

Reducing the Impacts of Agricultural Nutrients on Water Quality across a Changing Landscape



Complexities and difficulties associated with nutrient cycling and transport processes, management decisions and practice trade-offs, and federal conservation program effectiveness create immense challenges to achieving and measuring water quality improvement goals. (Photo courtesy of Lisa Gjersvik.)

ABSTRACT

Agricultural productivity in the United States has doubled over the last 50 years through agricultural intensification and adoption of new innovative technologies. Although efficiency of our agricultural systems has increased, water quality remains a concern with minimal measured improvements observed nationwide. The purpose of this paper is to provide an overview of the processes, conservation practices, and

programs that influence the impact of agriculture on surface and groundwater quality. Complexities and difficulties associated with nutrient cycling and transport processes, management decisions and practice trade-offs, and federal conservation program effectiveness create immense challenges to achieving and measuring water quality improvement goals. Development of more precise nutrient recommendations, advancement of water monitoring methods to better differentiate among

potential nutrient sources, design and implementation of novel conservation practices that address dissolved nutrient loss and in-stream nutrient retention, increased knowledge of processes influencing nutrient supply and transport, and increased cost-effectiveness of conservation programs integrating regional and industry-based collaboration are needed to continue to improve water quality in agricultural landscapes.

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INTRODUCTION

The Food and Agriculture Organization projects global population increases of two billion people by 2050 (FAO 2017). In order to sustainably meet the corresponding increased demand for food, feed, and fiber, agricultural productivity must focus on increased efficiency and decreased environmental losses. Agriculture serves as a foundation of the U.S. economy, supporting its relationship with the world through trade and humanitarian aid. Historical trends in U.S. cropland highlight the developments that have propelled U.S. productivity to today's record high levels, with crop yields, a core measurement of productivity, doubling since the 1980s (USDA–NASS 2018). As crop productivity increased, fertilizer use efficiency also increased (Figure 1) (IPNI 2015, 2018). These increases have been achieved through the adoption of improved agricultural technology, including crop breeding advancements, the availability of agricultural pesticides, innovations in farm machinery design, and precision agriculture methods (Wang et al. 2015).

Gains in production intensity over the 20th century have been accompanied by specialization of cropland farming

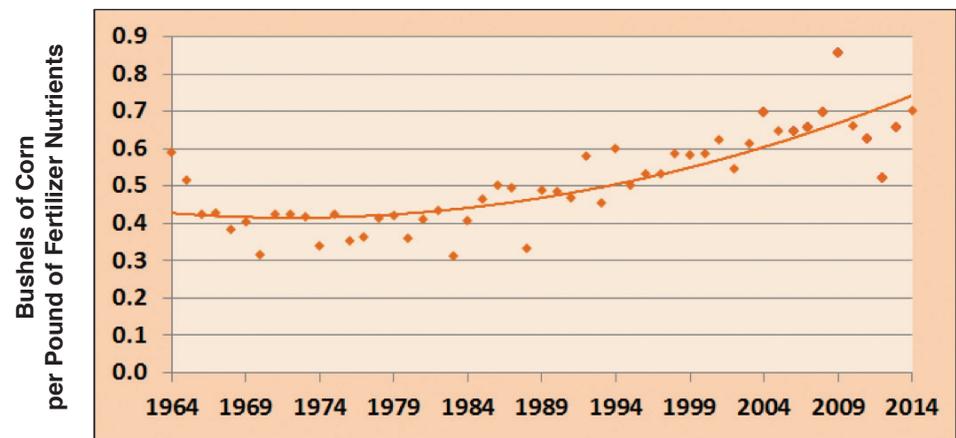


Figure 1. Productivity calculated by comparing the bushels of corn produced in the United States per pound of fertilizer nutrients applied from 1964 to 2014 based on U.S. Department of Agriculture (USDA)—National Agricultural Statistics Service yield data and USDA—Economic Research Service fertilizer application data shows that corn fertilizer use efficiency nearly doubled between 1980 and 2014 (IPNI 2015).

(Kleinman et al. 2018). Specialization has increased cost efficiencies (Winsberg 1982), but it has reduced on-farm crop diversity. Individual U.S. farms typically produced four to five commodities prior to World War II; however, the number of commodities decreased to two—primarily corn and soybean—by 2000 (Dimitri, Effland, and Conklin 2005). Specializa-

tion of agriculture has also resulted in animal production systems that are often geographically separated from the major cropping systems that serve as the source of feed. Uncoupling of cropping and animal production systems can result in regions in which cropland is primarily dependent on mineral fertilizers for crop nutrition, whereas other regions

with dense animal production and less cropland cannot feasibly land-apply their nutrient-rich manure without risk of applying at rates higher than crop utilization or adding transportation costs. Specialization of agriculture has altered flows of resources, particularly nutrients, compared to historical norms, which can contribute to an array of environmental concerns (Sharpley et al. 2013).

Supplying external inputs of nutrients such as nitrogen (N) and phosphorus (P) to cropland in order to maximize crop production was first recognized nearly 200 years ago (Sprenkel 1827), and today 40 to 60% of crop yield is attributable to fertilizer (Stewart et al. 2005). Improvement in crop production practices, the addition of precision farming tools, and modern data informatics support higher crop production to resource utilization ratios than previously possible (Balafoutis et al. 2017). Nevertheless, U.S. croplands continue to contribute to inland (e.g., streams, rivers, lakes) and coastal (e.g., estuaries, bays) water quality impairment. Agricultural nutrient water quality concerns focus on N and P, because these nutrients contribute directly to water quality degradation through eutrophication when lost from cropped fields through erosion, surface runoff, or leachate to environmentally sensitive groundwater or downstream water bodies.

Although many sources, such as wastewater treatment plants, septic systems, or urban nonpoint sources, contribute nutrients to water bodies, agriculture remains a significant source in many areas of the United States. Nutrient losses to receiving waters can result in eutrophication, which manifests as the nuisance growth of algae and algal blooms. Algal growth in surface waters directly responds to nutrient enrichment (Smith 2003). Bloom conditions might result in major water quality disruptions such as objectionable taste and odor in drinking water, fish kills, or acute toxic or poisoning events (i.e., harmful algal blooms). For instance, more frequent and intense algal blooms have occurred in Lake Erie due to increased soluble nutrient contributions from river tributaries (Kane et al. 2014). In August 2014, microcystin concentrations reached dangerous levels, which resulted in a three-day

drinking water supply shutdown for the City of Toledo, Ohio, impacting 500,000 residents.

The number of coastal areas in the United States experiencing low oxygen conditions or hypoxia resulting from nutrient enrichment has increased from 12 documented cases in 1960 to more than 300 cases in 2010 (Committee on Environment and Natural Resources 2010). In some areas of the country, such as the Chesapeake Bay and the mid-Atlantic, hypoxia-related issues have persisted since the 1950s. Concurrently, a growing prevalence of hypoxia in the Gulf of Mexico and south Atlantic has emerged since the 1980s (Committee on Environment and Natural Resources 2010). Similar to the toxic algal bloom reported in Lake Erie in 2014, toxic cyanobacteria have been implicated in human and animal illness and death in at least 43 states, with 19 states posting public health advisories in 2016 alone (Graham, Dubrovsky, and Eberts 2017).

Looking toward the future, U.S. agriculture faces an unprecedented challenge—support growing domestic and global agricultural product demands (FAO 2017) while minimizing environmental impacts on local and regional water resources. With the adoption of new technology, innovative conservation practices, and enhanced efficiency, increasing crop production and shrinking the environmental footprint of agriculture do not have to be at odds. The modern soil and water conservation movement was born largely over concerns about cropland agriculture in the first half of the 20th century. The Dust Bowl of the Great Plains in the 1930s, the general loss of farmland productivity, and impacts to water, air, and ecosystem services spawned conservation in the United States (Bennett and Chapline 1928; Leopold 1949). Conservation activities, however, can compete with other priorities, including the pursuit of profitability, belief systems, and local cultural practices (Ervin and Ervin 1982; Knowler and Bradshaw 2007).

The purpose of this publication is to provide background detailing the underlying complexities and challenges faced by U.S. cropland agriculture to sustainably meet water quality nutrient reduction goals while advancing productivity. There

are six main sections within the publication, with the first section describing soil N and P nutrient cycles, which serve as the foundation to both crop production and environmental loss. The second section describes the current knowledge of factors that control nutrient loss to groundwater and surface water, while the third and fourth sections document nutrient management practices and common conservation practices, respectively, that farmers, land managers, and conservation professionals currently use to decrease agriculture's impact on water quality. Section five highlights environmental policies, incentives, and programs, past and present, related to cropland agriculture and water quality. The final section summarizes the challenges and associated needs for agriculture to move toward improving efficiency, meeting water quality goals, and sustaining crop production levels.

SOIL NUTRIENT CYCLES

Nutrients, such as N and P, exist in the soil in multiple forms or pools. Nutrient cycles describe the pools in which nutrients reside and how they move into, out of, and among these different pools. In order to increase cropland productivity and reduce agriculture's environmental impact, land managers must understand nutrient cycling. Farmers apply mineral fertilizers and manures to fields to meet crop nutrient needs. The quantity and form of nutrient in the soil and available to the plant, the amount of crop biomass produced along with the tissue nutrient concentration, and the efficiency of the applied nutrient combine to determine total nutrient requirements for any crop. Four main processes are prevalent in soil nutrient cycling:

1. Addition—the input of nutrients to the soil system
2. Translocation—movement of nutrients within the soil without changing form
3. Transformation—chemical or biological conversion between chemical assemblages containing the element
4. Loss—the movement of nutrients out of the soil system through the harvested grain, with water, or to the atmosphere as a gas

Phosphorus Cycle

Mineral fertilizers, manures, and other organic residues (e.g., biosolids, plant residues) contain P in various forms, including plant available (soluble P), slowly available (P minerals and P attached to mineral surfaces), and plant unavailable. Phosphorus must be dissolved in the soil solution to be taken up by crops, typically as orthophosphate and soluble organic P compounds. Once P is removed from the soil solution, it is replenished by the residual P in the soil. The transformation of organic P forming into plant-available P occurs through mineralization. Inorganic P pools can also replenish the soil solution either through soil P minerals dissolving into the soil solution or desorption of P attached to soil particles such as clay or minerals containing iron or aluminum (see Figure 2).

Similar to the replenishing of soil solution P when the concentration of P becomes low, P can also be removed from the soil solution when P concentrations become too great. Depending on the soil pH, dissolved P will precipitate and form solid calcium phosphate minerals (high soil pH) or iron and aluminum phosphate minerals (low soil pH). Thus, plant-available P is often at its greatest concentration between a soil pH of 6.0 and 6.7. Phosphorus can also be removed from the soil solution and attach to soil particles like clays or iron and aluminum minerals via adsorption. Phosphorus can move with water running over the soil surface or percolating downward through the profile in either dissolved or particulate forms. Particulate P losses include P attached to eroded soil particles, P in plant residues, or recent P inputs moved through incidental transfer.

Nitrogen Cycle

The complexity of the N cycle exceeds that of the P cycle due to the variability introduced by the many microbial processes involved. Plants take up N from inorganic forms. Mineral fertilizer, manures, and other organic residues added to the soil as fertilizer contain N in both inorganic plant-available (ammonium $[\text{NH}_4^+]$, nitrate $[\text{NO}_3^-]$) and organic (amino acids, amino sugars, complex organic molecules) forms. The atmosphere also adds N to the soil cycle through

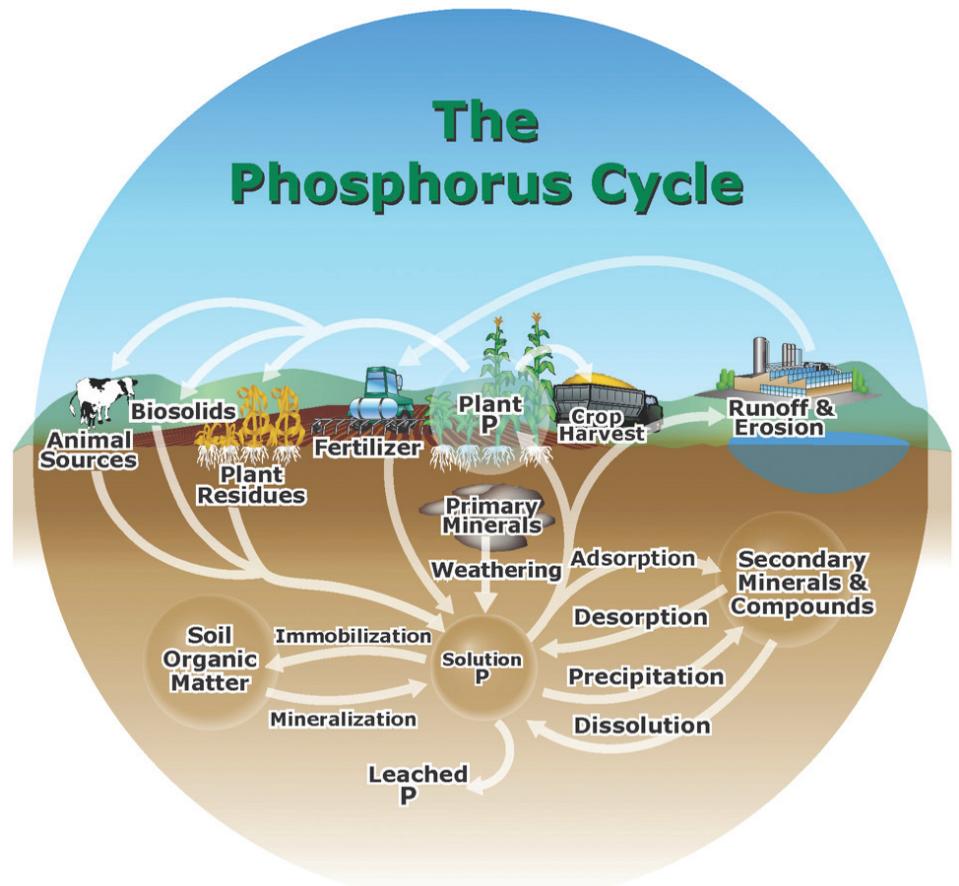


Figure 2. The P cycle. (Diagram courtesy of the International Plant Nutrition Institute.)

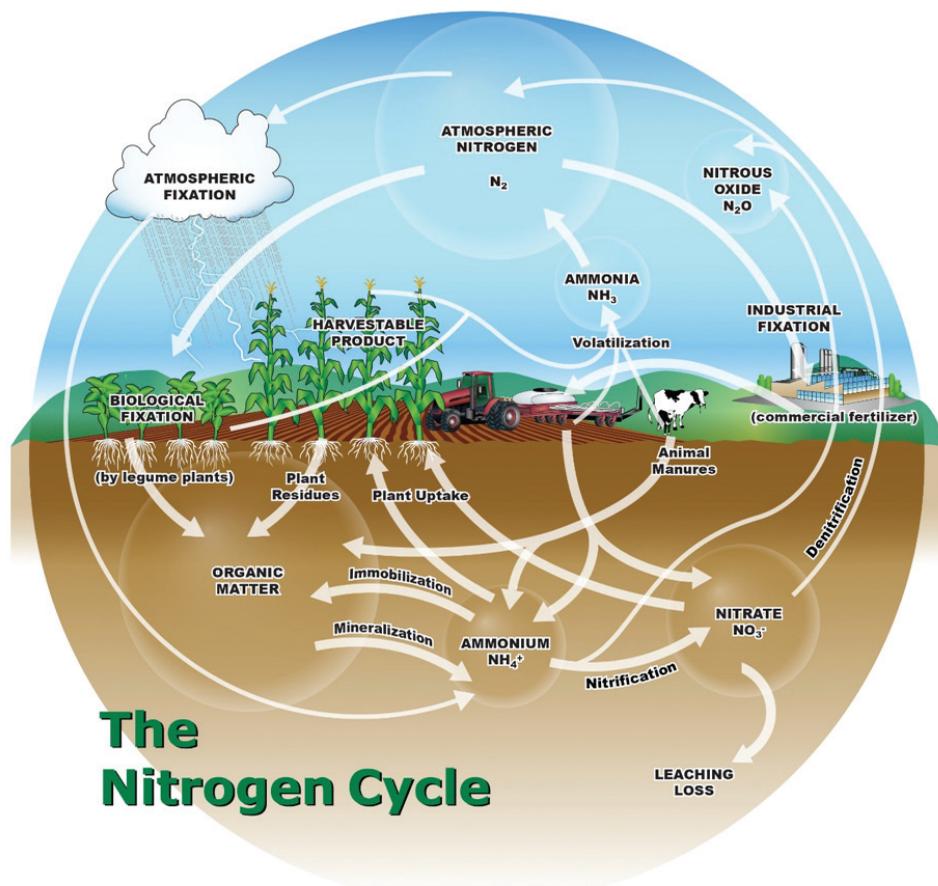
fixation and deposition. Nitrogen-fixing bacteria living in the soil or colonized on the roots of legumes, such as soybean, alfalfa, and clover, can convert nonreactive elemental nitrogen (N_2 gas) from the atmosphere into plant-available forms. In addition, thermal fixation, either natural (e.g., lightning) or anthropogenic (e.g., internal combustion engines), converts atmospheric N_2 to NH_4^+ , which can be deposited on the soil surface when it rains (see Figure 3).

Organic N in the soil can be transformed by soil microbes into plant-available NH_4^+ through mineralization. Ammonium at the soil surface can volatilize to the atmosphere as ammonia gas or it can be converted in a secondary process through nitrification by soil microbes to NO_3^- . Nitrate can be converted to nitrous oxide, a greenhouse gas, or N_2 gas through denitrification. Nitrogen lost from the soil system via the movement of water can occur as NH_4^+ , NO_3^- , or organic

N, with the form of N largely determining whether it is transported in surface runoff, attached to soil particles, or as leachate. As with P, N can be lost through surface runoff either dissolved as NO_3^- , NH_4^+ , or soluble organic molecules, or in particulate form. In addition, N can be lost as NO_3^- , which easily moves downward through the soil profile and connects to groundwater flow paths.

CONTROLS ON NUTRIENT LOSS TO SURFACE WATERS AND GROUNDWATER

Nutrient loss from agricultural fields and watersheds is determined by the complex interaction among numerous physical, chemical, and biological variables. These variables can be separated into two main factors—nutrient supply and nutrient transport—which are dis-



The Nitrogen Cycle

Figure 3. The N cycle. (Diagram courtesy of the International Plant Nutrition Institute.)

cussed independently in this section. It is important to note, however, that it is the combination of both factors that ultimately determines the risk of nutrient loss.

Nutrient Supply

Crop harvest removes nutrients from the field, which farmers replace through manure and fertilizer application. Fertilizers and manures have the potential to elevate nutrient concentrations in surface runoff and subsurface leachate, particularly if applied beyond crop need. Although multiple site and environmental factors control hydrologic processes, generally the risk of nutrient loss in surface runoff is greatest in the first precipitation or irrigation event after nutrient application, with losses generally decreasing over several weeks to months (Gascho et al. 1998; Kleinman et al. 2011). Thus, high rates of fertilizer and manure applied to soils prior to periods of snowmelt or high rainfall, or shortly before irriga-

tion events, produce the greatest potential risk for elevated nutrient loss in runoff to downstream water bodies (King et al. 2018; Kleinman and Sharpley 2003; Williams et al. 2011).

The magnitude of nutrient concentrations in runoff or leachate from recently applied nutrient sources is generally a function of the nutrient application rate, the nutrient form, the nutrient source solubility, and the dominant hydrologic transport mechanism (Kleinman et al. 2007; Mulla and Strock 2008). Fertilizer and manure application placement and timing can also affect both the magnitude and the duration of incidental nutrient transfer. Application placement methods that incorporate or mix the nutrient source into the soil often result in decreased nutrient wash-off in runoff and leachate compared to surface application methods (Little, Bennett, and Miller 2005; Williams et al. 2018).

Although incidental transfers of el-

evated nutrients lessen during the weeks and months after nutrient application, over time a portion of nutrient inputs can accumulate in the soil across fields and watersheds. Accumulated nutrients can be remobilized years, decades, or even centuries after their initial application and function as a chronic source of nutrients to downstream water bodies. This is often referred to as “legacy” nutrient loss (Sharpley et al. 2013) and results from local and regional nutrient imbalances (i.e., nutrient inputs > nutrient removal) and the buildup of nutrients in soils and groundwater over time. It is estimated that annual N accumulation in soils throughout the Mississippi River Basin from 1957 to 2010 was between 25 and 70 kilograms (kg)/hectare (ha) (Van Meter et al. 2016), whereas P accumulated across North America between 1965 and 2007 at an average annual rate of 11 kg/ha (Sattari et al. 2012).

Phosphorus accumulation in surface soils (0–20 centimeters [cm]) represents the most pervasive legacy source of P to the environment (Sharpley et al. 2013). Concentrations of total P in agricultural soils can range from two to ten times greater than background concentrations under forest soils due to the legacy of historical P additions (Vitousek et al. 2009; Sattari et al. 2012). This legacy buildup can be spatially disproportionate across a watershed, leaving some agricultural soils below critical soil P fertility levels. A large fraction of total P loss in surface runoff and subsurface drainage is in particulate form (i.e., P bound to soils); thus, erosion is an important mechanism for mobilizing P, including legacy P. Sharpley (1980) showed that eroded sediments can be enriched with P (up to five times) relative to the bulk soil due to the preferential erosion of fine soil particles with greater P content. The sorption and desorption of P by soil is also important for P losses in both surface and subsurface pathways. Correlations have been observed between P concentrations in soils and dissolved P concentrations in surface runoff (Pote et al. 1996) and subsurface tile drainage (Duncan et al. 2017a). The buildup of the P supply in soils can therefore result in persistent P losses across multiple spatial scales (King et al. 2017).

Nitrogen can also accumulate in soils over time when nutrient application rates exceed nutrient removal by crops and persist as a chronic source of N loss to downstream water bodies (Van Meter et al. 2016). Additionally, N in the form of NO_3^- is highly mobile in the soil because of the negative charge, and it can leach below the crop rooting zone and into subsurface tile drainage or groundwater. In some locations, concentrations of NO_3^- in groundwater under agricultural land use are documented to be five to seven times greater than concentrations of NO_3^- under forest (Pionke and Urban 1985). In a long-term watershed study conducted prior to the 1990s, measured increases in groundwater NO_3^- concentration were approximately proportional to the increase in N fertilizer use over the same period (Böhlke and Denver 1995).

The percent of applied N that leaches to groundwater varies because of many factors. Puckett, Tesoriero, and Dubrovsky (2011) showed that a larger frac-

tion of applied N is leached to groundwater in irrigated areas, whereas in areas with subsurface tile drains and surface ditches, N is often diverted to surface water through tile drains, leaving less available to leach directly into groundwater. Research has demonstrated that even after four years of not applying N fertilizer in a corn-corn and corn-soybean rotation, irrigated sandy soils can still have NO_3^- leachate losses above drinking water standards, illustrating the challenge of meeting water quality goals in corn cropping systems (Struffert et al. 2016). Where artificial drainage is not present, excess water at the land surface can infiltrate the soil and enter the groundwater system. Groundwater can discharge into surface water bodies where a stream intersects the water table. In most aquifers, groundwater moves at very slow rates, with groundwater residence times ranging from 0 to >50 years in many cases (Lindsey et al. 2003). Thus, depending on the residence time of groundwater

and the history of N fertilization, legacy N can be supplied to downstream water bodies for years to decades in the future (Gilmore et al. 2016; Sprague, Hirsch, and Aulenbach 2011).

Nutrient Transport

Research across diverse agricultural landscapes in the United States, including the Mississippi River Basin (Gall et al. 2013) and Western Lake Erie Basin (Williams, King, Baker, et al. 2016), has shown that hydrological processes are an important component driving nutrient loss. Nutrient losses tend to be proportional to water flux as the result of both the buildup of nutrient supply (i.e., legacy nutrients) and the alteration of nutrient transport pathways. The alteration of nutrient transport pathways is perhaps most evident throughout the midwestern United States, where many agricultural fields are artificially drained with dense surface and subsurface drainage networks (Figures 4 and 5). Artificial subsurface

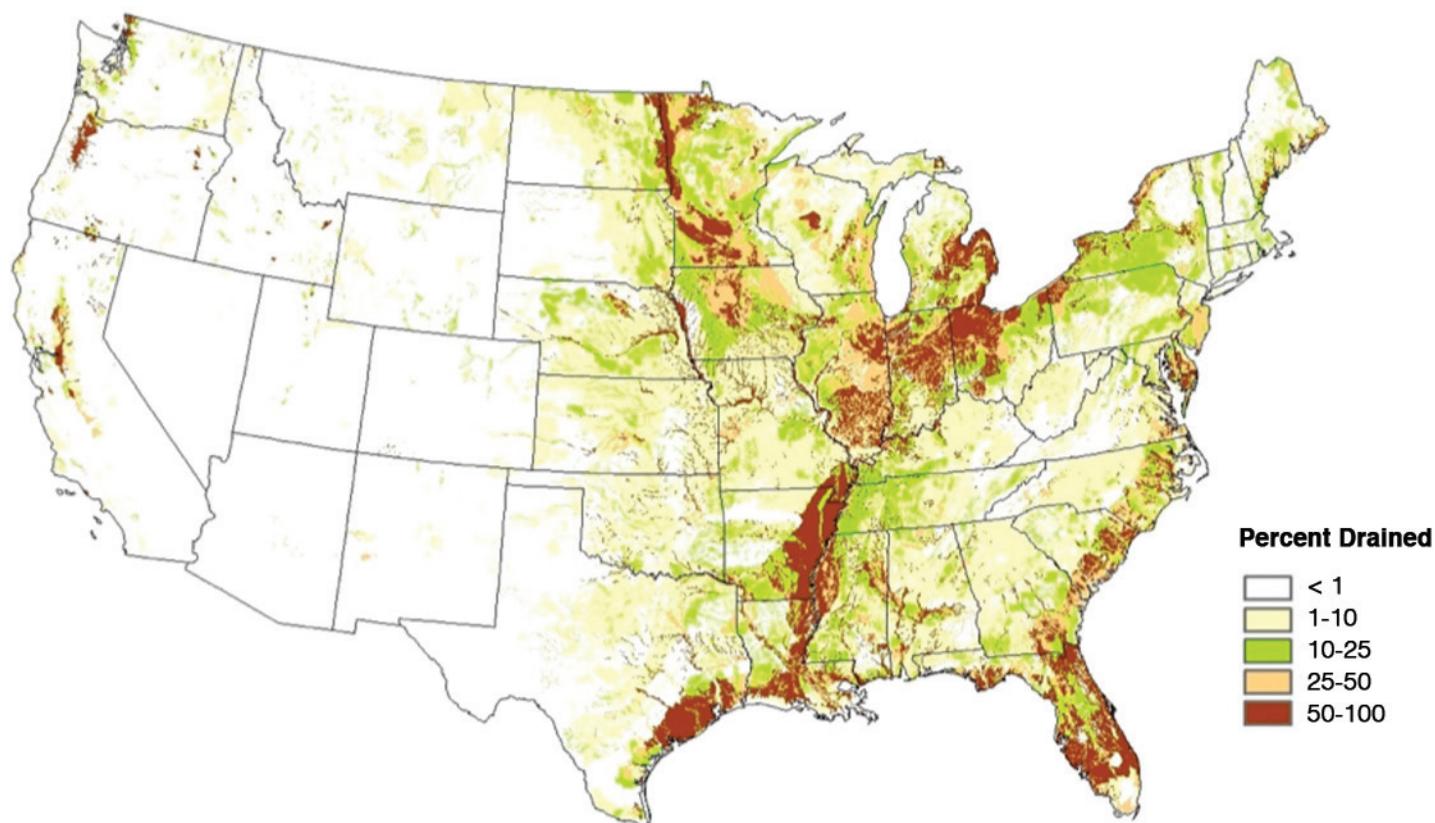


Figure 4. Artificial drainage of farmland soils has increased crop productivity, but as a result it created hydrologic connectivity between agricultural landscapes and waterways. In some regions, more than 50% of the cropland has been drained (Jaynes and James 2007).

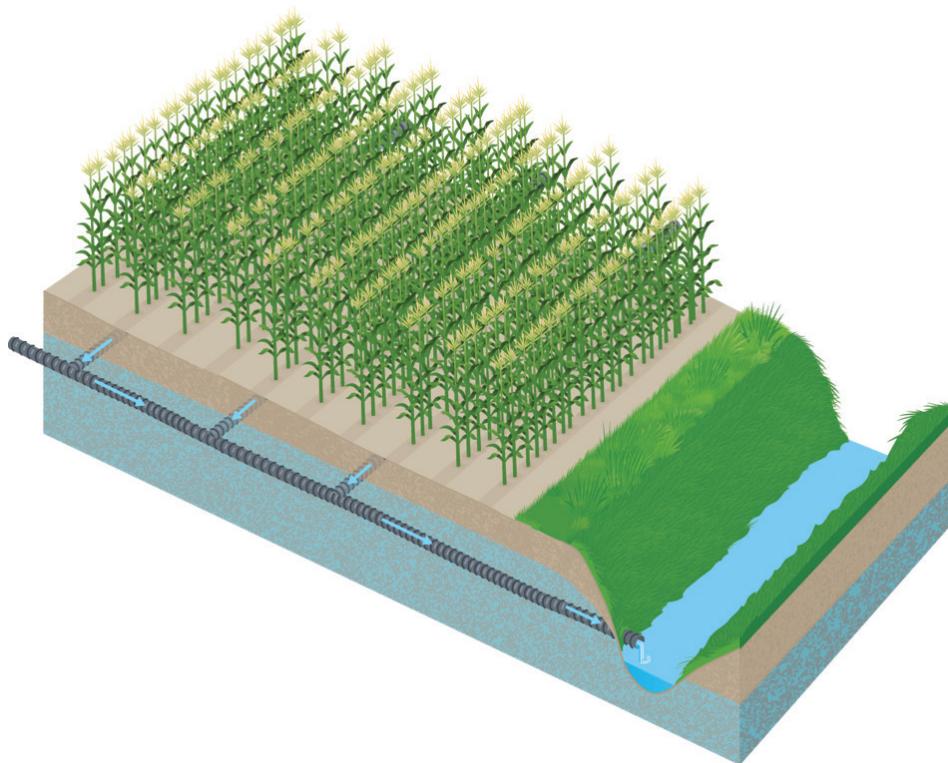


Figure 5. Agricultural fields are often artificially drained by a network of subsurface pipes, also known as tile drainage, to remove excess water off the fields to decrease the impact of flooding and allow for earlier spring planting. (Image courtesy of transformingdrainage.org.)

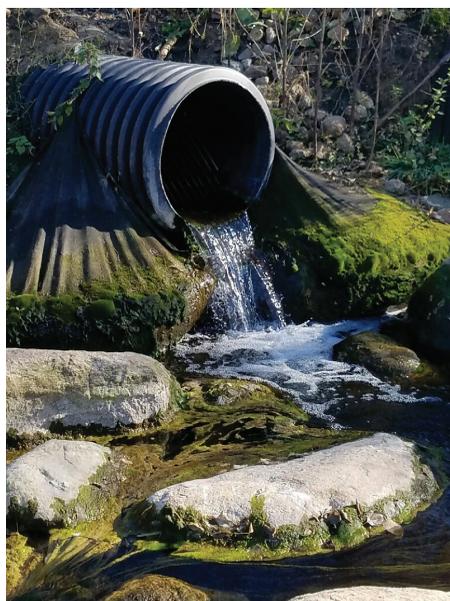


Figure 6. Artificial drainage networks, including subsurface tile lines and drainage ditches, increase hydrologic connectivity of agricultural landscapes. (Photo courtesy of Russell Derickson.)

drainage networks are often referred to as “tile” drainage, although the use of clay tile pipes has been replaced with the use of corrugated plastic tubing. Wetlands and soils with poor drainage throughout the Midwest have largely been drained over the past 150 years to facilitate crop production. For example, Bishop, Joens, and Zohrer (1998) estimated that between 95 and 99% of wetlands in Iowa have been lost due to artificial drainage. Efficient drainage systems provide numerous agronomic benefits, but they also greatly modify the magnitude, timing, and flow pathways of nutrient transport (King et al. 2015).

Surface ditches and subsurface tile drains increase the hydrologic connectivity of agricultural landscapes (Figure 6). For example, in the Prairie Pothole Region of the upper Midwest, surface water ponded in closed depressions is often routed via the artificial drainage network directly to a distant stream or ditch; whereas, prior to installation of the artificial drainage, this surface water (and

associated nutrients) would have remained isolated. Artificial drainage depth and spacing can also significantly affect nutrient transport. Ditches or tile drains spaced closer together often yield greater nutrient losses (Kladivko et al. 2004). Subsurface tile drains located deeper in the soil profile tend to increase nutrient loads (Strock et al. 2010). In watersheds with artificial drainage, the artificial drainage network can be the greatest contributor to nutrient loads at the watershed outlet. For example, subsurface tile drains accounted for 47%, 48%, and 62% of annual water, dissolved P, and NO_3^- loads, respectively, exported from a watershed in Ohio (King, Williams, and Fausey 2015; Williams, King, and Fausey 2015).

Agricultural management practices that influence soil surface properties, such as the amount of residue cover, aggregate stability, and surface crusting, can play a critical role in nutrient transport by altering the amount of water that infiltrates into the soil versus the amount of water that runs off. Tebrügge and Düring (1999) found that fields with long-term no-tillage were less susceptible to surface sealing and erosion, were more resistant to the effects of compaction, and developed continuous macropores, which improved water infiltration rates compared to fields with conventional tillage. The conversion of natural to agricultural ecosystems has resulted in substantial depletion of soil organic matter (Lal 2004). Soil organic matter influences soil structure, infiltration (Minasny and McBratney 2017), and nutrient retention (Lal 2009). For instance, fine-textured soils with lesser organic matter content may be more prone to develop preferential flow pathways, which can route nutrient-laden water from surface soils directly to subsurface tile drains (Jarvis 2007). Although P concentrations in surface runoff are often greater than concentrations in leachate (Pease et al. 2018), it is important to acknowledge that large dissolved P and NO_3^- leachate losses have been observed from no-till fields following surface application of fertilizers due to macropore transport (Daryanto, Wang, and Jacinthe 2017; Williams, King, Ford, et al. 2016).

Irrigation is important for reliable food production in many areas of the

United States, with approximately 22.7 million ha (56 million acres) of irrigated cropland in 2013 (USDA–NASS 2014). Excess irrigation water can become “irrigation return flow” and be transported to water bodies as both surface runoff and subsurface drainage water (Bjorneberg, Westermann, and Aase 2002). In many areas that require irrigation, natural rainfall does not typically produce either surface runoff or leaching (Bjorneberg et al. 2015); thus, irrigation and irrigation return flow that transports nutrients from agricultural fields to water bodies can significantly influence nutrient loss in these landscapes. For example, between 20 and 50% of water applied during furrow irrigation may run off a field depending on crop, management, water supply, and field conditions, carrying with it sediment and nutrients (Bjorneberg, Westermann, and Aase 2002). Irrigation practices can also increase the risk on NO_3^- leaching to subsurface tile lines and groundwater (Puckett, Tesoriero, and Dubrovsky 2011). The type of irrigation (e.g., furrow irrigation, sprinkler irrigation), frequency and timing of irrigation, and cropping system are important factors controlling nutrient loss from irrigated fields (Bjorneberg et al. 2015).

Although current and historic anthropogenic activities have substantially altered nutrient transport processes in agricultural fields and watersheds, climatic variables are also important in determining nutrient transport. Precipitation amount, duration, intensity, and timing not only influence the potential for nutrient transport within an event and the partitioning of water in surface and subsurface flow pathways, but also influence antecedent soil moisture, which can determine the potential for nutrient loss in subsequent events.

Increasing climate variability and frequency of extreme events are predicted over the next century (IPCC 2014), which is likely to influence both the quantity and quality of water transported from agricultural landscapes (Wang et al. 2018). Changes in the magnitude and variability of precipitation are expected to result in shifts in the seasonal timing and magnitude of flows (Bosch et al. 2014; Stone, Hotchkiss, and Mearns 2003). For example, climate scenarios modeled by

Masaki and colleagues (2014) found that more water, and therefore more nutrients, in large rivers was likely to be transported during a smaller fraction of the year across most of the United States. Phosphorus losses are expected to increase because of more intense precipitation (Ockenden et al. 2017), and the variability between dry and wet years will increase N losses and levels of concern in rivers (Loecke et al. 2017). Forecasted changes in air temperature and length of growing season may also influence the hydrologic cycle and potentially nutrient transport via changes in evapotranspiration (Marshall and Randhir 2008). As farmers adapt to changes in climatic variables, further alteration of the hydrologic processes controlling nutrient transport (e.g., increased subsurface drainage intensity, increased irrigation) may occur.

NUTRIENT MANAGEMENT PRACTICES TO DECREASE NITROGEN AND PHOSPHORUS LOSSES

Nutrient management not only has direct implications for crop productivity, but it can also strongly influence nutrient losses to groundwater and surface water bodies. To maximize crop productivity and minimize environmental impact, four primary components of nutrient management should be considered—the source of the nutrient, the rate of nutrient application, how the nutrient is applied, and when the nutrient is applied. Defining the right nutrient source at the right rate, at the right time, and in the right place within the context of a well-managed cropping system has been adopted by the fertilizer industry as the “4R Approach to Nutrient Stewardship” (Figure 7) and helps convey how nutrients should be managed to ensure alignment with economic, social, and environmental goals (IPNI 2012). This section reviews basic information on the 4R approach as it relates to N and P losses to groundwater and surface water.

Nutrient Source

The right source of nutrient is dependent on the nutrient content, its solubility, and whether it is regionally available.

Most commercially produced mineral fertilizers are highly soluble when applied to the soil and each contain different quantities of N and P, as well as other essential nutrients for crop development. Mineral fertilizers typically contain N and P in inorganic forms that are immediately available for crop uptake, but these nutrient forms can also be transported with water and potentially lead to large losses if a rainfall or irrigation event occurs shortly after application (Smith et al. 2016). Manure, which is composed of animal feces, urine, and, in some cases, bedding materials and water, is also a common nutrient source applied to croplands, which can also serve as a source of organic matter (Lorimor, Powers, and Sutton 2004). Manure nutrient, organic matter, and water content can vary greatly depending on the stage of animal growth, feeding practices, amount of bedding or water added to the manure, type of manure storage, and time that manure spends in storage. This variability makes manure more difficult to manage than mineral fertilizers (Lorimor, Powers, and Sutton 2004). Manure often contains nutrients in organic forms, which are less soluble than those found in mineral fertilizers. When applied to the soil, organic nutrients mineralize over time, releasing nutrients that may be susceptible to loss through runoff or leaching.

The N to P ratio in manures does not typically match the ratio of N to P required by plants; thus, meeting crop N demands with manure may result in applying three to five times more P than the crop needs. In many livestock production areas, nutrient imbalances occur as a result and lead to buildup of nutrients, especially P, in excess of crop needs. These legacy nutrients present in soils can result in substantial and persistent nutrient losses to groundwater and surface water. Manure testing to determine nutrient content is a critical component of manure nutrient management. Nutrient losses from croplands with applied manure are not necessarily greater than nutrient losses from croplands receiving mineral fertilizers (Lory, Massey, and Joern 2008). When manure and fertilizer are applied at the same application rate, annual nutrient losses are similar in magnitude (King et al. 2018).

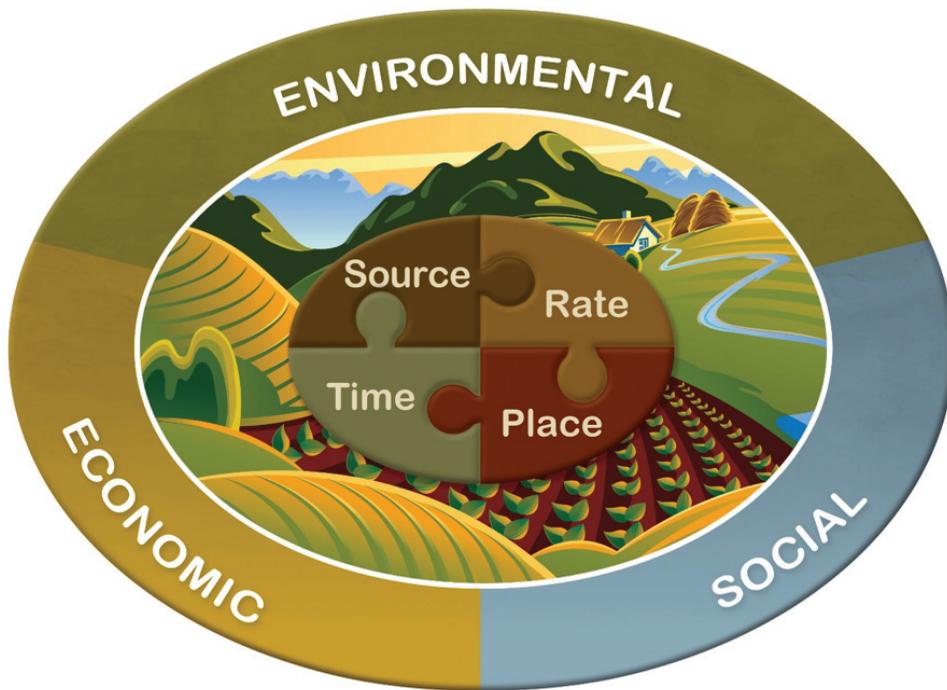


Figure 7. The 4R Approach to Nutrient Stewardship concept defines the right source, rate, time, and place for fertilizer application as those producing the economic, social, and environmental outcomes desired by all stakeholders to the plant ecosystem. (Image courtesy of the International Plant Nutrition Institute.)

Nutrient Application Rate

Nutrient application rates are determined differently for P and N. The right application rate for P is often based on soil sample collection and testing. Soil testing provides an index of nutrient availability or supply and not the actual quantity of nutrient present in the soil. Soil testing aims to predict the probability of profitable response to nutrient inputs and provide guidance on the amount of fertilizer needed to maximize economic return. It can also be used to confirm or diagnose a nutrient deficiency, to optimize plant health, and to potentially identify areas that may be at risk for nutrient loss. Soil test P concentrations can be determined for an entire field or different areas within a field if zone or grid soil sampling is performed. Soil test recommendations for P generally follow a “sufficiency” approach, recommending only enough nutrient to maximize crop yield in the current year, or a “build-and-maintain” approach, in which soil nutrient concentration is built up to an optimal range and additional applications of nutri-

ents are used to balance crop removal, or a hybrid of sufficiency and build and maintain. Build-and-maintain systems require less frequent soil testing (perhaps every two to three years) than sufficiency systems that would require annual soil testing and fertilizer application. Current P application rate recommendations are generally accurate across many years and fields, but they lack precision because of the spatial and temporal variability in soil chemistry, soil microbial communities, and plant uptake, which could result in either overapplication or underapplication of P.

Methods for determining the right application rate for N are varied and may be yield-based, use a maximum return to N (MRTN) approach, integrate crop reflectance sensors, or apply model-based N management. Yield-based N application rate recommendations use crop type, crop yield potential, N source, and, in some instances, soil texture and tillage methods to determine the right rate. In 2005, many upper midwestern states shifted from yield-based N recommendations for

corn to the MRTN approach (Sawyer et al. 2006), which uses a large university database of N rate response trials. With this approach, the MRTN response is calculated using the price of grain and the yield response production function, which can then be grouped by previous crop, region, or soil texture to generate optimum N rates. With a crop sensor-based recommendation, reflected light from the crop canopy is used to calculate vegetative indices such as the normalized differential vegetative index. Numerous algorithms exist to convert sensor readings into N application rates. Finally, model-based N management leverages the ability for modern computers to use large data sets, including soil, climate, and yield, to run complex simulations of N supply and uptake to provide N application rate recommendations.

Nutrient Placement

Nutrient placement can have significant implications for both crop uptake and nutrient loss. For example, P is only able to move short distances within the soil profile; therefore, P placement in or near the seed row where it can be accessed by roots can result in a positive crop yield response (IPNI 2012). Nutrients broadcasted on the soil surface pose a larger risk of nutrient loss compared to nutrients that are either incorporated into the soil with tillage or placed in the subsurface using banding or injection (Gascho et al. 1998; Williams, King, Ford, et al. 2016; Williams et al. 2018). Surface broadcast applications may also result in nutrient stratification, especially for P, where soil nutrient concentrations become elevated at or near the soil surface (0–5 cm) and result in an increased risk of loss (Baker et al. 2017). In addition to reducing runoff and leachate losses, incorporating or injecting manure can also help decrease N loss via volatilization (Duncan et al. 2017b).

The right placement also includes acknowledging spatial variability in soil nutrient concentrations and yield potential within a field. Soil nutrient concentrations and yield potential can vary across a field because of several factors, including soil texture, soil pH, past management activities, and topography. Variable rate application technology, which varies the

nutrient application rate according to the location within the field using geographic information systems and global positioning systems, can not only help improve nutrient use efficiency, but also decrease the potential for nutrient losses (Harmel et al. 2004).

Nutrient Timing

The right timing of nutrient application aims to ensure there is adequate nutrient supply during peak crop uptake and critical crop growth stages. Synchronizing nutrient applications with crop demand is not always easy because of several factors including crop type, environmental conditions (e.g., precipitation, soil moisture), and the time available for farming operations. Phosphorus is often recommended to be applied at planting to ensure the P is available to early developing plants. Nitrogen can also be applied at planting or in split applications, whereby a portion of the N is applied at planting and the remainder is applied later in the growing season to better match crop needs. Timing of nutrient applications can be further managed with slow- and controlled-release fertilizer technology, stabilizers, and inhibitors that can slow the transformation of applied nutrients within the soil.



Figure 8. Cover crops can be planted to minimize the presence of bare soil surfaces. (Photo courtesy of Heidi Peterson.)

Timing of nutrient application should also consider soil conditions and weather patterns to decrease the risk of nutrient loss. Nutrient application to frozen or snow-covered soils should generally be avoided because frozen soils can limit infiltration of meltwater or rainfall into the soil and lead to large runoff events (Srinivasan et al. 2006; Williams et al. 2011). Similarly, nutrient applications to saturated soils or immediately prior to large rainfall or irrigation events should be avoided to minimize the risk of nutrient loss. Events immediately following nutrient application can lead to significant nutrient loss (Smith et al. 2016) that, in some cases, can constitute a large proportion of the annual nutrient load (King et al. 2018).

IN-FIELD, EDGE-OF-FIELD, AND IN-STREAM CONSERVATION PRACTICES TO DECREASE THE IMPACT OF AGRICULTURE ON WATER QUALITY

Conservation practices, also referred to as best management practices (BMPs), can be used in combination with nutrient management to decrease nutrient loss from cropped fields (Strock et al. 2010). This section will provide an overview of some of the common conservation practices used on croplands across the United States. The nutrient reduction potential for many conservation practices is often site specific due to differences in soils, climate, hydrology, and management systems; thus, rather than focus on the range of nutrient removal potential, the authors highlight the primary mechanisms for nutrient removal for each practice and identify potential trade-offs.

Vegetated Filter Strips and Grassed Waterways

Vegetated filter strips, buffers, or riparian zones are often implemented between the edge of an agricultural field and a stream or drainage ditch. Similarly, grassed waterways can be established on sloping areas, with the primary purpose of stabilizing soil from erosion in natural

drainage ways. When positioned on the landscape in an area receiving surface runoff from agricultural fields, vegetated filter strips and grassed waterways can trap sediment and particulate nutrients (i.e., nutrients bound to the sediment) (Shipitalo et al. 2010). Vegetated filter strips can also intercept shallow groundwater flowing laterally toward the stream or ditch and decrease NO_3^- concentrations (Vidon, Welsh, and Hassanzadeh 2018). These practices tend to be the most efficient at removing nutrients when surface flow is uniformly distributed and shallow, with efficiency substantially decreased when surface flow becomes concentrated or subsurface flow is short circuited through a subsurface drainage system. When not properly maintained, they can also accumulate sediment and nutrients and, as a result, become a nutrient source over time.

Cover Crops

Integrating single or multispecies cover crops with the primary commodity crop system will decrease the amount of time that fields are left with bare soil (Figure 8). Cover crops, therefore, have the potential to decrease soil erosion and particulate nutrient loss in surface runoff, although their effectiveness is reduced when germination and stand cover are limited by lack of fall precipitation, poor seed-to-soil contact, or short growing seasons (Strock, Porter, and Russell 2004). Winter cover crops can also be effective at decreasing nitrate leaching losses in agricultural landscapes (Kaspar et al. 2007). As soils thaw after the winter freeze, however, decomposing cover crop residue can serve as a source of dissolved P that can be transported in runoff or leachate with snowmelt or spring precipitation events (Cober, Macrae, and Van Eerd 2018) (Figure 9).

Soil Residue Management

Conservation tillage leaves >30% of the crop residue on the soil surface through next season's planting, which results in a decrease in soil disturbance, erosion, and particulate nutrient losses. No-till, mulch-till, ridge-till, and strip-till are the most common conservation tillage methods. No-till leaves the soil undis-

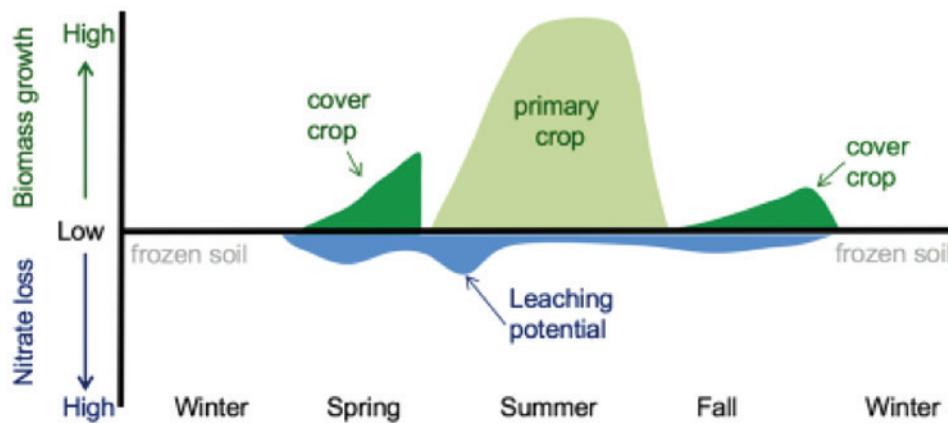


Figure 9. Proper implementation of a cover crop can decrease the opportunity for nutrient runoff and leaching by increasing plant biomass to cover the soil and use the available soil nutrients. (Image from Noland, Wells, and Peterson [2018]; adapted from Heggenstaller et al. [2008].)

turbed from harvest to planting. Ridge-till involves planting into a seedbed prepared on ridges. Mulch-till does not invert the soil, but rather keeps it rough with the use of a chisel plow, disks, or blades. Strip-till uses minimum tillage by disturbing only the seed row soil. The additional surface residue left with conservation tillage provides a protective soil cover to reduce raindrop energy upon contact with the soil surface, which can also increase infiltration into the soil. A consequence of increased infiltration and reduced soil disturbance is the formation of preferential flow paths in fine-textured soils, which can increase dissolved and particulate nutrient losses, especially P, in leachate to subsurface tile drains and groundwater (Williams et al. 2018). Additionally, conservation tillage practices may enhance P stratification in surface soil layers (0–20 cm), which could increase the risk for nutrient loss in surface runoff and leachate (Baker et al. 2017). If a soil is highly stratified, however, tillage can mix surface and subsoils to decrease the risk of nutrient loss (Sharpley 2003). Unfortunately, conservation tillage has little effect on mitigating nitrate losses to surface waters from agricultural fields (Daryanto, Wang, and Jacinthe 2017; Kanwar, Colvin, and Karlen 1997).

Sediment Detention Basins and Blind Inlets

Sediment detention basins capture ag-

ricultural surface and subsurface drainage water and allow sediment and particulate nutrients to settle out prior to the water entering a stream or ditch. Similarly, blind inlets, which are typically installed in closed depressions or potholes and replace either an open inlet or tile riser (Smith and Livingston 2013), filter out sediments and particulate nutrients in surface runoff prior to water entering the subsurface tile drainage system. The effectiveness of sediment detention basins and blind inlets is impacted by the residence time and the concentration and form of nutrient. As sediments and associated nutrients accumulate in sediment detention basins and blind inlets over time, they may become potential sources of dissolved P.

Constructed Wetlands

Constructed wetlands have the potential to remove nutrients from agricultural drainage water. Saturated soils in wetlands can facilitate the removal of NO_3^- through denitrification, whereas wetlands primarily decrease P loads through retention of sediments (Kalcic et al. 2018). The effectiveness of nutrient load reduction in constructed wetlands is influenced by several factors, including the timing and magnitude of nutrient loads, residence time within the wetland, nutrient concentrations, and the form of nutrients entering the wetland. Nutrient reduction is often greatest when residence



Figure 10. Drainage water management, also known as controlled drainage, uses a water control structure such as the one being installed at this site in Sac County, Iowa, to raise the depth of the drainage outlet, holding water in the field during periods when drainage is not needed. (Photo courtesy of Lynn Betts, National Resources Conservation Service [NRCS]/Soil and Water Conservation Society [SWCS].)

times are long and hydraulic loading rates are low. The capacity for constructed wetlands to remove P is finite. In many instances—including wetlands immediately following construction, wetlands that are periodically dry and then are reflooded, and permanently saturated wetlands—constructed wetlands may result in increased losses of P due to the complex interaction of chemical processes (Reddy et al. 1999).

Drainage Water Management and Saturated Riparian Buffers

For fields with subsurface tile drainage, drainage water management or controlled drainage can be used to



Figure 11. Saturated buffers divert tile drainage water into a shallow lateral pipeline distributed parallel to a water body, as shown in this installation in Story County, Iowa, to raise the water table within the buffer and enhance nutrient uptake by plants while slowing subsurface water outflow into the adjacent stream. (Photo courtesy of Lynn Betts, NRCS/SWCS.)

artificially adjust the outlet elevation of the drainage network to a specified depth by restricting flow (Skaggs, Fausey, and Evans 2012) (Figure 10). Drainage water management promotes moisture storage during periods when drainage is not necessary. The outlet elevation can be set at any level between the ground surface and the drainage depth through installation of a control structure, which is typically comprised of stackable boards or stop logs. By decreasing the flow volume from the tile drain, dissolved and particulate nutrient loads are also decreased (Ross et al. 2016). The altered field hydrology that occurs from drainage water management could result in increased nutrient losses in surface runoff and create saturated soil conditions. Saturated riparian buffers also use a control structure and an additional tile drain installed parallel to the stream or ditch (Figure 11). Water leaving the field from the tile drain is routed through the riparian zone before it enters the stream or ditch. Like the vegetative filter strip, this practice has shown promise for

reducing NO_3^- concentrations (Jaynes and Isenhardt 2018), but there has been little research investigating P losses.

Bioreactors and Phosphorus Removal Structures

Both bioreactors and P removal structures have been implemented using various designs and can be installed separately or in series. Most bioreactors consist of a solid carbon substrate (often fragmented wood products) that is placed in the flow path of nutrient-laden water (Christianson, Bhandari, and Helmers 2009). Nitrate in agricultural drainage water is denitrified through microbial processes as it passes through the carbon media. There have been several design variations of denitrifying bioreactors, including in-field denitrification walls (Jaynes et al. 2008), edge-of-field bioreactors (Woli et al. 2010), and stream bed bioreactors (Robertson and Merkley 2009). Most P removal structures are constructed using a media containing high concentrations

of calcium, aluminum, or iron (Penn et al. 2017). Dissolved P in agricultural drainage water that flows through the media is chemically removed. The effectiveness of these practices is controlled by the media used, hydraulic loading, influent nutrient concentration, and residence time of the water. Bioreactors and P removal structures are often susceptible to bypass flow, whereby the design capacity is exceeded during large events and, as a result, nutrient removal potential is decreased.

Two-stage Ditches

Two-stage ditch systems incorporate benches that function as flood plains in an attempt to restore or create natural alluvial channel processes. The two-stage ditch is designed to provide improved physical stability of ditch banks (Powell et al. 2007). When water level in the ditch rises during a flow event, water can flood the benches and potentially trap sediment and particulate nutrients. Flood plain inundation may also increase NO_3^- removal through denitrification (Davis

et al. 2015). Nutrient removal benefits of two-stage ditches largely depend on the frequency and length of flood plain inundation, the soil properties of the constructed flood plain, and the vegetation, which influence the ability of the benches to trap sediment and their biological activity.

HISTORICAL AND CURRENT ENVIRONMENTAL POLICY, INCENTIVES, PROGRAMS, AND INITIATIVES

The public invests heavily in conservation programs to mitigate the impacts of nutrient loss on water quality from agricultural land, both through Farm Bill programs administered by the USDA and through the Clean Water Act (CWA) and other programs administered by the U.S. Environmental Protection Agency (EPA) and the states. This section reviews these programs and the approaches used.

U.S. Department of Agriculture Programs

The USDA has assisted farmers in improving soil and water conservation since the 1930s, and this has resulted in widespread implementation of conservation practices (USDA–NRCS 2016). Early programs focused primarily on soil conservation to maintain or improve soil productivity. Recently, water quality and protecting downstream resources have become a primary objective of USDA programs. United States Department of Agriculture conservation programs use combinations of technical assistance, education, and financial assistance to address conservation needs across U.S. croplands. Financial assistance is primarily provided through voluntary financial incentives that pay a portion of the cost of practice implementation, usually 50 to 75% of the costs, often referred to as cost sharing. The idea is that farmers pay part of the cost because they receive part of the benefit, typically conservation of productive soil that can result in long-term benefits to landowners. These methods have resulted in significant progress in controlling soil erosion and loss of those nutrients that are associated

with sediment; however, this strategy is less effective in addressing losses of dissolved nutrients because the on-farm benefits are far less than the cost or effort required to reduce their loss. Although USDA programs are voluntary, conservation compliance, which was enacted as part of the 1985 Farm Act, made eligibility for federal financial assistance subject to compliance with the application of a soil conservation plan on highly erodible land and wetland protection for land that had not yet been converted to cropland. There currently are no similar compliance requirements that apply to decreasing nutrient losses.

Removing vulnerable and low-productivity cropland from agricultural production is one tool for improving soil and water conservation. Land retirement is therefore an important component of USDA conservation programs, starting with the Agricultural Adjustment Act (1936), continuing with the Soil Bank (1956), and since 1985 through the Conservation Reserve Program (CRP) (Claassen, Cattaneo, and Johansson 2008). The CRP is the largest USDA conservation program, funded at \$1.8 billion in 2018 (Stubbs 2018). Currently 9.7 million ha (24 million acres) are enrolled in the program, down from peak enrollment of approximately 15 million ha (37 million acres) in 2007, when the program acreage cap was set much higher in the 2002 Farm Bill. The CRP addresses water quality along with other enrollment goals, including wildlife habitat, soil productivity, flood damage reduction, and carbon sequestration. In 2010, it was estimated to have reduced the loss of about 275 million kg and 55 million kg of N and P, respectively (USDA–FSA 2011). Johnson and colleagues (2016) conducted a CRP case study in Iowa to quantify the environmental benefits of CRP lands, and results suggested that the investments were justified based upon the value of public and private provided.

Working lands programs, which fund conservation on land that remains in crop or livestock production, have also been part of the USDA conservation programs since 1936 with the Agricultural Conservation Program. In 1996, the Environmental Quality Incentives Program (EQIP) succeeded the Agricul-

tural Conservation Program and several smaller programs, aiding eligible farmers to address soil, water, and related natural resource concerns on their lands. In 2018, EQIP was funded at \$1.6 billion (Stubbs 2018). The Conservation Stewardship Program (originally Conservation Security Program) provides payments for achieving certain levels of environmental service provision (Claassen, Cattaneo, and Johansson 2008) and is the third major USDA conservation program, with 2018 funding of \$1.3 billion (Stubbs 2018). The effect of these programs is difficult to measure because many farmers have installed practices without assistance, efficiency will vary depending on landscape position and practice maintenance, and water quality improvements are not an immediate response.

U.S. Environmental Protection Agency and State Programs

The Clean Water Act (passed in 1972 as the Federal Water Pollution Control Act) made point source pollution control a federal responsibility, but it allocated control of nonpoint source (NPS) pollution to the states. With some exceptions, the states have opted for voluntary compliance strategies for agricultural NPS control. Most agricultural entities are NPSs, although large concentrated animal feeding operations that discharge directly to surface waters through a pipe or ditch are treated as point sources and must obtain permits. Although the requirements vary by state, manure management plans are commonly required for livestock operations with a capacity of more than 500 animal units to minimize manure or other wastewater runoff from fields to surface waters or groundwater.

In addition to permits for point sources, the CWA requires states to establish water quality standards that include designated uses and criteria to protect those uses. Narrative nutrient criteria exist in most states, although not the numeric criteria required by the CWA. The CWA also requires the development of total maximum daily loads (TMDLs) for water bodies that do not meet water quality standards due to nutrients (or other pollutants). A TMDL for nutrients sets limits on the total discharge from all sources,

including NPSs. Although the TMDL allows states to rely on voluntary approaches to decrease nonpoint discharges, it could potentially lead to implementation of more regulatory approaches. For example, at least 11 states have implemented a law banning P in lawn fertilizers in order to limit P discharge to urban surface waters. The EPA has responsibility for developing TMDLs if a state fails to act (USEPA 1993). Section 319 of the CWA established the EPA's Nonpoint Source Program in 1987, granting states funds to develop and promote NPS management plans and other programs, in which the EPA provides program guidance and technical support. The EPA has encouraged states to work together to address regional water quality problems (e.g., The Mississippi River/Gulf of Mexico Hypoxia Task Force).

Other federal laws may also affect agricultural NPS pollution. The Coastal Zone Management Act Reauthorization Amendments (CZARA) is a federally mandated program that does require specific measures to deal with agricultural NPSs. The CZARA requires each state with an approved coastal zone management program to submit a program to implement management measures for NPS pollution to restore and protect coastal waters. The Safe Drinking Water Act requires the EPA to set standards for drinking water quality and requirements for water treatment by public water systems (Morandi 1989). Source water assessment also requires states to determine the susceptibility of source waters to adverse impacts prior to intake (Price and Heberling 2018).

WATER QUALITY CHALLENGES FACING U.S. AGRICULTURE IN A CHANGING LANDSCAPE

The combined demands of increased agricultural production with reduced environmental impact require management strategies that can be sustained over the long term. These strategies must recognize the diversity in agriculture that results from differences in climate, physiography, ecology, economics, and culture (Zhang et al. 2007). This diversity

confounds uniform approaches to achieving sustainable production systems. The previous sections presented current knowledge on nutrient cycling and transport together with nutrient management and structural conservation practices that land managers might use to reduce agriculture's impact on water quality. These sections highlighted the additional progress needed and the complexities associated with managing NPS pollution. This final section presents several current and future challenges related to agriculture intensification and water quality.

Current Knowledge of N and P Rates Is Imprecise

Providing crops with an adequate supply of nutrients during critical growth stages is essential for increasing productivity and profitability, but management decisions need to factor in the potential for environmental loss. Although the methods and approaches used to provide nutrient recommendations differ for N and P, current strategies for both N and P are generally imprecise and may result in over- or underapplication of nutrients. Even when farmers and crop advisors follow the best available nutrient recommendations, the applied nutrients may not be taken up by the crops that same season, resulting in less than 100% recovery. Crop nutrient needs can vary within and among fields as a function of soil physical, chemical, and biological properties; previous management activities (especially for P); and year-to-year differences in weather (especially for N). In addition, advancements in crop genetics and the large variety of crop hybrids further complicate nutrient recommendations, because differences in crop genetics may alter plant nutrient requirements and the efficiency of the plant at taking up nutrients from the soil. These complicating variables often favor risk-adverse strategies, which may unintentionally result in the overapplication of nutrients to prevent yield loss, especially for high-value crops.

Nutrients applied inefficiently or in excess of crop demand are susceptible to loss; thus, there is a need to develop more precise nutrient recommendations that account for differences in soil, plant,

and weather conditions. Smith and colleagues (2018) reviewed survey results of row crop farmers in the Maumee River watershed, located in the Lake Erie Basin. They indicated that many farmers use science-based nutrient management recommendations with regard to soil testing as a guide for P fertilizer applications. Phosphorus applications either met or were below fertilizer recommendations in more than 90% of the surveyed fields, yet the anticipated benefits had yet to be realized (Smith et al. 2018). Although it is difficult for nutrient rate recommendations to be perfectly precise because of the imprecision within an agricultural system that complicates the ability to obtain representative soil samples and the complex variations at the time of nutrient application, the development of new technology is needed to better inform and guide application rates in conjunction with research aimed at better understanding the impacts of nutrient placement and timing.

Legacy Nutrients May Mask Water Quality Impacts of Current Conservation Efforts

Despite widespread adoption of nutrient management and structural conservation practices across the United States, measurable reductions in nutrient load at watershed outlets are rarely apparent (Sharpley et al. 2013). Failure to decrease nutrient loads at the watershed scale may not imply that current conservation efforts are having no effect because legacy nutrients, which have built up in soils and groundwater over time, may mask or buffer the efforts of current conservation. Lag times between conservation implementation and observed nutrient reductions can range from years to decades, depending on the amount of legacy nutrients stored throughout the watershed (Gilmore et al. 2016; Meals, Dressing, and Davenport 2010). The presence of rills, gullies, soil macropores, drainage ditches, and tile lines all serve as potential conduits for mobilizing nutrients, saturating the soil's sorption capacity, and releasing this stored nutrient as a source (Sharpley et al. 2013). Additionally, it can take several years for most conservation practices to achieve maximum reduction

potential (Daniels et al. 2018). Currently, it is not possible to differentiate between recently applied nutrients and legacy nutrients, which makes it difficult to assess the effect of conservation on water quality at large watershed scales. Land modifications that increase erosion or hydrologic connectivity can rapidly increase the transfer of legacy P (Sharpley et al. 2013). These factors may lead to an incorrect conclusion that BMPs are not working. New methods or techniques for separating these sources of nutrient loss are needed to further evaluate the impact of current conservation. Differentiation of sources would also facilitate improved conservation practice design and implementation by providing better information on how and where nutrients are being transported through the landscape.

Most Implemented Conservation Practices Do Not Address Dissolved Nutrients

Many current conservation practices—including vegetated filter strips,

grassed waterways, cover crops, tillage practices, sediment detention basins, and blind inlets—are designed to primarily address sediment and particulate nutrient loss. Regarding dissolved P loss, research results for these practices are often inconclusive or suggest that they only result in marginal reductions or, in some cases, increases. Dissolved sources of nutrients represent a substantial proportion of annual nutrient loss that cannot be directly managed or treated in the short term with soil conservation practices. Currently available conservation practices that address dissolved nutrient loss typically have little or no economic benefits for producers. One practice that addresses multiple benefits, along with reduction of dissolved nutrients where tile drainage is prevalent and the crop season may have both excessively wet and dry periods, is drainage water recycling (Figure 12). Excess water is captured through the tile drains, stored in a pond or reservoir, and then used to irrigate crops during a water deficit. Additional research is needed to address site-specific design and imple-

mentation requirements of drainage water recycling and other innovative conservation practices to economically address dissolved nutrients.

Few Conservation Practices Provide In-stream Nutrient Removal

The majority of conservation practices is designed for in-field or edge-of-field application, with few practices available to address nutrient loss within streams or ditches. Focusing on in-field and edge-of-field practices will likely result in unmet nutrient load reduction goals at the watershed scale due to the importance of groundwater, streambank erosion, and in-stream nutrient cycling on watershed nutrient loading (Williams, King, and Penn 2018). Implementation of in-stream conservation is further complicated by the fact that most agricultural ditches are managed by drainage boards or counties rather than individuals. There is a need to develop and implement conservation practices that address both dissolved nutrient loss and in-stream nutrient removal to help mitigate the downstream impacts of agriculture on water quality.

Nutrient Reductions for Both Nutrient Management Practices and Conservation Practices Are Field Specific

The nutrient reduction potential for many nutrient management and structural conservation practices is often site specific. For example, a grassed waterway or buffer strip may result in a greater nutrient loss reduction on one field where runoff is more concentrated, with less reduction effectiveness from a second similar field having less slope and more diffuse runoff. This creates a large challenge for both conservation programs that are providing recommendations on practices and farmers trying to decide which practice will fit within their management system. Throughout the United States, there is an urgency to solve water quality issues, which often leads to a tendency toward applied research that focuses on quantifying practice effectiveness outcomes. Recognizing that some level of nutrient loss is unavoidable from natural

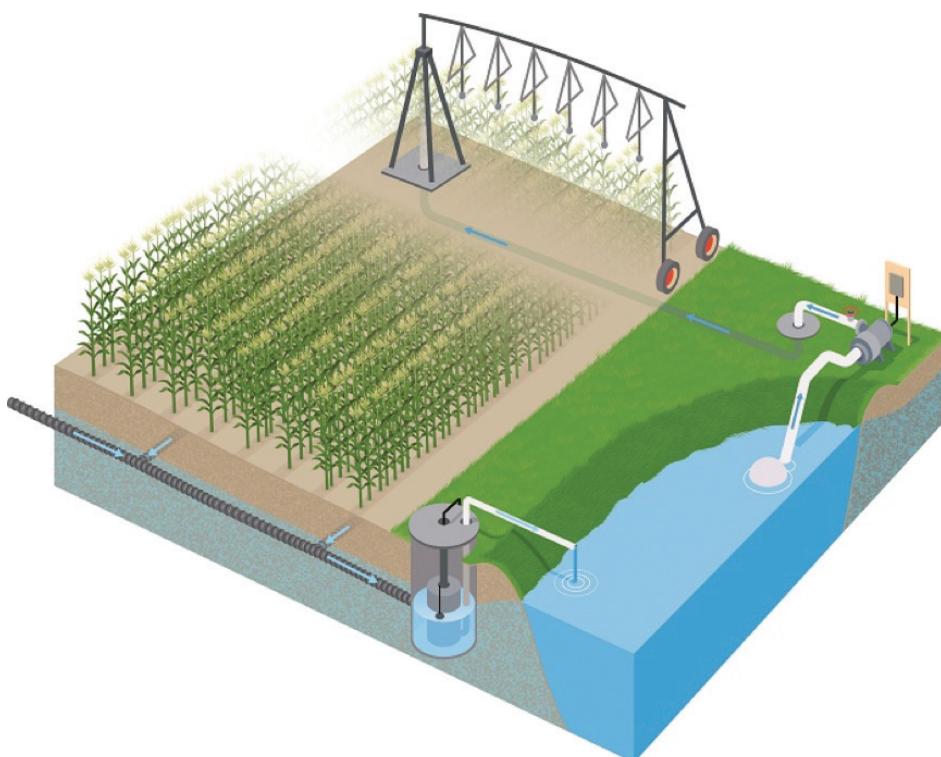


Figure 12. Drainage water recycling is an innovative technology that diverts surface and subsurface drainage water into reservoirs where it is stored for crop reuse during dry months. (Image courtesy of transformingdrainage.org.)

and anthropogenic sources, however, one question that must be answered is what to consider an acceptable level of nutrients in runoff from agricultural fields. A review of watershed analyses where edge-of-field water quality measurements were collected indicated that <5% of the applied N and P tends to be lost in runoff (Daniels et al. 2018; Galloway and Nustad 2017; Smith et al. 1983; Tomer et al. 2016), yet this small quantity can still result in environmentally significant impacts. Even natural, undisturbed landscapes, such as native grasslands, contribute average loads of 2 kg/ha total N and 0.5 kg/ha total P annually to surface water bodies (Harmel et al. 2006, 2008). Due to the complexity of nutrient cycles and transport processes that control how practices function (e.g., the role of soils, climate, hydrology, and management systems), multiple approaches such as certainty programs, field-scale and watershed models, and edge-of-field monitoring must be considered for assessing the impact on downstream water quality (Harmel et al. 2018). Meeting water quality nutrient reduction goals requires both basic and applied research programs to understand the complex processes controlling nutrient loss in agricultural landscapes and develop new, innovative practices.

Conservation Program Success Requires Collaboration and Cost-effective Implementation

Current conservation programs are voluntary and rely on farmers to approach the agency to enroll and choose which resource issues to address. A potential weakness of this approach is that conservation improvements tend to be supply driven rather than demand driven (Shortle et al. 2012). That is, farmers propose contracting for conservation practices on a first-come, first-served basis, centered on their farm's needs and management schedule. Many conservation decisions in this voluntary setting are driven by the bottom line, and the environmental impacts felt downstream, off the farm, are not necessarily considered. This may result in the implementation of practices

in areas that are less environmentally sensitive with minimum impact on downstream water quality compared to other potential sites within the same or adjacent watershed.

Several conservation program features, including the targeting of critical source areas or pay-for-performance, could increase the investment in conservation measures that address off-farm issues and improve cost-effectiveness. Conservation programs can be more cost effective when specific fields or watersheds contributing a disproportionate amount of nutrients are targeted for conservation (Galzki, Birr, and Mulla 2011). These targeted areas and specialized conservation initiatives, however, should be selected using a collaborative approach that considers local and regional input and data. Effective conservation programs reward local conservationists who build collaborative working relationships with landowners and managers who have the most critical environmental concerns (Nowak 2011). Targeting conservation is difficult when programs follow a uniform and standardized pathway that requires a consistent set of requirements rather than directly engaging the land user in the resource issue (Nowak 2011). Farmers who can provide the most cost-effective control may not enroll in programs or may want to address other issues on their farms that more directly affect their net returns. In addition to federal conservation programs, there are other considerations that farmers must include in their management decisions, such as crop insurance, lender agreements, and landowner expectations, and some of these may be at odds with conservation initiative stipulations. Farmers who are ready to implement a practice typically apply when it fits within the complexity of their farm's management schedule, which is not always consistent with a strictly defined program's protocol or technical engineering staff availability.

Paying for performance based on the amount of nutrient loss reduction is also more cost effective than basing financial assistance payments on practice costs (Savage and Ribaudo 2016), and it does not require targeting of practices. Under this approach, farmers who can provide the most abatement at the lowest costs

have the greatest economic incentive to act. Practice-based payments tend to limit choice to practices that are cost shared, whereas performance-based policies award innovations that lower costs. Regardless of the approach, the success of any voluntary conservation program relies on the availability of willing participants.

Several state programs also use regulation or involuntary economic incentives to address water quality issues when voluntary approaches fail to make desired progress. This includes performance taxes that create incentive to adopt conservation practices and peer pressure from farmers to their recalcitrant neighbors; development of watershed designations, such as a watershed in distress, that triggers the implementation of regulation (Jacquemin et al. 2018); and monitoring-based "trigger" policies that place more stringent controls on nutrient management as concentrations in groundwater and surface water increase (Ribaudo and Caswell 1999). Implementing a TMDL on the Chesapeake Bay watershed by the EPA resulted in more resources devoted to the problem and initiated new regulatory and nonregulatory programs (Ribaudo, Savage, and Aillery 2014). Increasing program cost-effectiveness, through multiple approaches that engage industry organizations and companies and consider incentives for downstream ecological enhancement, is needed to more effectively and efficiently decrease nutrient loss from agriculture.

CONCLUSIONS

In conclusion, agriculture today reflects the outcome of historical shifts in management in which ambitious goals meet with challenges of production potential, profitability, cultural norms, and environmental resources (Kleinman et al. 2018). Complexities and difficulties associated with nutrient cycling and transport processes, management decisions and practice trade-offs, and program effectiveness highlight both significant progress and immense challenges to achieving sustainable intensification while meeting water quality goals. To overcome these challenges, it is imperative that agriculture balances short-term management

decisions (e.g., nutrient source, rate, placement, and timing) with long-term planning (e.g., processes controlling nutrient loss, conservation practice implementation) such that crop production and profitability enhance rather than compete with environmental objectives. Achieving water quality goals will require continued research to better understand the complex processes controlling crop nutrient requirements and nutrient losses, creation of new conservation practices and technologies to limit losses, and development of sustainable conservation programs that engage regional stakeholders to prioritize water quality objectives and integrate the complex attitudes and constraints associated with landowner conservation adoption.

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