to threefold greater (0.94, 1.87, and 2.20 kg ha yr^{-1}) for the hurricane year (2016) than loads from the years before and after the hurricane; as such, increased hurricane activity will affect P transfers in impacted areas.

Because climate change will intensify duration and intensity of rainfall, P losses from all pathways will increase. Current CPs are often insufficient to retard and reduce P under storm flow conditions; these practices will become woefully inadequate as the climate changes. If serious progress in reducing P losses to our water resources is to occur, we must focus on more intentional, targeted watershed planning, implementation, and management. Lessons that can be applied from the NIFA-CEAP watershed project synthesis are summarized here (Osmond et al., 2012c).

Before CPs are implemented, the following steps are vital for intentional, deliberate, and effective watershed planning:

- Establish a network of stakeholders who are willing to set goals for implementing CPs on a sufficient scale to influence water quality.
- Define objectives in a collaborative approach that facilitates positive public discussion and includes farmer input about trade-offs between water quality and farming operations.
- Accurately identify the water quality impairments, pollutant(s), pollutant source(s), and hydrologic transport pathways.
- Match the CP(s) to the pollutant(s), pollutant source(s), and hydrologic transport pathways.
- Determine the critical source areas within the watershed where most of the practices are implemented to lower costs and improve the effectiveness of conservation.

Watershed planning is not simple; none of the steps are straightforward, and considerable expertise is involved. Determining critical source areas can be very difficult and will depend on the pollutant(s) and cropping system(s). Three NIFA-CEAP projects (Kansas, Missouri, and Utah) using different techniques determined that less than 30% of CPs were placed in the critical source area (Osmond et al., 2012c,e). If the watershed is tiled, all drained fields will be part of the critical source area, even if P is the pollutant of concern.

Watershed planning is difficult in part because expertise and tools are limited. Recently, a planning tool framework, Agricultural Conservation Planning Framework, was developed and expanded by the USDA-ARS (Tomer et al., 2013, 2017). The framework develops multiple strategies that can be evaluated by farmers and watershed planners and crosses scales from fields to watersheds. These types of tools are important because they provide technical guidance, but more important, they offer scenario-building features, which allows communication among the different groups or agencies.

During and after the watershed planning process, an effective outreach and education program to engage farmers in CP(s) implementation must be developed. These programs, which are critical for successful implementation and continued utilization, include the following:

- Understand how farmers make decisions about CP adoption and maintenance. There is significant literature regarding farmer decision making. Although economics and agronomics are important in CP decision-making, many additional factors affect individual farmers' decisions (e.g., personal beliefs, social networks, geographic areas, scale of the farm); decision-making is ultimately multidimensional and personal for each farmer (Hoag et al., 2012b; O'Connell and Osmond, 2018; Reimer et al., 2013, 2014; Woods et al., 2014).
- Develop a comprehensive outreach education plan with goals, objectives, target audiences, implementation strategies, and responsibilities at the beginning of the project; evaluate progress and use adaptive management throughout the process to optimize educational outcomes.
- Provide sufficient personnel so there is sustained one-on-one contact between conservation agency staff and farmers.
- Encourage farmer-to-farmer learning opportunities.

Finally, to enhance inducements for CP adoption, the following steps are necessary:

- Develop better incentives that include funding, flexibility, and ease of management. Often, ease of management (time management) is more important to producers than are practice costs (Luloff et al., 2012).
- Recognize that some practices are easier or harder for farmers to adopt. The two most disliked CPs among the

NIFA-CEAP farmers were riparian buffers and nutrient management (Luloff et al., 2012; Woods et al., 2014). Producers disliked riparian buffers because land was removed from production, and they disliked nutrient management because they either did not believe the recommendations, the practice was too time intensive, or both.
- Recognize that despite the best possible incentives, some farmers will not adopt CPs.
- Follow up after installation of CPs by tracking the location and timing of implemented CPs and making sure this information is available in a format useful for project assessment.
- Ensure that operation and maintenance of CPs are sustained over time, and encourage continued use of the practices, particularly for those with less farmer acceptance. The Pennsylvania project reviewed through the NIFA-CEAP protocol documented discontinuation of exclusion fencing as land changed hands or was transferred to other family members (Osmond et al., 2012a).
- Ensure that new practices do not reduce or transform the functionality of CPs, especially as farmers alter agricultural management. For example, conservation tillage that replaces terraces may or may not decrease erosion (Osmond et al., 2012e), or changes in practice may have unintended consequences (Jarvie et al., 2017).

After the planning stage, there must be intentional and deliberate CP implementation that accounts for all the factors informed by planning. Additionally, CPs must be tailored to the farm management systems in which they are installed. Finally, plans must be updated in a continuous, adaptive framework.

Intentional watershed management will involve ensuring amounts of P adequate to grow crops but low enough to reduce off-site losses. Intentional watershed management will also need to consider the contradictory effects of practices (i.e., conservation tillage reducing particulate P but increasing SRP) or practices that can become P sources (i.e., riparian buffers or wetlands). Intentional watershed management does not only involve technical solutions but must proactively engage landowners. This will take substantial human and economic resource investment to assist producers to cover the costs of appropriate systems of CPs, and fund watershed managers to ensure compliance. Working with farmer focus groups, Ockenden et al. (2017) discussed the difficulty of intentional watershed planning as the climate changes and ever greater conservation is required. Additionally, both Jarvie et al. (2017) and Smith et al. (2018) discussed "the law of unintended consequences" as drainage intensity, modifications in fertilization rate, placement, and timing, tillage, and rainfall intensity and timing have changed to increase P losses into Lake Erie. Intentional watershed management will need to be adaptive to reduce P losses, avoid unintended consequences, and protect water quality as the climate changes.

Conclusion

Reducing P losses from agriculture will require more action by researchers to ensure that conservation implementation agencies (i.e., NRCS and soil and water conservation districts) and farmers understand the effects of CPs on P transfers and forms (erosion, surface SRP, and P leaching). It will be important for agencies and farmers to learn about the trade-offs between controlling sediment, N, sediment-attached P, surface SRP runoff, and leached P relative to different CPs, systems of practices, cropping systems, and soils. Agencies will need to be less prescriptive regarding CPs, use watershed planning and implementation, and provide adaptive P management strategies to farmers. Additional resources will be required to more effectively work with farmers to implement and maintain practices at a watershed scale.

A key factor in supporting practice development, implementation, and maintenance is the need for long-term data sets (e.g., STP, P losses in drainage, and water quality monitoring) that will provide the technical basis and support for all CPs. As we look to the future, we need to accept that some approaches have simply not been effective in overcoming societal, economic, and regulatory barriers to increased CP adoption. Instead, with more collaborative approaches, the broader watershed community, supply chain companies, and not-for-profit and nongovernmental organizations should become involved in conservation adoption, tracking, and compliance. These approaches could also overcome some hurdles, such as the targeting of vulnerable areas that can be perceived to be unfair to some farmers.

It is imperative that agricultural tools—CPs—be used more effectively to reduce and retard off-site P losses. We also must consider the need for new and innovative CPs that can improve control of P leaching and legacy stores of STP and that can mitigate the increased P losses expected as the climate changes. Without immediate changes to CP implementation, P losses will increase due to climate change with the concomitant degradation of water quality. These changes must be done at a watershed scale in an intentional and transparent manner.

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