Evaluating the Success of Phosphorus Management from Field to Watershed

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Studies have demonstrated some P loss reduction following implementation of remedial strategies at field scales. However, there has been little coordinated evaluation of best management practices (BMPs) on a watershed scale to show where, when, and which work most effectively. Thus, it is still difficult to answer with a degree of certainty, critical questions such as, how long before we see a response and where would we expect to observe the greatest or least response? In cases where field and watershed scales are monitored, it is not uncommon for trends in P loss to be disconnected. We review case studies demonstrating that potential causes of the disconnect varies, from competing sources of P at watershed scales that are not reflected in field monitoring to an abundance of sinks at watershed scales that buffer field sources. To be successful, P-based mitigation strategies need to occur iteratively, involve stakeholder driven programs, and address the inherent complexity of all P sources within watersheds.

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Reduction in P loads and associated water quality problems at field, farm, and watershed scales has been shown following implementation of conservation measures or BMPs that were targeted to specific and identifiable nutrient sources (e.g., Baker and Richards, 2002; Jokela et al., 2004). At the scale of major impact; for example, hypoxia in the Gulf of Mexico; toxic algal blooms in Chesapeake Bay; ecosystem changes in the Baltic Sea; management strategies become much more complex and the ability to assign a change in water quality to altered management becomes more difficult. Furthermore, when political borders are crossed, at either intra- or international levels, then conflicting political strategies often complicate solutions, which can override local water quality issues and interests (Kronvang et al., 2005; Singh and Gosain, 2004). To a large extent, this is due to an increased diversity of land and water users with different concerns, views of what constitutes impairment, and thus solutions (National Academy of Sciences, 2007a). A good example of this is the Baltic Sea, which is surrounded by several nations that have previously differed considerably in their environmental focus (Kronvang et al., 2005).

The intensification of crop and animal production systems over the last 20 yr has increased P related freshwater problems, such as summer fish kills, unpalatable drinking water, and formation of carcinogens during water chlorination, with agricultural nonpoint sources still considered in need of being reduced (Friedman et al., 2007; Upper Mississippi River Subbasin Hypoxia Nutrient Committee, 2006). In the European Union (EU), the water framework directive (WFD) now requires widespread control of nutrient inputs to rivers specifically to maintain and improve surface water ecology (Council of European Communities, 2000; Hilton et al., 2006). While research has identified agricultural land-use practices that are of highest risk for P loss, their impact on surface water quality at a watershed scale is less well defined (Sharpley et al., 2006; Withers and Lord, 2002; Smith et al., 2005). This results from the fact that P loss at a watershed scale is an aggregation of field level interactions, periodic storm hydrology, in-stream processing,

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Abbreviations: BMP, best management practices; EU, European Union; WFD, water framework directive.

and contributions from other rural sources, such as runoff from farmyards, septic systems, and sewage treatment facilities (Arnscheidt et al., 2007; Jarvie et al., 2006; Johnes et al., 2007). Thus, several questions still remain with regard to the effectiveness of implementing source (e.g., rate, method, and timing of applied P) and transport (e.g., runoff and erosion via reduced tillage, contour plowing, and vegetative buffers) measures in controlling P loss from agriculture. Does implementation of BMPs (i.e., structural and nonstructural methods that reduce the movement of sediment or nutrients from land to surface or groundwater) at field or farm scales translate into decreased P loss and improved chemical/biological water quality at watershed scales? How long will it be before an environmental response is manifest? And where would we expect the greatest response to occur?

This paper discusses the management of P in agricultural systems at field or farm scales to decrease P loss to surface waters at watershed scales, the potential measures and mechanisms by which P loss reductions might be achieved, and the adoption of remedial measures to maximize effectiveness.

Watershed Management

Watershed management of P requires an initial definition of water quality goals. Acceptable baseline conditions have been established in the United States to define and quantify what actually constitutes trophic impairment (Table 1; USEPA, 2000a, 2000b). Because it can be argued that most, if not all, lakes have been impacted by human activity to some degree, reference conditions represent the least impacted conditions or what is considered to be most attainable. However, the difference between baseline and current P concentrations are used to determine and target reductions required in a watershed to alleviate water quality impairment.

In Europe, restoration, protection, and maintenance of "good" water quality is a key goal in managing agricultural watersheds. The recently introduced WFD requires the setting of "reference conditions" for different water body types, to define the degree of impairment and as an aspiration guide for maintenance or improvement (Council of European Communities, 2000). Clearly, nutrient status is only one of many factors influencing the quality of aquatic ecosystems but is equally regarded as one of the most pervasive water quality issues throughout the EU. As a result, EU countries have governmental powers to implement the WFD and water quality goals are achieved through river basin management planning. Throughout the EU, such goals are intended to be considered in the program of preventative measures that will be established by 2009 for each national/international River Basin District.

Scale of Management and System Response

The complexity of P management, in terms of source identification, remedial efforts, and assessing which strategies have actually led to P load reductions, increases with scale for agricultural and mixed land-use watersheds. Water quality concerns for P are driven at a watershed rather than farm scale, reflecting an aggregation of component field activities affecting P loss. Nonagricultural nonpoint and point sources also contribute to P loss at this scale, confounding estimates of agricultures' contribution to water quality impairment. As a result, the success of remedial efforts depends on addressing the variability of scale (spatial and temporal), and the need to consider all sources of P within a watershed. Further, remedial success is more difficult to define at a watershed scale, as it becomes harder to link practice change and water quality response in an adaptive management context.

In the United States, the 167,000 km^2 Chesapeake Bay watershed drains six states and is the center of intensive activities and research directed at curtailing nutrient loads from land to water. A 1987 compact called for a 40% reduction in N and P loadings to the Chesapeake Bay from point and nonpoint sources by 2000, with agriculture being the major source of nonpoint source P loadings (Taylor and Pionke, 1999). Early efforts to minimize agricultural P losses were largely geared toward erosion control, and were insufficient to control P export from areas of intensive animal production, such as the Atlantic coastal plain (Boesch et al., 2001). In some areas of the Atlantic coastal plain, subsurface transfers of P account for the majority of P export (Kleinman et al., 2007). As a consequence, excess P inputs continue to impair the quality of Chesapeake Bay watershed (USEPA, 2006), and recent efforts have been expanded to account for sources and pathways of P not addressed by conventional conservation practices (Friedman et al., 2007; Sims and Kleinman, 2005). Practices include managing P application rates and timing to avoid the potential for direct transfers to surface waters, novel application methods for direct P incorporation, drainage management, and improved site assessment tools for targeting practices.

Part of the dilemma with BMP implementation and assessment at a watershed scale is the differential flow pathways and mechanisms controlling P loss. While P loss tends to be well defined spatially, N losses are generally less scale dependent and more management related, occurring from a large area of a watershed (Heathwaite et al., 2000). For P, BMPs are targeted at critical source areas based on watershed research showing that the majority (~80%) of the loss originates from only a small proportion (~20%) of the watershed; the 80:20 rule. These are essentially P hotspots with active hydrological connectivity by fast storm flow paths such as overland or near surface flow (Pionke et al., 1996, 2000). In watersheds where subsurface flow dominates, however, critical source or hot spot areas are less evident. For example, most of the P exported from the Dutch Schuitenbeek watershed (up to 14 kg P ha^{-1}) originated from a relatively large area at a lower rate (2–4 kg P ha^{-1}; Schoumans and Chardon, 2003). Thus, it is also important to obtain quantitative apportionment of water outflow from a field or watershed into overland flow and subsurface leaching, since reduction in P losses through different pathways requires different BMPs.

In the United States, BMPs were targeted to agricultural nonpoint sources, to remediate deteriorating Great Lakes water quality. Between 1975 and 1995, in the Maumee and Sandusky River tributary watersheds of Lake Erie, conservation tillage increased from virtually nothing to 50% of cropland {mainly no-till soybean [*Glycine max* (L.) Merr.] and some corn [*Zea mays* L.]}; 75,000 ha (<5% of total farmland in the watersheds) were taken out of production (i.e., Conservation Reserve Program), and applied fertilizer and manure P decreased (Baker and Richards, 2002). These measures translated into significant decreases in total (40%) and dissolved P (77%) over the 20-yr monitoring period.

In areas of intensive livestock production, the inefficient transfer of fertilizer nutrients from manures to crops has and continues to be the primary concern, as manure P invariably exceeds crop P requirements when manure is applied on an N basis. The effect of transitioning from N to P-based rates has been evaluated in a number of settings, including on an Othello silt loam (fine-silty, mixed, active, mesic Typic Endoaquults) at the University of Maryland Eastern Shores Research Farm, Princess Anne, MD. The farm is situated on the Atlantic coastal plain, bordering the Chesapeake Bay to which it drains and has high soil P concentrations as a result of 20+ yr of poultry litter additions (for additional details see Kleinman et al., 2007). From 2000 to 2004 poultry litter was applied at three rates, using anhydrous ammonia fertilizer to achieve crop N requirements where litter rate was insufficient: crop N requirement (corresponding to 40–116 kg P ha^{-1} yr^{-1}); crop P requirement (20–58 kg P ha^{-1} yr^{-1}); and no litter. After 3 yr, the effect of P-based and no litter strategies on decreasing P loss compared to N-based strategy began to become apparent (Table 2). The increase in P losses over time for all three strategies coincided with annual rainfall and runoff volumes, which clearly overwhelmed the management of source controls (Table 2). However, on a relative basis, dissolved and total P losses were a respective 83 and 80% lower from the no litter treatment than from the N-based treatment by the fifth year after treatment implementation (Fig. 1). Notably, soil P (0–5 cm depth) did not exhibit a consistent change with litter P application rate, nor did the quality of adjacent ditches appear to be affected by trends in P loss from the litter-amended fields they drained (Fig. 2). Therefore, relative advances at one scale are readily overwhelmed by variables that may not react on the same temporal or spatial scales.

This inherent spatial and temporal variability in system response to management change begs the question of how long monitoring programs are needed to reliably demonstrate the success or lack of, of implemented remedial measures. For instance, Moosmann et al. (2005) found that for several agricultural watersheds (3–42 km^2) dominated by livestock and cropping activities that drain into two Swiss Plateau lakes, at least 30 flow and concentration measurements were needed to show a 3% change in dissolved P loads over a 5-yr period. It was concluded that to detect an expected trend, fewer measurements were required the longer the monitoring program (Moosmann et al., 2005).

Effective reduction of P loads requires careful selection and targeting of conservation practices and management strategies. Even so, conservation practices vary substantially in effectiveness within and among watersheds. For example, previously reported total P reduction efficiencies for BMPs, such as cover crops can range from 7 to 63%, contour plowing 30 to 75%, livestock exclusion 32 to 76%, and riparian buffers 40 to 93% (Table 3). Such variability results from inherent heterogeneity of landscape topography, hydrology, climate, and prior land use, which influences soil test P. This large variability clearly demonstrates the site-specificity of BMP reduction efficiencies and highlights the dangers of having to assign an absolute value, as required by nutrient trading programs (USEPA, 2003). Briefly, nutrient management trading programs involve buying and selling credits that are based on the load reduction achieved by implementing a specific practice. In theory, nutrient trading allows sources with high cost solutions (e.g., industrial point sources) to obtain credits from sources that can reduce their nutrient loads via low cost solutions.

There can also be synergistic effects of BMPs on P loss reductions, where combinations of practices produce more (sometimes less) than the sum of their individual reductions. Understanding the potential for such interactions is important to properly designing BMP strategies (Simpson and Weammert,

Table 1. Baseline P concentrations for each of the aggregated nutrient ecoregions in the United States for freshwater systems (adapted from USEPA, 2000a, 2000b).

Region	Aggregated ecoregion description	Total P: Rivers and streams	Total P: Lakes and reservoirs
		mg L^{-1}	
I	Willamette and Central Valleys	0.047	–
II	Western Forested Mountains	0.010	0.009
III	Xeric West	0.022	0.017
IV	Great Plain Grass and Shrub Lands	0.023	0.020
V	South Central Cultivated Great Plains	0.067	0.033
VI	Corn Belt and Northern Great Plains	0.076	0.038
VII	Mostly Glaciated Dairy Region	0.033	0.015
VIII	Nutrient Poor Largely Glaciated Upper Midwest and Northeast	0.010	0.008
IX	Southeastern Temperate Forested Plains and Hills	0.037	0.020
X	Texas-Louisiana Coastal and Mississippi Alluvial Plain	0.128†	–
XI	Central and Eastern Forested Uplands	0.010	0.008
XII	Southern Coastal Plains	0.040	0.010
XIII	Southern Florida Coastal Plain	–	0.018
XIV	Eastern Coastal Plain	0.031	0.008

† This high value may be either a statistical anomaly or reflects a unique condition.

Table 2. Mehlich-3 extractable soil P, runoff, and P loss as a function of basing poultry litter applications on a crop N requirement (N-based), crop P requirement (P-based), and soil test P for 0.1 ha plots in Coastal Plain region of Maryland.

Parameter	Treatment†	2000	2001	2002	2003	2004
Rainfall, cm		60.9	45.4	63.6	61.7	94.5
Runoff, cm		0.05	4.97	3.39	2.48	8.00
Mehlich-3 extractable soil P, mg kg^{-1}						
	N based	480	446	511	523	499
	P based	482	470	464	500	452
	Soil test P	488	430	480	480	465
Runoff P loss, g ha^{-1}						
Dissolved P	N based	0.03	0.75	173	1239	3112
	P based	0.05	0.21	21	ND‡	1063
	Soil test P	0.28	25	2	767	517
Total P	N based	0.74	142	590	1335	3493
	P based	1.32	3	162	ND	1386
	Soil test P	2.13	187	20	870	689

† Amounts of P applied in poultry litter averaged 75, 35, and 0 kg P ha^{-1} for N-based, P-based, and soil test P treatments, respectively.

‡ No flow recorded for the P-based treatment in 2003.

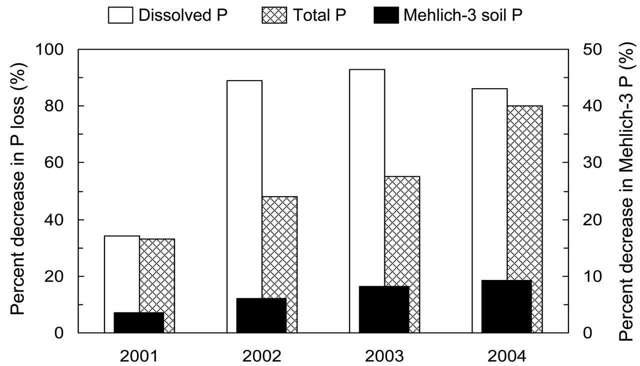

Fig. 1. Decrease in dissolved and total P loss in runoff and Mehlich-3 soil P for fields receiving no P compared with N-based poultry litter applications as a function of year after nutrient management implementation in 2000 for a corn-soybean rotation in the Coastal Plains region of Maryland.

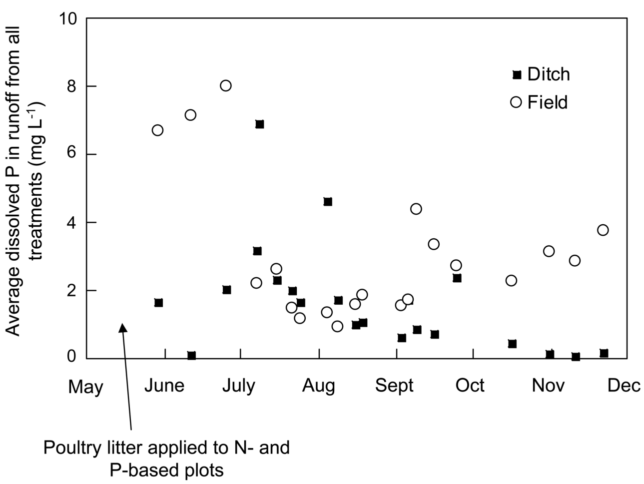

Fig. 2. Disconnected trends in field runoff and receiving ditch water quality for the 2003 growing season at Princess Anne, MD (data adapted from Kleinman et al., 2007).

2007). It follows that detection of the results of these strategies must depend on a clear monitoring strategy. At a watershed scale, assessing system response requires measurement of both P loadings and the extent and duration of specific water quality impairment. However, at a smaller field or farm scale, remedial activities and P loss response can provide specific information to guide selection and adoption of future remedial measures.

Resolution Level of Watershed Assessment

Even when improvements are made at the field scale, they often do not translate immediately to broader scales. As described above for the poultry litter trials on the Atlantic coastal plain soil, changes in P runoff at the field scale did not immediately impact flow from ditches draining the experimental area (Kleinman et al., 2007). Dissolved P concentrations in field runoff following poultry litter application peaked as a result of soluble P additions in the poultry litter for both N- and P-based treatments but no similar trend was observed in ditch flow (Fig. 2). It is likely that well buffered field subsoils and ditch sediments impeded the immediate translation of field runoff improvements to ditch water quality results, even though the ditches drained watersheds of only 2.2 ha.

Even when considering the cases for successful mitigation of P sources and transfers in agricultural watersheds, care needs to be applied in more complex watersheds with multiple P sources with different hydrological controls. For example, in Northern Ireland, diffuse P loss from storm events on grassland soils is considered to be the greatest single contributor to eutrophication in inland (Loughs Neagh and Erne) and coastal waters (Smith et al., 2005). Hydrology is influenced by impermeable clay dominated soils of glacial origin, resulting in flashy flood runoff and suppressed baseflows (Wilcock, 1997), and the bulk of annual P loads generally occur during the winter (Jordan et al., 2005b; Douglas et al., 2007). However, consistently high (often >0.25mg L^{-1}) total P concentrations between storm events have been shown to dominate the trophic status of receiving rivers and linked to nonagricultural and nonpoint sources, such as from poorly maintained septic systems and paved surfaces (Arnscheidt et al., 2007).

Table 3. Potential total P reduction efficiencies (percent change) in surface runoff. Estimates are average values for a multiple year basis.

Conservation practice	Total P reduction	Reference
	%	
Source measures		
P rate balanced to crop use vs. above recommended rate	15–47	Dinnes, 2004
Subsurface applied P vs. surface broadcast	8–92	Dinnes, 2004
Adoption of nutrient management plan	0–45	Devlin et al., 2003; Gitau et al., 2005
Transport measures		
No-till vs. conventional tillage	35–70	Devlin et al., 2003; Dinnes, 2004
Cover crops	7–63	Dinnes, 2004
Diverse cropping systems and rotations within row cropping	25–88	Dinnes, 2004
Contour plowing and terracing	30–75	Devlin et al., 2003; Gitau et al., 2005
Conversion to perennials crops	75–95	Smith et al., 1992
Livestock exclusion from streams vs. constant intensive grazing	32–76	Dinnes, 2004; Gitau et al., 2005; Smith et al., 1992
Managed grazing vs. constant intensive grazing	0–78	Dinnes, 2004; Gitau et al., 2005
In-field vegetative buffers	4–67	Devlin et al., 2003; Dinnes, 2004; Gitau et al., 2005
Sedimentation basins	65	Gitau et al., 2005
Riparian buffers	40–93	Dinnes, 2004; Gitau et al., 2005; Smith et al., 1992
Wetlands	0–79	Dinnes, 2004; Gitau et al., 2005; Smith et al., 1992

The ability to discriminate predominant sources of P transfer at any time or hydrological condition was investigated by Jordan et al. (2005a, 2007b) at a small watershed scale, using high-resolution monitoring and assigning patterns of P transfers to "event-types" (Fig. 3). Continuous flow measurements were synchronized with stream-side continuous total P analysis (10 min resolution) in the Oona watershed, County Tyrone, a tributary of the Blackwater River watershed that drains into Lough Neagh, Northern Ireland (Jordan et al., 2007b). The watershed (5 km^2) is predominantly in pastures with grazing-based beef and dairy farms, with scattered single dwelling houses.

High resolution total P concentrations were measured over a 2-yr period to provide a unique data set to evaluate not only sampling strategies but "real time" losses of P from the watershed. With monthly sampling, there are clearly many flow events that would not have a measured total P concentration, leading to unreliable loading estimates. At both weekly and monthly sampling intervals, there are several high concentrations of total P from the Oona watershed that are not associated with any major flow event that would indicate storm runoff as the main source of P (Fig. 3). These elevated total P concentrations are likely due to sources other than traditional agricultural runoff, particularly septic system and waste water discharge. While intensive monitoring of P concentrations such as in the Oona watershed cannot be replicated widely, results clearly highlight the complexity of P sources within even small watersheds and that widespread implementation of traditional agricultural BMPs (e.g., filter strips, reduced tillage, nutrient management) may not bring about as great a reduction in P loads as might be expected (Jordan et al., 2005b, 2007b).

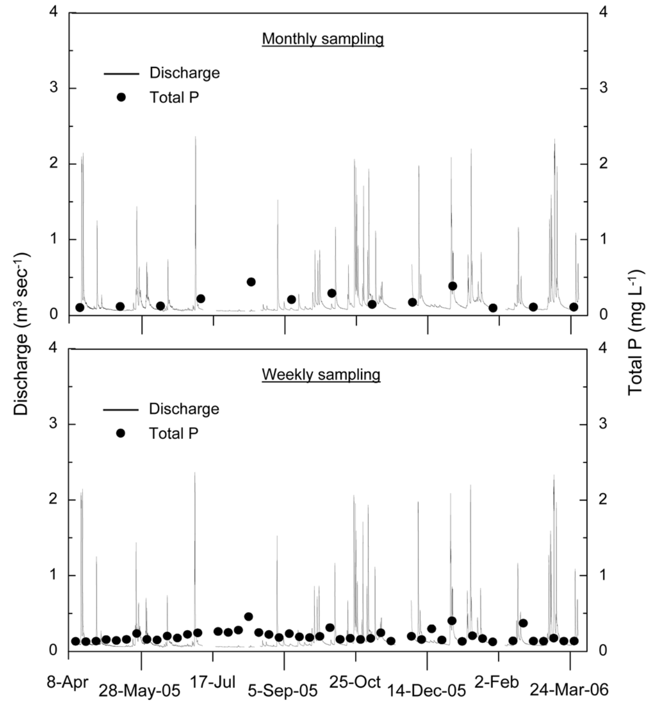

Fig. 3. High resolution synchronous flow and total P concentrations (continuous P analysis at a 10-min resolution) in the Oona Watershed, County Tyrone, Northern Ireland (adapted from Jordan et al., 2007b). Few storm events are captured with the coarse sampling regime and none of the peak flow concentrations (1–2 mg L^{-1}). The data also show a high ambient (nonstorm) concentration between June and October.

Transitions in Watershed Management

Strategic Shifts

In the EU, there has been a fundamental shift from current general guidance on *Good Agricultural Practice* (e.g., Department for Environment, Food and Rural Affairs, 2002) to more proactive implementation of cost-effective and targeted BMPs (Department for Environment, Food and Rural Affairs, 2003), with mutual farmer-regulator agreement of local solutions to local problems. In turn, this will require provision for additional farmer awareness, training, and advisory support, involve a commitment to better record keeping and farm planning, and incur variable levels of cost including capital grant support (Withers et al., 2003). Nevertheless, P already accumulated within some watershed systems is such that even if P was no longer added to agricultural systems, there would be a considerable time-lag (years or decades) before improvements in water quality, or regeneration of diverse habitats, might become apparent. Therefore, it is questionable if, for example, reaching the Swedish environmental quality objective of removing 20% of the P anthropogenic load to coastal waters by 2015 can be detected as improved water quality.

In the United States, the Conservation Reserve and Conservation Reserve Enhancement Programs have proved successful in improving wildlife habitats and water quality through establishing perennial ground cover. Also, there are a growing number of examples in the United States where BMPs have gained wider adoption with a programmatic shift to address socioeconomic barriers that may hinder their adoption. For example, there is a 93% farmer participation in volunteer conservation programs in the Cannonsville Watershed (1180 km^2), which is a drinking water supply for New York City (Watershed Agricultural Council, 2004). A survey of Conservation Reserve Enhancement Program enrollees by James (2005) found that participants were generally older and more likely to obtain information from extension agents and farm advisors than nonparticipants, but there was no difference in educational level or farming status (full or part time). The farmers main concerns to voluntary adoption of BMPs were the loss of productive land and not being able to decide independently what to do on their own land. This survey touches on some of the complexities of BMP adoption in any given watershed, complexities that are related not only to targeting appropriate BMPs to critical source areas in heterogeneous watersheds but to socioeconomic pressures.

Most evaluations of BMP effectiveness at reducing P export from watersheds conclude that nutrient management is an effective measure for controlling P loss (Sharpley et al., 2006). A survey of 127 farms (90% of all farms) in two northeastern Wisconsin watersheds shows that nutrient management can achieve some success in reducing P applications and thereby watershed losses (Shepard, 2005). Farmers with a nutrient management plan (53% of farms) applied less P (31 kg ha^{-1}) than farms without a plan (44 kg P ha^{-1}), and only 75% fully implemented their plans on a majority of their land. Critically, for successful nutrient management planning to decrease P loss, technical and financial assistance programs should focus on plan implementation and maintenance as a whole, rather than on achieving goals set for the number of plans written in a given period.

Production Shifts

Shifts in agricultural production often occur due to external pressures. For example, the increased demand for grain-based ethanol production is likely to have a dramatic impact on agriculture and watershed management that could have unintended yet adverse affects on water quality (Chesapeake Bay Commission, 2007; National Academy of Sciences, 2007b; Simpson et al., 2008). The drive for biofuel production to be a greater share of consumed energy, has led to a 6.5 million ha (16 million acres) increase in corn acreage in the United States from 2006 to 2007 (USDA-NASS, 2007). Projections for 2007 corn planting shows this increase to come from land currently in soybean, Conservation Reserve, and pastures (Elobeid et al., 2006; Wisner, 2007). Assuming fertilizer application rates will be maximized to obtain optimum yields as a consequence of high corn prices, it is expected that the potential for P loss will increase about 25% compared with losses from pre-corn land use (Simpson et al., 2008). Further, dry distillers grain (DDG; 0.8–0.9% P), a by-product of ethanol production, is being used in animal feed (Lawrence, 2006). Even with <20% DDG supplementation of dairy cow (*Bos taurus*) diets, this elevates ration P to 0.5% P (0.33–0.36% P recommended), offsetting reductions gained through feed management (Simpson et al., 2004). This will increase the P content of manure and potential P loss in runoff if land applied (Ebeling et al., 2002; Maguire et al., 2007). However, the process of degerming corn kernels to increase the yield and quality of dry-milled product used in ethanol production (http://www.satake.co.uk/cereal_milling/maize_degerming.htm), has the potential to decrease the P concentration of DDGs and thereby P excreted by animals fed this material. At present, production shifts toward increased ethanol production is stimulated by many governments; but cellulosic biofuel production will eventually increase (Datar et al., 2004; Jordan et al., 2007a; Parrish and Fike, 2005), causing another production shift.

Another example of potentially increased P losses that may occur in response to a shift in agricultural production is when green manure crops are introduced, such as in organic cropping systems. In a Swedish study on a clay soil, average annual P leaching loads were significantly higher ($P < 0.05$) in an organic system with green manures than in a conventional system (Aronsson et al., 2007). Ulén and Jakobsson (2005) presented results from different experiments at the same site, also showing that organic plots with green manure had significantly higher P leaching loads than other plots. In both these studies, incorporation of green manure and subsequent P mineralization were identified as the most critical factors for increased P losses. Watershed management strategies should plan to minimize the potential for possible unintended water quality degradation associated with these production shifts.

The importance of management shifts in watershed, via targeting BMP adoption can be successful at achieving localized P loss reduction as shown by several studies in the Little Washita River watershed (54,000 ha) in central Oklahoma (Sharpley and Smith, 1994). Phosphorus export from two subwatersheds (2 and 5 ha) were measured from 1980 to 1994, while BMPs were installed on about 50% of the main watershed. Practices included construction of flood control impoundments, eroding gully treatment, and conservation tillage. Following conversion of conventional-till (moldboard and chisel plow) to no-till wheat (*Triticum aestivum* L.) in 1983, P loss was reduced 10-fold (2.9 kg ha^{-1} yr^{-1}; Sharpley and Smith, 1994). A year later, shaping eroding gullies decreased P loss fivefold and construction of an impoundment decreased P loss from the subwatersheds 13-fold (Sharpley et al., 1996). However, there was no consistent decrease in P concentration in flow at the outlet of the main Little Washita River watershed. The lack of remedial success at a larger scale is most likely a result of in-stream processes and the continued release of P already stored within the watershed system (McDowell et al., 2004).

Conclusions

As demands for greater P loss reductions from agriculture increase, so does the cost and complexity of remediation. If policies are striving for a 40% reduction, for example, the first 30% may be relatively inexpensive to achieve compared to the remaining 10% (Simpson and Weammert, 2007). Thus, it will be this remaining 10% which will present one the greatest challenges. So who pays? Watersheds are naturally leaky and thus, part of the responsibility should be borne by the public who require clean water along with cheap food. To a large extent this is being accomplished at a "grass roots" level via voluntary alliances and partnerships among all vested stakeholders within a watershed. In the EU WFD, the recovery of costs is requested according to the "polluter pays principle," including both environmental and resource costs. This is clearly stated in the WFD guidance documents (USEPA, 2006), which advocate a mix of public participation, the "polluter pays principle," and cost-effective watershed-wide mitigation measures. However, the costs involved with mitigation strategies to reduce environmental pollution, such as high nutrient loadings from agricultural activities, tend to be underestimated in the pragmatic approach recommended by the WFD Guidelines.

Emphasis needs to be placed on consumer-driven programs for real and lasting changes to occur in farm management that is successful in improving water quality, rather than assuming that farmers will absorb the burden of watershed remediation costs. Except for farm-gate measures, BMPs are "band-aids" to minimizing the environmental impacts of land management. In an attempt to address this, cost-share monies for confined animal feeding operations in northeastern U.S. watersheds are now linked to farmers demonstrating that P inputs to the farm are reduced by feeding animals at a level consistent with National Research Council requirements (Watershed Agricultural Council, 2004).

Even so, consumer-driven programs or stakeholder involvement do not always ensure adoption of remedial measures that decrease P loss. For example, construction of small wetlands to trap P in agricultural drainage waters of central Switzerland only retained 2% of the bioavailable P input (i.e., dissolved P plus a fraction of particulate P) (Reinhardt et al., 2005). While longer residence times were needed for the constructed wetlands to more effectively retain P, Reinhardt et al. (2005) suggest that measures, which inconvenienced farmers least were most likely to be implemented. Similarly, there is reluctance toward streambank fencing to exclude grazing cattle and direct deposition of P in several areas of the United States (e.g., Cannonsville Watershed, New York; James, 2005). This suggests that either regulations are required to force adoption, which may polarize perspectives reducing the possibility of cooperative outcomes, or that adoption of BMPs requires a process of give and take that may likely lengthen the remedial process.

While there are effective P-based BMPs (Table 3), none should be seen or used individually as the primary mechanism by which a farmer reduces P losses. For example, within the EU, subsidies are given for establishment of grass-covered buffer strips along water courses to reduce P losses, making them quite common. However, in many locations where such buffer strips are established, there is no surface runoff, which makes their efficiency in reducing P losses negligible. Furthermore, without targeting source areas, implementation of BMPs over broad areas of a watershed does not always reduce P exports from the watershed as a whole (Meals, 1990; Sharpley and Rekolainen, 1997; Sharpley et al., 1996). At the same time that remedial BMP strategies are being implemented, a robust monitoring program needs to be in place to document a change in water quality. The results from a long-term monitoring program in 22 Swedish watersheds will illustrate the value of baseline monitoring in evaluating the effectiveness of agricultural practices (Kyllmar et al., 2006). Downward trends in P transport in stream outlets occurred in 17 of the watersheds. In seven of those, the trends were significant ($P < 0.05$), and for three, P transport could be correlated with changed cropping strategies and less manure application.

Because of the lag time between BMP implementation and water quality improvements, remedial strategies should consider the re-equilibration of watershed and water-body behavior, where nutrient sinks may become sources of P with only slight changes in watershed management and hydrologic response. A better understanding of the spatial and temporal aspects of watershed response to nutrient load reductions in both flowing and standing water bodies is needed, as well as the scale at which responses may occur in a more timely fashion. This would likely be at a smaller subwatershed scale, where local water quality and quantity benefits may become evident more quickly; and which will enhance practice adoption. However, as shown for the Coastal Plain poultry litter experiments, even at smaller scales, improvements at the field level may not immediately convey to the subwatershed. It is also important to accept in any watershed-P loss reduction strategy, that it is essential to address the overall physical and social complexity of individual systems and the mitigation of nonagricultural sources of P. Only this will bring about lasting improvements in water quality as evidenced in all hydrological (storm and nonstorm) conditions.

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