
*Agricultural Management, Water Quality and Ecology: Putting Practice into Policy*¹

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Since the late 1960s, point sources of water-quality impairment have been reduced due to their ease of identification and the passage of the Clean Water Act in 1972. However, water-quality problems remain, and as further point-source control becomes less cost-effective, attention is being directed towards the role of agricultural nonpoint sources in water-quality degradation. In a 1996 US Environmental Protection Agency (EPA) report, nonpoint-source nutrients were the primary source of concern in 40% of rivers, 50% of lakes, and 60% of estuaries surveyed and listed as impaired (US Environmental Protection Agency, 1996). Some 15 years on, continuing water-quality impairment has led to major initiatives to reduce losses from the Chesapeake Bay Watershed (Kovzelove *et al.*, 2010; US Environmental Protection Agency, 2010a) and the Mississippi River Basin (National Research Council, 2008). As in the United States, the European Union Water Framework Directive (WFD) (Council of European Communities, 2000) requires widespread control of nitrogen (N) and phosphorus (P) inputs to rivers specifically to improve riverine ecology (Hilton *et al.*, 2006).

Phosphorus inputs to fresh waters can accelerate eutrophication (Carpenter *et al.*, 1998). Although nitrogen and carbon (C) are also essential to the growth of aquatic biota, most attention has focused on P because of the difficulty in controlling the exchange of N and C between the atmosphere and water, and fixation of atmospheric N by some blue-green algae (Schindler *et al.*, 2008). Thus, control of P inputs is critical to reducing freshwater eutrophication. For water bodies with naturally higher salt content, as in estuaries, there are likely unique site-specific critical concentrations of both N and P that generally limit aquatic productivity (Howarth *et al.*, 2000).

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Eutrophication has been identified as the main problem in surface waters having impaired water quality (US Environmental Protection Agency, 2002). Eutrophication restricts water use for fisheries, recreation, industry, and drinking, due to the increased growth of undesirable algae and aquatic weeds and oxygen shortages caused by their death and decomposition (Table 1). Also, many drinking-water supplies throughout the world exhibit periodic massive surface blooms of cyanobacteria. These blooms contribute to a wide range of water-related problems including summer fish kills, unpalatability of drinking water, and formation of trihalomethane during water chlorination. Outbreaks of the dinoflagellate *Pfiesteria piscicida* in the eastern United States were linked to excess nutrients in affected waters (Burkholder and Glasgow, 1997).

Nitrate is a water-quality concern because it has been linked to methemoglobinemia in infants, to toxicities in animals, and to increased eutrophication in both fresh and saline (*e.g.*, estuaries) waters. In response, EPA has established a maximum contaminant level for nitrate-N in drinking water of 10 mg L⁻¹ (45 mg nitrate L⁻¹). Under anaerobic soil conditions, denitrifying bacteria readily convert excess nitrate to N gases (primarily N₂), reducing the quantity of nitrate that can potentially leach to groundwater supplies. However, the production of N_xO gases contributes to the accumulation of greenhouse gases in the atmosphere (Figure 1).

**TABLE 1. ADVERSE IMPACTS OF EUTROPHICATION ON FRESHWATER
LAKES, RIVERS, AND STREAMS.**

- Increased phytoplankton biomass
 - Shifts in phytoplankton to bloom-forming species that may be toxic or inedible
 - Increased biomass of benthic and epiphytic algae
 - Changes in macrophyte species composition and Biomass
 - Decreases in water transparency
 - Taste, odor, and water-treatment problems
 - Oxygen depletion
 - Increased incidence of fish kills
 - Loss of desirable fish species
 - Reductions in harvestable fish and shellfish
 - Decreases in aesthetic value of water body
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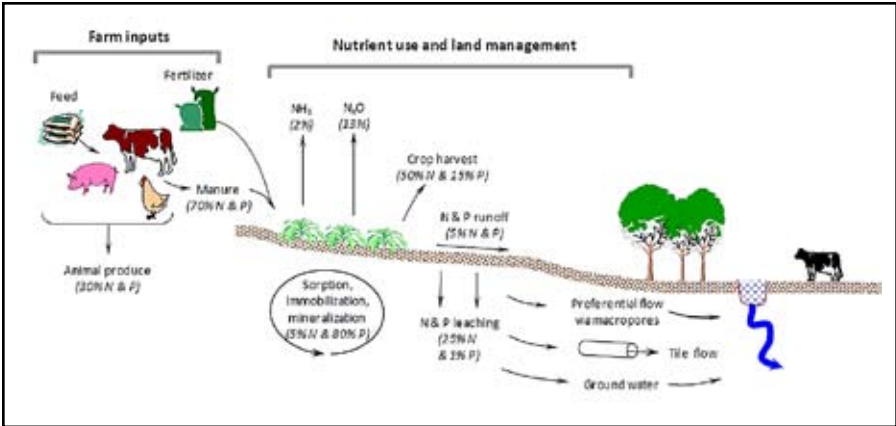


Figure 1. Factors affecting the fate of N and P in agriculture, representing their potential impact on soil, water, and air quality. Numbers in parentheses are based on an approximate farm nutrient balance and relative fate of N and P as a percentage of load (“Farm inputs”) or percentage of fertilizer and manure (“Nutrient use and land management”) (adapted from Duxbury *et al.*, 1993; Bouwman and Booi, 1998; Howarth *et al.*, 2000; Sims and Sharpley, 2005).

In the last 20 years, there has been a “U-turn” in strategic planning for nutrient management and water-quality impacts. It is now cheaper to treat the cause of water-quality impairment rather than its effects. In the early 1990s for example, New York City decided it would be more cost-effective to identify and target for remediation, the sources of P in its water-supply watersheds, rather than build a new \$8-billion water-treatment facility. As a result, the state invested \$10 million to identify and reduce nutrient sources in its supply watersheds. A similar targeted management strategy is now in place in most impaired waters of the United States, as, for example, in the Chesapeake Bay Watershed, Florida inland and coastal waters, Lake Erie Basin, and Mississippi River Basin.

Phosphorus and N are typically treated separately by scientists and environmental managers. The theoretical parsing of these elements may be partly attributed to the differing mobilities of P and N in soils; P is often insoluble and primarily transported in erosion and runoff, whereas N is highly soluble and readily leached. Even so, such a separation is artificial as P and N occur simultaneously in watersheds and farmers manage them together. Thus, as we move forward to remediate water quality, both N and P must be considered in concert.

THE EVOLUTION OF AGRICULTURAL SYSTEMS

Post-WWII improvements in agriculture have dramatically increased grain and protein production in a very cost-effective manner. The specialization and fragmentation of crop and animal-production systems, however, has brought new pressures to bear on agricul-

tural management within watersheds. Watersheds generally had a sustainable nutrient balance, whereas nutrients are now moved, either as inputs (fertilizer and feed products) or produce on a global scale, which brings new pressures, challenges and therefore, solutions. For instance, increased grain and animal production in Brazil is making inroads into traditional US markets and US producers supply a large percent of the meat consumed in Japan as water-quality constraints in Japan limit cost-effective production there.

As a consequence of the spatial separation of crop and animal-production systems, fertilizers are produced (*i.e.*, N) or imported (*i.e.*, P) to areas of grain production (Figure 2). The grain and harvested N and P are then transported to areas of animal production, where inefficient animal utilization of nutrients in feed (<30% utilized) results in their excretion as manure. This has led to a large-scale, one-way transfer of nutrients from grain-to animal-producing areas that crosses watersheds and even national boundaries and has dramatically broadened the emphasis on watershed-management strategies (Figure 2).

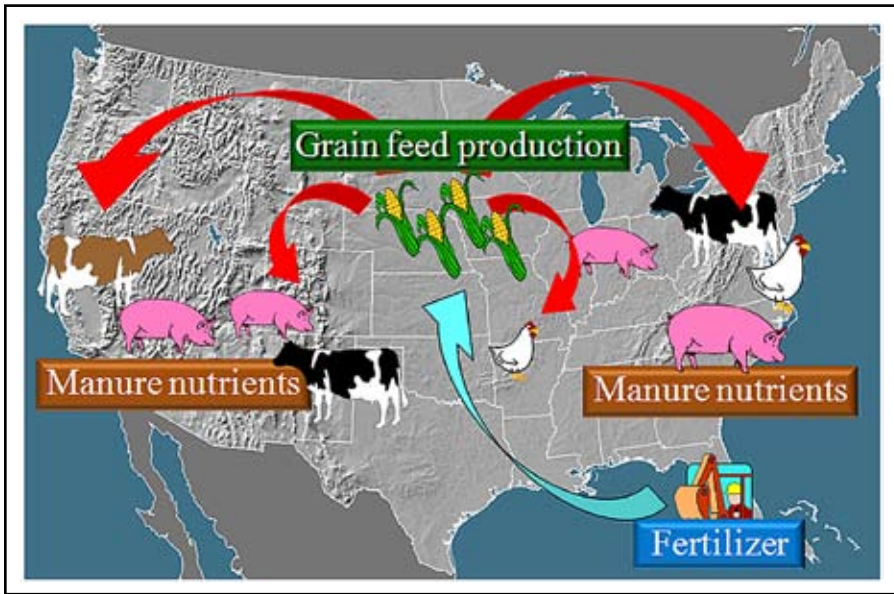


Figure 2. Grain- and animal-production systems have evolved in spatially separate areas leading to localized accumulations of nutrients in manure.

In general, watersheds dominated by animal-feeding operations (AFOs) have become net sinks for nutrients imported in fertilizer to apply to local crops or in animal feed, as animals utilize <30% of the nutrients they are fed. Consequently, watersheds that include AFOs determine the magnitude of nutrient surpluses at farm and watershed scales depending on the type and intensity of livestock operations and the land area available for spreading of the manure produced. For example, the potential for N and P surpluses on farms with AFOs can be much greater than in cropping systems where nutrient inputs

become dominated by livestock feed rather than fertilizer purchased explicitly for application to crops. As the intensity of animal production within a watershed increases, more P must be recycled, the P farm surplus (input-output) becomes greater, soil P levels increase, and the overall risk of nutrient loss tends to increase (Pote *et al.*, 1996; Haygarth *et al.*, 1998; Sharpley, 2000; Torbert *et al.*, 2002; Daverede *et al.*, 2003; Withers *et al.*, 2003).

THE FATE OF N AND P IN AGRICULTURE

The fate of N and P in agriculture is depicted in Figure 1. Approximate percentage inputs of N and P, as feed or fertilizer, and their respective fate when fed to animals or applied to land, were obtained from several studies, as summarized below.

Nitrogen

In rain-fed systems, an average of 40 to 60% of the N applied in fertilizer and manure is removed in crop harvest (Figure 1; Howarth *et al.*, 2000). Of the remainder, varying amounts are stored as organic N in the soil, volatilized to the atmosphere, and leached. A variety of factors, including soil properties, climate, fertilizer, manure, and land management, influence the fate of applied N (Howarth *et al.*, 1996). For typical rain-fed farming in the United States, roughly 15% of field-applied N is volatilized to the atmosphere as ammonia and N_xO (Figure 1). This N is deposited back onto the landscape, often near the source of volatilization, although sometimes traveling long distances within an airshed (Holland *et al.*, 1999). Again, for typical farming systems in the United States, the percentage of applied N that leaches varies between 10 and 40% for loam and clay soils, and 25 and 80% for sandy soils (Howarth *et al.*, 1996).

Phosphorus

Several soil and crop factors, including soil-P-sorption capacity, crop type, P-application type, method and rate, and land management, influence plant uptake of P (Pierzynski and Logan, 1993; Sims and Sharpley, 2005). Phosphorus loss by leaching is generally greater in sandy, organic, or peaty soils—those with low P-adsorption capacities—and in soils with substantial preferential flow pathways (Sharpley and Syers, 1979; Bengston *et al.*, 1988; Sims *et al.*, 1998) (Figure 1). Phosphorus transport in surface runoff is generally greater than in subsurface flow, depending on: the rate, time, and method of P application; form of fertilizer or manure applied; amount and time of rainfall after application; and land cover (Sharpley and Rekolainen, 1997). While P loss from tile-drained soils receiving manure is generally low, there has been a dramatic increase in the extent of tile drainage since 2005 in the Lake Erie Basin, which has connected more fields to ditches and streams and, along with a major shift to no-till cropping, has contributed to increased inputs of P to Lake Erie (Richards *et al.*, 2010).

REMEDIAL MEASURES

Many beneficial management practices (BMPs) can be implemented over a wide range of scales to minimize the loss of N and P from agriculture to surface and ground waters (Table 2). These are commonly grouped into measures that seek to reduce the inputs of

TABLE 2. BENEFICIAL MANAGEMENT PRACTICES TO MINIMIZE THE LOSS OF N AND P FROM AGRICULTURE.

Practice	Description	Impact on loss [‡]	
		N	P
Farm Inputs			
Crop hybrids	Low phytic-acid corn reduces P in manure	neutral	decrease
Feed additives	Enzymes increase nutrient utilization by animals	decrease	decrease
Feed supplements	Match animals nutritional requirements	decrease	decrease
Source Management			
Crop requirements	Nutrient applications based on crop N &/or P needs	decrease	decrease
Pre-sidedress N Test	PSNT can aid accurate split N applications	decrease	neutral
Soil P testing	Nutrient applications based on soil P availability	neutral	decrease
Tissue testing	N applications can be tailored to crop needs	decrease	neutral
Cover crops/residues	If harvested can reduce residual soil nutrients	decrease	decrease TP increase DP
Site-specific management	Use of GIS & GPS to apply and manage nutrient sources	decrease	decrease
Method of application	Incorporated, banded, or injected in soil	decrease	decrease
Rate of application	Match crop needs	decrease	decrease
Source application	Sources can differ in their P & N availability	decrease	decrease
Timing of application	Avoid application to frozen ground Apply during season with low runoff probability	decrease	decrease
Transport Management			
Conservation tillage	Reduced and no-till increases infiltration and reduces soil erosion	decrease TN increase NO ₃	decrease TP increase DP
Strip cropping, contour tillage, terraces	Reduces transport of sediment-bound nutrients	decrease TN neutral NO ₃	decrease TP neutral DP
Conservation cover	Permanent vegetative cover increases soil infiltration and water holding capacity	decrease	decrease
Invert stratified soils	Redistribution of surface P through profile by plowing	neutral	decrease
Buffer, riparian, wetland areas, grassed waterways	Removes sediment-bound nutrients, enhances denitrification	decrease	decrease TP neutral DP
Critical source area	Target sources of nutrients in a watershed for treatment	decrease	decrease

N and P onto farms and bring them into closer balance with outputs in produce; manage on-farm nutrient sources through appropriate rate, timing, and method of N and P application; and measuring the potential for N and P transport to surface and ground waters (Table 2). These measures are also depicted in Figure 3.

Remedial Input Decisions

Carefully matching dietary inputs to animal requirements can reduce the amount of P excreted. In a survey of Wisconsin dairy farms, Powell *et al.* (2001) found that decreasing dietary P from an excessive 0.55% P to the NRC-recommended level of 0.36% P would reduce by about two thirds the number of farms and acreage with an excess P balance (Powell *et al.*, 2002). Implementing a carefully planned diet tailored to meet the specific

[‡]TN is total N, NO₃ is nitrate, TP is total P, and DP is dissolved P.

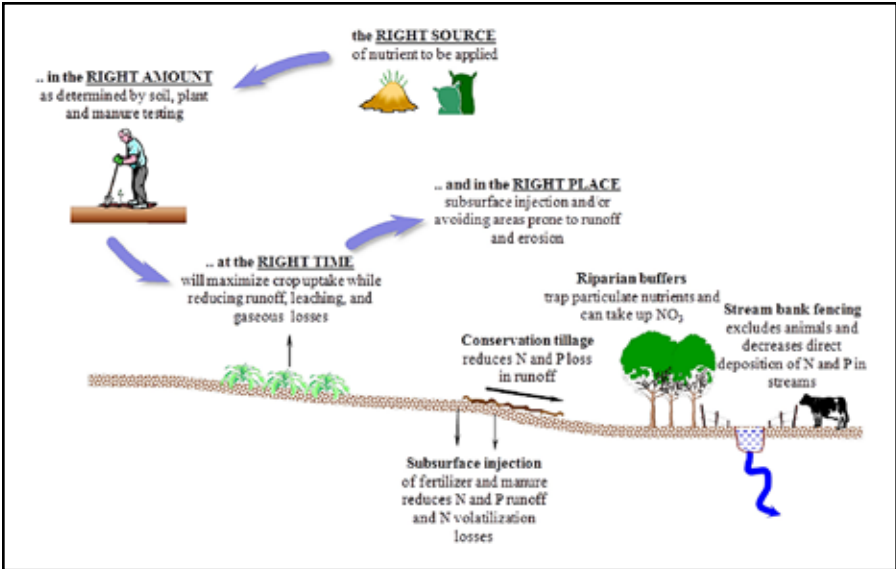


Figure 3. The management of N and P in agricultural systems.

N and P requirements of animals in each phase of their growth will minimize nutrient loss to the environment in feces, urine, and gases. Reducing farm inputs of N and P in animal feed is a very effective BMP that can contribute to bringing about lasting decreases in N and P losses to the environment. In fact, other nutrient-management measures (Table 2) are generally aimed at decreasing the potential for N and P losses and are seen as short-term “band aids” and not solutions.

Nutrient Management

Careful nutrient-management planning on a field-by-field and farm basis is a major component of any remedial action plan to minimize the risk of nutrient loss from agricultural lands. This basically follows the traditional “4R” nutrient management approach, which is adding the Right form of nutrient, at the Right rate to match crop needs, at the Right time, and in the Right place (Figure 3).

The Right Source Fertilizer N, P, and K can be formulated to match crop needs; however, manures have relatively more P than N compared with crop requirements. For instance, the ratio of N:P in manure (2 to 4:1) is about three to four times lower than that taken up by major grain and hay crops (8:1) (Figure 4). As a result, application of manure to crops on an N-basis, where the N requirements of the crop are met by manure, over-applies P three to four times than that taken up annually by the crop (Figure 4). Repeating N-based manure applications over a number of years will eventually increase soil P levels above those optimum for crop needs, increase the source of P, and thereby the potential for increased P runoff. On the other hand, application of manures based on the P require-

ments of the crop will generally apply insufficient N to meet crop needs (Figure 4). The under application of N is a major economic drawback to P-based-manure applications, as farmers are forced to purchase costly mineral N fertilizer to offset the N shortfall.

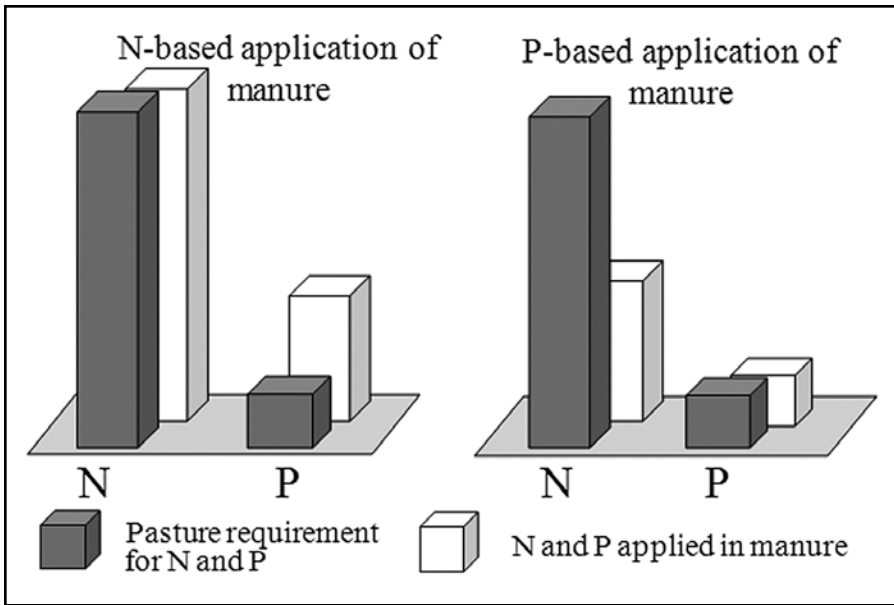


Figure 4. The manure-nutrient balancing dilemma.

The Right Rate Considerable attention has been given to developing soil-N tests for corn, with the preplant nitrate test successfully used in semi-arid regions of the Western and Great Plains (Hergert, 1987); however, the preplant nitrate test does not work well in more humid regions. A modified approach to the preplant assay was proposed by Magdoff *et al.* (1984) to be used in the more humid corn-production areas, known as the pre-sidedress soil-nitrate test to determine if and how much N fertilizer is needed to supplement any early-season N mineralization.

Because N availability is a function of multiple processes within the soil, varying spatially and temporally, more recent research efforts have focused on methods that evaluate the N status of the growing crop. Varvel *et al.* (1997) demonstrated that chlorophyll meters could be used as an index for N status in corn, and N-fertilizer-use efficiency could be improved with this technique. Remotely sensing crop reflectance has been successfully used in a similar capacity, detecting N deficiency (Osborn *et al.*, 2002) and using an index for developing N recommendations.

Fertilizer-P rates are usually established by crop need and modified by the amount already in the soil, as determined by established soil-test P methods (Cox, 1994). In the case of commercial fertilizer P, applications can easily be tailored to match crop needs and minimize excessive soil-P accumulation, because an economic disincentive exists to avoid applying too much costly fertilizer.

The Right Time Nitrogen fertilizer should be applied as close as possible to the time of maximum crop needs and uptake. This minimizes leaching below the root zone and gaseous losses via denitrification and, consequently, should result in the greatest proportion of applied N used by the crop. Although farmers might recognize this situation, deciding when to apply fertilizer depends upon logistical and time constraints unique to every farm. Often, N fertilizer is applied much earlier than required by the crop. In the upper Midwest and Northeast, farmers sometimes are faced with the need to apply N fertilizer on frozen soils. As a general rule, fertilizer should be applied when runoff potential is low.

Many studies show the loss of P in runoff relates directly to the rate and frequency of applied P (Sims and Kleinman, 2005; Sharpley *et al.*, 2007). Avoiding applications within a few days of expected rainfall can minimize runoff losses (Sims and Kleinman, 2005). Several studies show, for example, reductions in N and P losses with an increase in the length of time between manure application and surface runoff (Westerman *et al.*, 1983; Edwards and Daniel, 1993; Sharpley, 1997; Djodjic *et al.*, 2000). These reductions can be attributed to the reaction of added P with soil and dilution of applied P by infiltrating water from rainfall that did not cause surface runoff.

Rainfall intensity and duration, as well as when rainfall occurs relative to when manure is applied, all are factors that influence the concentration and overall loss of manure N and P in runoff. The relationship between potential loss and application rate, however, is critical to establishing BMP strategies. Whereas soil P clearly is important in determining P loss in surface runoff, the rate and frequency of applying P to soil can override soil P in determining P loss (Sharpley *et al.*, 2007). Also evident are the long-lasting effects of applied manure on increased concentrations of P in surface runoff. For instance, Pierson *et al.* (2001) found that applying poultry litter to pastures at N-based rates, elevated runoff P for up to 19 months after application.

The Right Place Nitrogen fertilizers are applied by various methods, including injection, broadcasting, banding, and with irrigation water. The method selected usually depends upon machinery, fertilizer type, and compatibility with a farmer's overall crop-management practices and usually has little impact on crop response to or environmental impact of the fertilizer. There are three exceptions: (1) when urea is applied to the soil surface, especially in no-till, (2) when N fertilizer is band-applied or injected into a micro-feature that improves localized water infiltration, and (3) when N fertilizer is sprinkler-applied on sandy soils.

Urea applied to the soil surface is susceptible to considerable loss as volatilization of ammonia gas (as much as 30% of applied N; Fox *et al.*, 1986). Urea volatilized as ammonia poses an environmental risk as atmospheric ammonium fallout. Ammonium deposition poses an environmental threat to pristine ecosystems that are sensitive to slight changes in N inputs (*e.g.*, estuaries or native forests); on the other hand, the impact of ammonia volatilization from urea fertilizers on such ecosystems is not fully understood. However, because of the relative immobility of P in the soil profile, placement of fertilizer P generally is more critical for plant availability than in the case of fertilizer N.

The incorporation of manure into the soil profile, either by tillage or subsurface placement, reduces the potential for P runoff. Rapid incorporation of manure also reduces ammonia volatilization and potential loss in runoff, as well as improving the N:P ratio for crop growth. For example, Mueller *et al.* (1984) showed that incorporation of dairy manure by chisel plowing reduced loss of total P (TP) in runoff from corn 20-fold, compared to no-till areas receiving surface applications. In fact, P loss in runoff declined because of a lower concentration of P at the soil surface and a reduction in runoff with incorporation of manure (Mueller *et al.*, 1984; Pote *et al.*, 1996).

TRANSPORT MANAGEMENT

Transport management refers to efforts to control the movement of P from soils to sensitive locations such as bodies of fresh water.

Runoff Potential

The potential for runoff from a given site is important in determining the loss of P and, to a lesser extent, N, and is thus a critical component of nutrient-management strategies. Distance from where runoff is generated to a stream channel influences N and P loss and, thus, must be a nutrient-management consideration (Gburek and Sharpley, 1998). Runoff, and nutrients carried by it, can be reduced or even intercepted by infiltration and deposition, respectively, prior to reaching a stream channel. Generally, the further a field is from a stream channel the lower the potential for runoff to contribute nutrients to the stream. Therefore, many states have adopted the premise of implementing more restrictive nutrient-management strategies, particularly for P, on fields close to streams.

Erosion Potential

Phosphorus loss via erosion may be reduced by conservation tillage and crop-residue management, buffer strips, riparian zones, terracing, contour tillage, cover crops and impoundments (*e.g.*, settling basins). These practices tend to reduce rainfall impact on the soil surface, reduce runoff volume and velocity, and increase soil resistance to erosion.

Permanent Vegetation

Keeping land in permanent cover, such as grass or cover crops, reduces the risk of runoff and erosion, increases infiltration, and thereby minimizes losses of N and P. Cover crops protect the soil surface from raindrop impact, improve infiltration relative to bare soil, and trap eroded soil particles (Sharpley and Smith, 1994). Where dissolved P transport is the primary concern, cover crops may reduce runoff and, consequently, runoff-P load. Cover crops are unlikely to affect dissolved P concentrations in runoff, however. Kleinman *et al.* (2005) found that cover crops reduced TP concentration in springtime runoff to 36% of the dissolved P in runoff from conventional corn. But dissolved-P concentration was not significantly different between cover crops and conventional corn because it was controlled by soil-P content rather than by soil erosion.

Grassed waterways are designed to trap sediment and reduce channel erosion. In some cases, waterways are installed as cross-slope diversions to intercept runoff and reduce

effective slope length. Chow *et al.* (1999) estimated that installation of grassed waterways and terraces in combination reduced annual soil erosion 20-fold in a New Brunswick, Canada, potato field.

Riparian/Buffer Areas

Healthy riparian areas can reduce N and P export, increase wildlife numbers and diversity, and improve aquatic habitats. In addition to acting as physical buffers to sediment-bound nutrients, plant uptake captures N and P, resulting in short-term and long-term accumulations of nutrients in biomass (Peterjohn and Correll, 1984; Groffman *et al.*, 1992; Uusi-Kämpä, 2000). Enhanced denitrification in riparian areas can reduce the loss of N from agricultural fields to stream corridors (Lowrance *et al.*, 1985; Howarth *et al.*, 2000).

The effectiveness of conservation buffers as a nutrient-management practice can vary significantly. For instance, the route and depth of subsurface water-flow paths through riparian areas can influence nutrient retention. Conservation buffers are most efficient when sheet flow occurs, rather than channelized flow, which often bypasses some of the retention mechanisms. Those areas must be managed carefully to realize their full retention and filtration capabilities.

Critical Source-Area Management

Transport of N and P from agricultural watersheds depends to a large extent upon the coincidence of source (soil, crop, and management) and transport factors (runoff, erosion, and proximity to water course or body). Source factors relate to watershed areas with a high potential to contribute to N and P export. For N, amounts applied in excess of crop requirements can be leached from the soil profile by percolating water. For P, source areas often are spatially confined and limited in extent, generally reflecting soil-P status and fertilizer- and manure-P inputs (Gburek and Sharpley, 1998; Pionke *et al.*, 2000).

Transport factors determine whether nutrient sources translate into nutrient losses. Nitrogen transport from agricultural land generally occurs on a watershed scale. Source factors tend, therefore, to govern the magnitude of N loss, while transport factors dictate the time lag or delay caused by percolation through various soil layers. Nitrogen loss from cropland depends upon the balance between N added (amount, timing, and form of N) and crop removal. The rate of N loss through leaching (transport factor) depends upon soil properties (primarily texture and permeability) and the amount of water available to drain through the soil profile (rainfall and irrigation).

THE EVOLUTION OF PHOSPHORUS INDEXING

Although many BMPs are available to mitigate nutrient runoff under a variety of site-specific settings, selection and targeting of these practices remains a critical need. Site-risk assessment and P Indexing goes a long way to addressing this need. The site-assessment tool, or P Index, was first proposed in 1993 and eventually adopted into the US Department of Agriculture's Natural Resource Conservation Service (NRCS) Conservation Practice Standard for nutrient management (*i.e.*, the NRCS 590 Standard). The P Index was

designed to identify and rank critical source areas of P loss based on site-specific source factors (soil P, rate, method, timing, and type of P applied) and transport factors (runoff, erosion, and proximity to streams) (Lemunyon and Gilbert, 1993). The fundamental advantage of the P Index is to enable targeting of remedial management to critical source areas where high P sources and transport potential coincide (Figure 5).

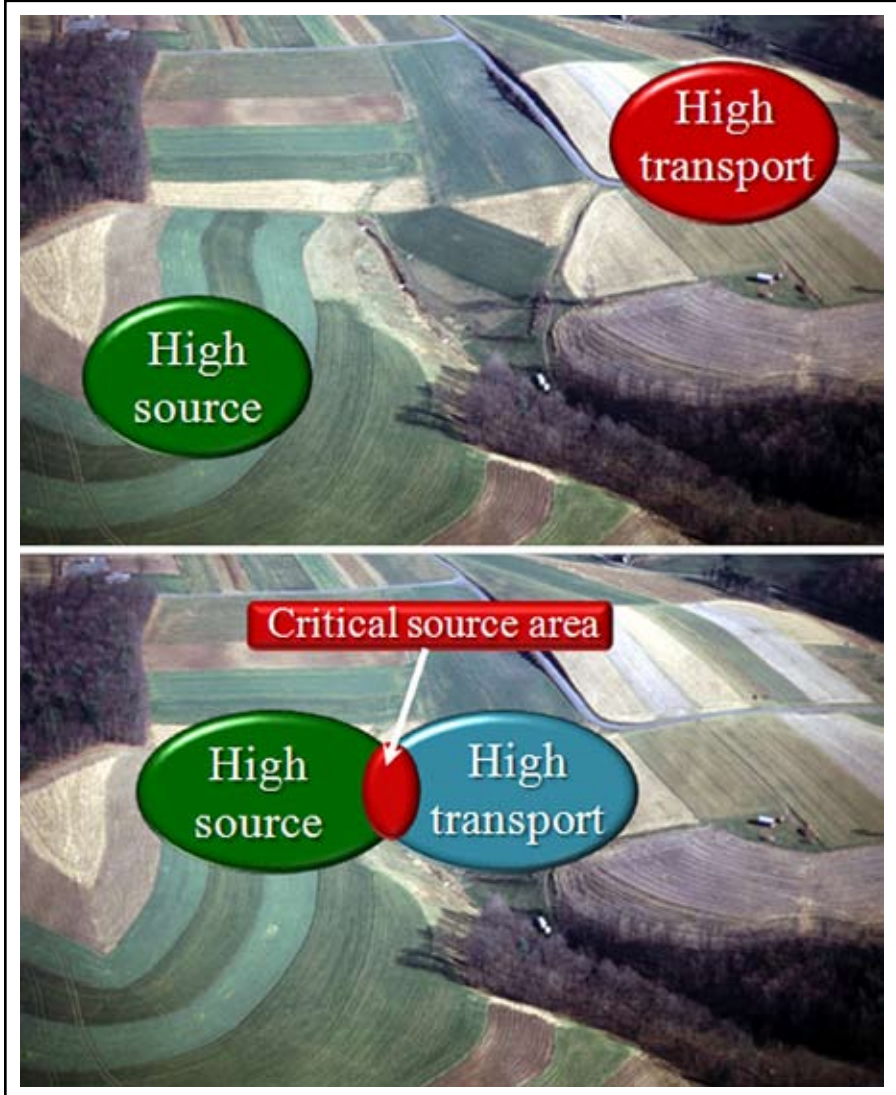


Figure 5. The critical source area concept defining where both high P sources coincide with areas of high transport potential on a landscape delineating high risk areas for remediation efforts.

The scientific basis of P Indices came from research that showed that the majority of the P loss (>80%) originated from only a small proportion of a watershed (<20%) (Pionke *et al.*, 1997; 2000). These critical source areas, as visualized in Figure 5, are essentially P hotspots with active hydrological connectivity by overland flow (Walter *et al.*, 2000; Gburek *et al.*, 2007).

This approach differs profoundly from others that are based solely on soil-P concentration. Although they require more information on site source and transport conditions, P Indices more reliably identify nonpoint sources of agricultural P and provide greater flexibility in remedial options and more cost-effective management recommendations. Currently, 47 US states have adopted the P Index as a site-assessment tool to identify critical source areas and target remedial practices (Sharpley *et al.*, 2003).

The highly significant, positive correlation between soil-test P or degree of P saturation and runoff dissolved P concentration is well established (Vadas *et al.*, 2005) and is frequently used to justify the use of thresholds to limit P application. However, a wealth of scientific evidence is available documenting that, in addition to soil-test P and/or degree of P saturation, P application rate, timing, and method, erosion, runoff, and drainage all influence field-P loss. Use of soil-test P or degree of P saturation alone will not accurately portray a site's risk for P loss because it does not capture the potential for transport potential of a field. If soil-test P is the only assessment used, P runoff and/or leaching losses might be allowed to continue on sites with high P-transport potentials and conversely P application may be restricted although the risk of P loss is low (Figure 6). The data in Figure 6 are from the FD-36 watershed in south-central Pennsylvania (adapted from those presented in Sharpley *et al.*, 2001). Runoff was collected from 2-m² plots subject to 70 mm hr⁻¹ rainfall (to create 30 min of runoff) across the watershed and related to plot Mehlich-3 soil P and soil P saturation of 0- to 5-cm samples collected after rainfall, as well as P-Index ratings determined by the Pennsylvania P Index (Sharpley *et al.*, 2001).

An important lesson from the above analysis is that there were sites with “low” soil-test P and soil-P saturation that had high losses of P due to combinations of factors that include high runoff volumes and/or application of fertilizer or manure. It should be noted that these “low” P sites are above the agronomic response range (*i.e.*, >50 mg P kg⁻¹ as Mehlich-3 soil P). On the other hand, there were sites with low P loss but with high soil-test P or soil-P saturation values (Figure 6).

Clearly, consideration of site hydrology is critical for determining P loss (Gburek and Sharpley, 1998). For instance, Buda *et al.* (2009) monitored contour-cropped fields on a Pennsylvania hillslope, where the bottom field possessed the lowest relative soil-test P (roughly two-fold lower than the other fields). Although this bottom field was the only one that did not receive P amendments during the study period, it yielded runoff volumes roughly 50-fold greater than the other fields included in the study. Annual loads of P from this hydrologically active field were >8 kg ha⁻¹, in comparison to 1 kg ha⁻¹ or less from the other fields. This study highlights the possibility of site hydrology overwhelming source factors and converting a modest source of P into a major P load.

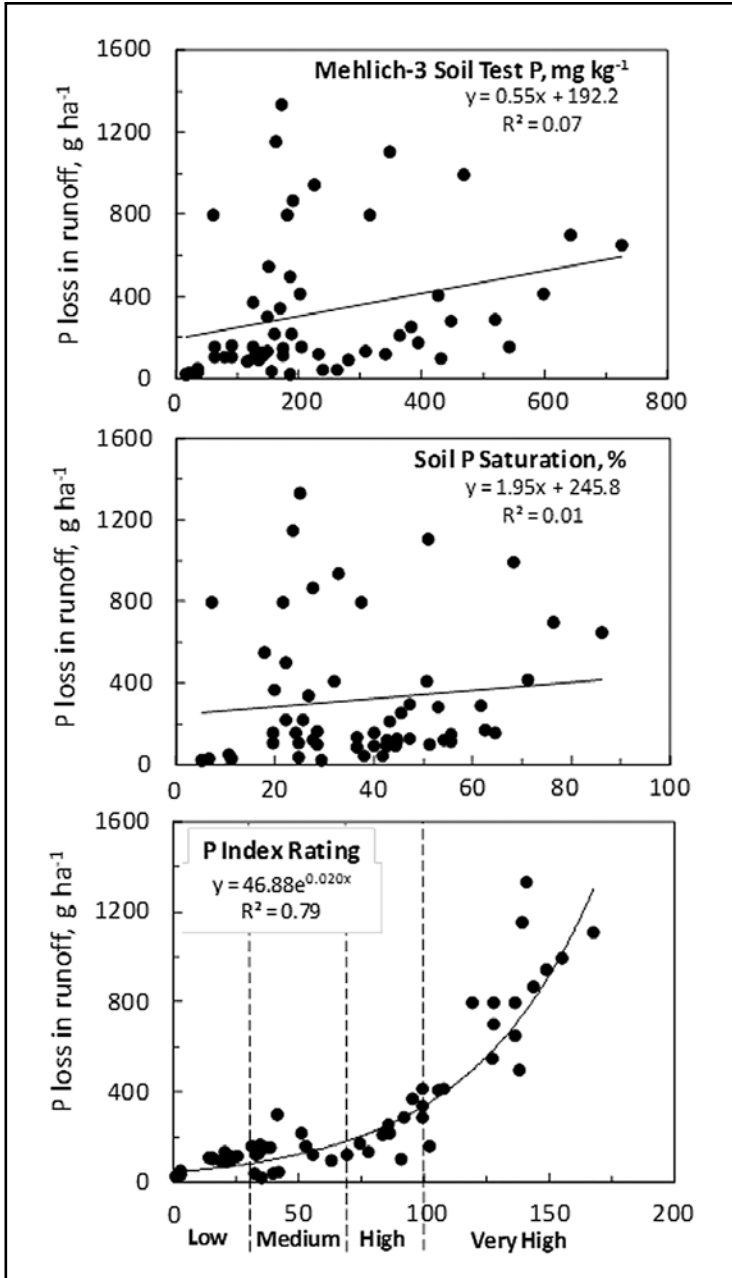


Figure 6. Relationship between the loss of TP in runoff and Mehlich-3 soil test P, soil P saturation, and the Pennsylvania P-Index ratings for the plots in the FD-36 watershed, PA (adapted from Sharpley *et al.*, 2001).

Even in regions where subsurface-flow pathways dominate, areas contributing P to drainage water appear to be restricted to soils with high soil-P saturation and hydrologic connectivity to the drainage network. For example, Schoumans and Breeuwsma (1997) found that soils with high P saturation contributed only 40% of TP load, whereas another 40% came from areas where the soils had only moderate P saturation but some degree of hydrological connectivity with the drainage network.

ADAPTIVE MANAGEMENT

Several land-management practices and system options are available, which, in and of themselves, can lead to nutrient-loss reductions. To be effective in decreasing nutrient loads, where there is a wide range in production systems, however, there must be careful selection and targeting of conservation practices and management strategies. These practices vary in effectiveness among watersheds and there will be synergistic effects on nutrient-loss reductions, where combinations of these practices can further enhance overall reduction, when appropriately targeted to areas of high nutrient-source availability and those that are hydrologically active. This is the basis for adaptive management of these practices, such that if nutrient-loss reductions are not achieved by implementing nutrient efficiency, edge-of-field buffers or offsite wetlands, then one or all are reassessed and modified.

An adaptive management approach provides an appropriate way for decision makers to deal with the uncertainties inherent in the environmental repercussions of prescribed actions and their influences on water quality at multiple scales. At a basin scale, adaptive management requires measurement of both nutrient loadings and the extent and duration of specific water-quality impairment. Although it will not be possible to relate these changes to specific changes in a basin, these data will provide better understanding of the relationships between nutrients and water-quality impairment. On smaller scales, management actions can be treated as experiments that test hypotheses, answer questions, and thus provide future management guidance. This approach requires that conceptual models are developed and used and relevant data are collected and analyzed to improve understanding of the implications of alternative practices. Research driven by adaptive management is conducted in a framework where the testing of hypotheses and the new knowledge gained is then used to drive management adaptations, new hypotheses and new data gathering on endpoints. Unlike the traditional model of hypothesis-driven research, adaptive management implies coordination with stakeholders and consideration of the economic and technological limitations on management.

A basin-level response to practices cannot be expected to be observed for some time. We need a better understanding of the spatial and temporal aspects of basin-level responses, but must also focus on other scales at which responses can occur in a more timely fashion. This would likely be smaller sub-watershed scales, where local water quality and quantity benefits may become evident more quickly, and which enhance practice adoption.

DEMONSTRATING THE BENEFICIAL IMPACTS OF AGRICULTURAL MANAGEMENT ON RIVER-WATER QUALITY

Best management practices addressing source controls (*e.g.*, rate, method, and timing of

applied P) and transport controls (e.g., conservation tillage, contour plowing, and riparian buffers) have reduced concentrations and loads of P in agricultural runoff both at field and at farm scales (Maguire *et al.*, 2009; Sharpley *et al.*, 2009). The Maumee and Sandusky Rivers (tributaries of Lake Erie, Ohio) exemplify how widespread adoption of conservation tillage can dramatically reduce P loads (Richards *et al.*, 2009). However, as described earlier, a range of factors, including surface soil build up of P increased dissolved P transport (Joosse and Baker, 2011). In general, there has been more limited success in reducing river-water P concentrations and loads at a watershed scale (Sharpley *et al.*, 2009; Reckhow *et al.*, 2011). Also, large-scale initiatives such as the Management Systems Evaluation Areas (MSEA) and Agricultural Systems for Environmental Quality (ASEQ) have revealed some of the difficulties in demonstrating the effectiveness of BMPs at watershed scales (Mulla *et al.*, 2008). These difficulties include multiple and complex P sources, inadequate intensity of BMPs moving to larger spatial scales and the role of long-term re-release of “legacy P” stored within the watershed. As a result of this legacy and variable response times, longer-term monitoring (decadal-scale) will likely be needed to demonstrate the benefits of agricultural management on river-water quality at the wider watershed scale (Gassman *et al.*, 2010).

LEGACIES OF PAST MANAGEMENT

Past land use or land-management activities can lead to a long-term legacy of P in watersheds, stored in surface soils, ditches, riparian zones, wetlands and stream and lake sediments. The stored P can be subsequently re-released as the P-storage capacity gradually becomes saturated, or after a change in land use, land management, or effluent management (Kleinman *et al.*, 2011). Legacy P stores may be widely distributed across the watershed, but the precise locations and impacts are currently poorly understood. The lag times associated with release of P from legacy P stores may help to explain the difficulties in detecting water-quality improvements at a watershed scale (Meals *et al.*, 2010). For example, where soil-test-P levels have risen to more than 10 times crop-sufficiency levels, it can take a decade or more to “draw down” soil-P reserves to levels where dissolved P in runoff is substantially reduced (Sharpley, 2003; Hamilton, 2012).

Beyond the field boundary, legacy and water-quality response times are variable and highly dependent on sediment and water-residence times. For example, the lag time for release of legacy P after point-source mitigation in a small, lowland chalk river (the River Lambourn, UK), was only around 8 months (Jarvie *et al.*, 2006). Impoundments, such as ponds, wetlands, lakes, reservoirs and canals, significantly delay recovery, owing to the longer sediment-retention times (Bosch *et al.*, 2009). The “draw-down” of legacy P in lake sediments, via internal re-cycling, can take years to decades to achieve the required P-reduction goals (Seo, 1999; May *et al.*, 2011).

Legacy P and the associated time lags for recovery, often mask the effectiveness of BMPs on river-water quality at a watershed scale. However, by developing watershed-scale monitoring that identifies local-scale improvements and associated time lags in water quality as they occur, watershed planners can start to better understand and plan nutrient-reduction measures.

SETTING NUTRIENT CRITERIA TO MANAGE ECOLOGICAL IMPAIRMENT OF RIVERS

Nutrient criteria are designed to protect water quality for designated uses, which include drinking, contact recreation, and/or ecological quality and biodiversity. Numerical nutrient criteria concentration standards offer regulatory agencies a means to initiate and encourage changes in land use or land management within a watershed. Frequency distributions and stressor-response relationships are two approaches that have been recommended for establishing P criteria (US Environmental Protection Agency, 2000). The frequency-distribution approach involves assessing TP concentrations for either selected “reference” sites or for both reference and potentially impacted rivers over broad spatial and temporal scales (Haggard and Scott 2011). The 75th percentile of TP concentrations in reference rivers, and the 25th percentile of TP concentrations of all rivers covering a broad spatial scale, have been proposed as potential benchmarks for P criteria. The stressor-response approach relies on the relationships between TP (stressor) and various direct and indirect responses of river biological communities such as algal biomass (Smith and Tran, 2010). However, linkages between ecological impairments in rivers and nutrient enrichment are not always well understood (Dodds, 2007).

The frequency-distribution approach assumes that a reference baseline trophic state naturally occurs within a region and that P concentrations need to be reduced to re-establish pre-impacted or pristine “reference” conditions (Soranno *et al.*, 2011). Often, there is a scarcity of data from truly undisturbed, pristine reference sites in many regions (US Environmental Protection Agency, 2010b) and, whilst achieving “pristine” conditions may be a laudable aspiration, it is uncertain whether this can ever be feasible in watersheds where there is agricultural, human population and/or industrial activities. Both the frequency distribution and stressor-response approaches assume that temporal changes in algal growth, at any given river location, will respond to changes in P concentration in direct accordance with spatial patterns in TP-algal response relationships. However, this underlying assumption does not always apply, because relationships between TP and algal biomass are complex. In fact, TP-algal biomass linkages can become decoupled by interactions among many different local environmental factors that influence the accrual or loss of algal biomass in rivers (discussed below).

RECOVERY OF AQUATIC ECOSYSTEMS THROUGH P MITIGATION

The use of nutrient criteria to manage river eutrophication is based upon an assumption of “smooth reversible ecosystem dynamics” (Dent *et al.*, 2002), that is, simple, predictable ecological recovery will result from the introduction of P-mitigation measures. In some cases, reduced P concentrations have indeed resulted in improved river ecology (Kelly and Wilson, 2004; Bowes *et al.*, 2011a). However, in other cases, even after dramatic reductions in river-water P concentrations have been achieved through P-source mitigation, ecological improvements have not occurred and nuisance algal growth has actually increased (Jarvie *et al.*, 2004; Neal *et al.*, 2010; Bowes *et al.*, 2011b). Algal biomass production can become decoupled from water-column P concentrations, for example: when P concentrations remain above the limiting threshold for algal growth (Bowes *et al.*, 2011b); as a result of

luxury uptake during periods of higher P availability; or where high productivity in algal biomass production results in a high supply and turnover rate of P (Dodds, 2003). Algal biomass production is also regulated “top-down” by invertebrate and fish grazers (Kohler *et al.*, 2011). Moreover, *both* N and P can co-limit primary productivity in streams (Dodds and Welch, 2000). Physical factors such as high flow velocities, light availability/shading, temperature, turbidity, channel morphology, and substrate and hydraulic disturbance can also influence accrual of both benthic and phytoplankton biomass (Maret *et al.*, 2010). Also, growing evidence suggests that river-ecosystem recovery is complicated by non-linear responses with hystereses resulting from a range of internal and external feedback mechanisms (Clements *et al.*, 2010).

The River Kennet in southern England is an example where reductions in P concentrations to below nutrient-criteria levels, instead of improving stream ecology, actually resulted in worsening of aquatic ecological status, owing to proliferation of nuisance diatoms (Jarvie *et al.*, 2004). Disturbances to a stressed ecosystem, by introduction of foodweb changes through fish stocking, water abstraction, and changes in flow regime may also contribute to non-linear or hysteretic recovery trajectories. In some cases, the final endpoint may shift or recovery may not occur until P concentrations are reduced far below those concentrations that triggered the original onset of eutrophication. Despite the importance of P for nuisance algal growth in rivers, we cannot always assume that ecological recovery will result from agricultural management to reduce P losses, owing to a range of other controls and feedbacks. Time lags and hystereses may be involved in the recovery path, and we may even have to have to accept that aquatic ecology may get worse before it gets better!

BALANCING COMPETING DEMANDS

As we move forward with putting our knowledge of agricultural management, water quality and ecologically healthy waters into practice, we will need to provide a balance among competing demands and uses for farm production and designated water uses. The main questions that will need to be answered to provide this balance are:

- *How can we equitably balance demands for restoring impaired aquatic ecosystems with the need to ensure food security and meet demands for increased food production?* Inputs to agriculture are required to achieve and maintain maximum crop (and forage) yields at the level of productivity we expect from our agricultural landscape in today’s world. In fact, inputs will likely have to increase to raise food production needed to feed a rapidly growing and more affluent global population. Further, P is an essential component of animal diets to ensure bone development and reproduction, and dietary feed grains are often supplemented with mineral phosphates. Intensification and decoupling of crop and animal systems and the segregation of livestock farms from arable farms to meet market demands for cheap agricultural grain and protein produce, led to greater inputs of P during the last 50 years. This economically driven intensification leads to large surpluses of P in localized areas that far exceed crop needs. At the same time, we have learned,

and have attempted to come to terms with, the fact (with limited success) that P inputs to freshwater systems accelerate eutrophication, which impairs many designated water uses. However, the global dilemma we now face is, how do we raise agricultural production on the same land acreage to feed twice as many, and a generally more affluent population, by 2050?

Clearly, to achieve increased yields and P-use efficiency will require greater, or greater coordinated, recycling of P globally. For instance, efforts need to be focused on how to efficiently use or exploit accumulated soil-P resources by rhizosphere management, *i.e.*, by manipulating rhizosphere chemistry and biology (the thin layer of soil surrounding roots to increase P mobilization and acquisition) and reducing the reliance on chemical fertilizer P. Opportunities include optimization and control of the input of chemical P into soil, use of different plant genotypes, and rhizosphere-management strategies to stimulate P mobilization. Also, P-recycling efforts can provide a major opportunity to better recognise the fertilizer value of manure and to recycle and reuse P from manures and other by-products. Coupled with this, we need to “encourage” more efficient recycling of manure, biosolids, and other by-product nutrients via innovative integration of financial incentives and stricter regulations that could close the P cycle. This will be a fundamental question that needs to be answered in order to meet stricter water-quality standards and nutrient criteria, while still producing cheap food.

- *What mechanisms are available to foster a more open dialogue on uncertainties in the likely outcomes and timescales in use of P-based nutrient management to control eutrophication?* We are dealing with complex systems, the legacy of past land-use practices and recovery pathways that can be long and tortuous. Modern agriculture has developed into very efficient production systems, albeit a one-way transfer from mined areas to grain production and to animal production and human consumption, with a small proportion of P returned to crop-productions areas. Thus, there are large accumulations of P in intensive animal confinement and urban areas. Where P is applied to agricultural land, a portion (~5% to 10%) makes its way to streams, rivers, lakes and oceans. However, a large portion (~50%) is stored within the ecosystem and released slowly to the environment, exacerbated by the problem that there is at least an order of magnitude between optimum levels in soil for crop production (-0.2 mg L^{-1}) compared with waters for algal biomass enrichment (-0.02 mg L^{-1}). In fact, a large proportion of this is stored in agricultural systems, whereas in urban systems there is much more efficient hydrological connectivity/delivery to surface waters.

Mineral resources are part of our natural capital and economy, but are vulnerable to interruptions of supply. There are many “scare stories” regarding the security of supply of minerals for food security like P. This is a consequence of the exploitation of the “low hanging fruit” of major mineral reserves for much of the developed world. Extracting lower-grade reserves carries both a high energy

demand and magnifies environmental impact through enhanced resource dissipation. It is important to recognize that, unlike peak oil, there is no “peak P.” What does exist is a complex juxtaposition of risk-factors that govern P supply. These factors are multidimensional (geological, technical, environmental, social, political, and economic), and closing the loop requires navigating all these risks.

- *What sorts of river and rural environments are achievable and affordable, given that for many of our rivers it may not be possible to achieve “pristine” conditions if we want people, farming and economic growth within our watersheds?* It is clear that remediation of nonpoint and point sources in most watersheds can be expensive, even with cost-share and subsidy programs. Upgrading wastewater-treatment plants to meet stricter nutrient criteria can increase costs exponentially, once discharge-consent thresholds have been lowered to below 0.5 mg L^{-1} , for example. This has put a burden on many small communities, such as in the Chesapeake Bay Watershed where total maximum daily loads (TMDLs) have predicated point-source reductions. Similarly, conservation practices to minimize nutrient loss (in surface and subsurface flows), required by each Bay State with implementation of Watershed Improvement Plans (WIPs), can be extremely expensive to implement and maintain. Thus, there are numerous sources of technical assistance and financial cost-share and loan programs to help defray the costs of constructing or implementing practices that safeguard soil and water resources. Some of these sources are Conservation Technical Assistance (CTA), Conservation Reserve Program (CRP), Conservation Security Program (CSP), Environmental Quality Incentives Program (EQIP), Small Watershed Dam Restoration (SWDR), Special Water Quality Incentives (SWQI), Wetlands Reserve Program (WRP), and Wildlife Habitat Incentive Program (WHIP).

The amount of federal funds available for the EQIP program for the period 2002–2007 was \$5.8 billion, or more than 4.5 times the total spent under the 1996 Farm Bill. Sixty percent of these funds were used to address livestock problems, many of which are nutrient related. Further issues related to whether this program will reach water-quality goals include whether the EQIP funds will be effectively targeted to problem areas and integrated well with other water-quality programs (Clean Water Act’s Total Maximum Daily Load program), and if sufficient public and private organizational resources will exist for effective implementation of the program. In 2010, \$1.2 billion were available for EQIP, which was a shortfall of \$250 million from budgeted amounts. Some watershed-based programs have been established to provide technical assistance and financial support to farmers participating in water-quality-protection programs. Perhaps the most prominent among these is the New York City Watershed Agriculture Program, where savings the city achieved through filtration avoidance have been used to subsidize farmer BMPs at up to 100% cost-share rates in upstate areas that feed the municipal drinking-water reservoirs.

Clearly, with rising costs of water-quality remediation and protection, the question of when “enough is enough” will have to be addressed. For example, the cost of implementing measures to meet a TMDL at the mouth of the Mississippi River to decrease and maintain the hypoxic zone in the Gulf of Mexico to a size of 5,000 km², will be extremely large. Detailed and realistic cost-benefit analyses will be needed on this and similar waters to answer this fundamental question of what is achievable, affordable, and even desired by the majority of watershed stakeholders.

- *Do we need better convergence between science and public perception in defining what constitutes impaired waters?* The current science focus is on numerical standards of biomass accrual or subtle shifts in algal or macroinvertebrate community structure. However, the general public tends not to be concerned until there is visible impairment or species shifts, such as toxic algae, which result in water-quality problems. One way of bridging this gap may be through exploring ideas and gaining consensus on what constitutes “river health” (e.g. Boulton, 1999). Such approaches are helping to bring together ecologists, water-quality scientists, the general public and other watershed stakeholders, through a common goal of achieving healthier and sustainable river environments. “River health” provides a more pluralistic definition, which relies not solely on water quality or ecological criteria, but incorporates wider society values. “River health” also recognizes that rivers are valued for a range of functions, not only as recreation and aesthetic amenities, but for agriculture, industry and to support both urban and rural populations. This highlights potential conflicts between different river uses and the desire to achieve ecosystem structure and function as close to pristine conditions as possible. It also brings challenges about how to best develop simple metrics and tools that incorporate aspects of what society values as healthy river environments alongside more easily-measured chemical, physical, and ecological criteria.
- *Can we implement strategies to move beyond single P-based nutrient criteria and consider other nutrient and pollution controls, together with physical habitat and top-down controls linked to invertebrate and fish interactions to promote more resilient aquatic-ecosystem functioning?* While ecosystem health and aquatic habitat are more fundamental primers of true water-quality status, the use of site factors controlling ecosystem health and functioning adds great complexity to water-quality standards and directives. Numeric nutrient criteria, while simplistic and broad sweeping, are easier to set, implement, and enforce, albeit with limited technical rigor. A compromise approach among various strategies to define critical criteria and conditions leading to use impairment is needed. This should be based on a weight-of-evidence approach, rather than one specific numeric nutrient criterion for a large ecoregion or water system. To be successful, the development and implementation of these tools and strategies will require an honest and forthright two-way dialogue between those developing the foundational science of these

tools and those making and implementing nutrient-management policies. Such dialogue will be essential to limit the softening of technically rigorous and politically difficult approaches to truly reducing excess nutrient loading.

- *Can the principles of adaptive management be used to support more flexible and responsive monitoring, which with stakeholder involvement, evaluates water quality and ecological responses to a range of management options?* Adaptive management implies an understanding that complex problems will require iterative solutions that will be possible only through monitoring, wider community and stakeholder consultation and generation of new knowledge as successive approximations to problem solving are attempted. Stakeholder involvement helps with consideration of the economic and technological limitations on management, focuses attention on where the priorities for remediation lie, and how increasingly squeezed resources might best be allocated. However, perhaps the biggest difficulties for adaptive management are the time lags in water quality and ecological recovery, linked to the legacy of past land-use management and the “re-equilibration” of ecosystems after implementation of management change. These lags in recovery hamper the iterative approach, because they make it difficult to detect whether a management intervention simply has not worked or whether more time (perhaps on a scale of decades) may be needed for improvements to be seen.

CONCLUSIONS

In the past, separate strategies for P and N have been developed and implemented at the farm or watershed scale. Because of differing biology, chemistry and flow pathways of P and N in soil, these narrowly targeted strategies may lead to conflicting or sub-optimal advice. As a result, the prevention of P and N losses from agricultural systems needs to focus on defining, targeting, and remediating source areas of P that combine high soil-P levels with high erosion and surface runoff potentials and source areas of N that coincide with soils of high permeability. Thus, differing levels of management may be appropriate for different areas of a watershed. Overall, inputs in fertilizers and manures should be carefully matched with crop needs over the whole watershed. Short-term remediation of P loss should focus on critical source areas of P export where high soil P, P application, and zones of surface runoff and erosion coincide. However, lasting improvements in water quality can be achieved only by balancing system inputs and outputs of both P and N. Clearly, the management of agricultural nutrients involves a complex suite of options that must be customized to meet site-specific needs that are depicted in Figure 3. Even so, the long-term impacts have been and remain difficult to quantify.

A range of voluntary and regulatory measures can be used to encourage implementation of nutrient-management strategies as part of conservation programs to protect soil and water resources. In general, the success of these measures relates to how well farmers can afford to implement new management strategies and the concomitant level of support or incentives for their adoption. Unfortunately, less attention has been given to mechanisms or programs that support the maintenance of implemented BMPs. Oftentimes, maintenance costs are appreciably greater than implementation costs, particularly as farm labor.

Research that better quantifies the sinks and sources of nutrients as they are transported through a watershed, and the legacies and lags from past land use, will help develop realistic expectations for BMP use and the timescales for aquatic-ecosystem recovery. In addition, continuing educational efforts with the public and farmers regarding the importance and impact of BMPs on environmental quality parameters will be essential to reach environmental goals. In some instances, local or regional governmental controls may be necessary to enhance prompt adoption of practices that will have a positive influence on environmental outcomes.

As we have moved from nutrient management that improves crop production to the environmental-quality arena, we face many challenges in balancing competing demands for protecting and restoring water quality and aquatic ecology, with sustainable and efficient agricultural production. Measures have become more costly to farmers and have raised the old dilemma, “who benefits and who pays?” It is important to recognize that market prices do not always motivate farmers to manage nutrients in an environmentally sustainable way. Consumers can be given a choice about which products they buy, with premiums paid to farmers who provide more environmentally friendly products. But, clearly, current technology and water-quality policy will not permit an unlimited number of animals in a region, or allow production of high-risk crops on land with a high nutrient-loss risk. Relatedly, another important question that should be asked of the public and agricultural communities is: what price are we willing to pay for cheap clean water and low-cost agricultural grains, protein, milk, *etc*?

In some areas, we cannot and should not expect that pristine waters are achievable with ever-increasing population densities and more intensive agricultural production systems to meet demand. The bottom line is that this may require either a reassessment of water-use designations and/or far-reaching societal commitment and support of agricultural system changes.

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