ENVIRONMENTAL ISSUES

Managing Agricultural Phosphorus for Protection of Surface Waters: Issues and Options

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ABSTRACT

The accelerated eutrophication of most freshwaters is limited by P inputs. Nonpoint sources of P in agricultural runoff now contribute a greater portion of freshwater inputs, due to easier identification and recent control of point sources. Although P management is an integral part of profitable agrisystems, continued inputs of fertilizer and manure P in excess of crop requirements have led to a build-up of soil P levels, which are of environmental rather than agronomic concern, particularly in areas of intensive crop and livestock production. Thus, the main issues facing the establishment of economically and environmentally sound P management systems are the identification of soil P levels that are of environmental concern; targeting specific controls for different water quality objectives within watersheds; and balancing economic with environmental values. In developing effective options, we have brought together agricultural and limnological expertise to prioritize watershed management practices and remedial strategies to mitigate nonpoint-source impacts of agricultural P. Options include runoff and erosion control and P-source management, based on eutrophic rather than agronomic considerations. Current soil test P methods may screen soils on which the aquatic bioavailability of P should be estimated. Landowner options to more efficiently utilize manure P include basing application rates on soil vulnerability to P loss in runoff, manure analysis, and programs encouraging manure movement to a greater hectareage. Targeting source areas may be achieved by use of indices to rank soil vulnerability to P loss in runoff and lake sensitivity to P inputs.

Since passage of the Clean Water Act in 1972, great progress has been made on the control of point sources of pollution. As further control of the remaining point source problems become increasingly less cost-effective, and as significant water quality problems remain unresolved, more attention is being placed on controlling runoff from agriculture and other nonpoint sources of pollution. The USEPA, in a compilation of state reports, has identified agricultural runoff as the cause of impairment of 55% of surveyed river length and 58% of surveyed lake area still having water quality problems (USEPA, 1990). Along with the increased attention to nonpoint-source controls comes the need for new water quality criteria and standards, to identify critical sources needing control, and to target specific controls for different water quality objectives within different watersheds.

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Published in J. Environ. Qual. 23:437–451 (1994).

A major unresolved problem is the accelerated or *cultural* eutrophication of surface waters resulting from nutrient inputs stimulating algal and rooted aquatic plant growth (Thomann and Mueller, 1987). Eutrophication restricts the use of surface waters for aesthetics, fisheries, recreation, industry, and drinking, and thus has serious local and regional economic impacts. Although N and C are required for algal growth, much of the concern with eutrophication has focused on P. This is because the difficulties in controlling the air–water exchange of N and C, and fixation of atmospheric N_2 by some blue-green algae, often result in P being the nutrient that promotes accelerated eutrophication.

In the late 1970s, the International Joint Commission between the USA and Canada placed emphasis on limiting P from nonpoint sources in the Great Lakes Basin (Rohlich and O'Connor, 1980). In the 1980s, increased total P concentrations in Lake Okeechobee, Florida, has raised concern that the lake is becoming hypereutrophic (Federico et al., 1981). To abate this pollution problem, agencies in Florida are developing management strategies to reduce P loads to the lake and other freshwaters. In the Netherlands, the national strategy for minimizing nonpoint-source pollution, especially eutrophication due to animal wastes, is to limit entry of P into both surface and groundwater (Breeuwsma and Silva, 1992).

Due to both easier identification and control of point-source inputs of P and a recent realization of direct human health risks associated with eutrophication, less attention has been given to management strategies to minimize nonpoint P losses. As a result, nonpoint-source pollution by P now accounts for a larger share of the nation's water quality problems than a decade ago (Crowder and Young, 1988; Schultz et al., 1992). The negative impacts of P in receiving waters are balanced by its beneficial use, however. Profitable crop production depends on a sound P management program (as well as several other factors). Judicious fertilizer use can reduce erosion and runoff potential by increased vegetative cover. Clearly, P management is of agronomic and environmental importance.

Sources of P in agricultural runoff include commercial fertilizer and manure. Amounts lost depend on management factors such as time, rate, and method of applica-

Abbreviations: BAP, bioavailable phosphorus; BPP, bioavailable particulate phosphorus; C, carbon; DP, dissolved phosphorus; ECP_o, equilibrium phosphorus concentration; EUTROWASP, Eutrophication Water Analysis Simulation Program; HSPF, Hydrologic Simulation Program–Fortran; N, nitrogen; PP, particulate phosphorus; QUAL-2E, stream water quality model; STP, soil test phosphorus; SWRBB, simulator for water resources in rural basins; TP, total phosphorus.

tion. In recent years, the number of soils with plant-available P (soil test P, STP) exceeding levels required for optimum crop yields has increased in areas of intensive agricultural and livestock production (Alley, 1991; Sims, 1992). In 1989, several state soil test laboratories reported that the majority of soils analyzed had STP levels in the high or very high categories that require little or no P fertilization (Fig. 1). Actual values of STP used to define these categories vary between states and depend on the soil testing extractant used. For instance, very high levels of STP, as measured by the Bray-1 and Mehlich-1 methods, range from 40 to 100 and 50 to 120 mg P kg^{-1}, respectively.

Clearly, high STP levels are a regional problem, with the majority of soils in several states testing medium or low (Fig. 1). For example, most Great Plains soils still require fertilizer P for optimum crop yields. However, Fig. 1 illustrates that problems associated with high STP soils are aggravated by the fact that many of these soils are located in lake-rich states and near sensitive water bodies such as the Great Lakes, Chesapeake Bay, Delaware Bay, Lake Okeechobee, and the Everglades. Thus, one of the main issues facing the establishment of effective nonpoint-source management controls is the development of economically and environmentally sound P management systems and the balancing of productivity with environmental values.

We must start to address these issues, because by the time P-related eutrophication of freshwaters is visible, it is often difficult and expensive to implement remedial strategies. In addition, the benefits of reduced freshwater inputs of P are often not manifest for several years due to internal recycling of P.

This article examines the issues associated with agricultural P use in terms of stimulating, accelerated eutrophication transport by erosion and runoff, environmental soil P testing, and animal manure utilization. Options available to minimize the environmental impact of agricultural P management are presented along with economic and political compromises. We have attempted to balance agricultural and limnological considerations to prioritize watershed management practices and remedial strategies that will mitigate nonpoint-source impacts of agricultural P.

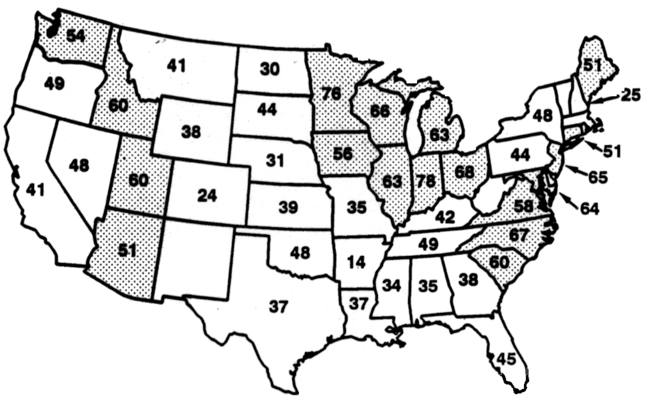

Fig. 1. Percent of soil samples testing high or above for P in 1989. Highlighted states have 50% or greater of soil samples testing in the high or above range (data adapted from Potash and Phosphate Inst., 1989, and Sims, 1993).

ISSUES

Accelerated Eutrophication

Eutrophication Process

The principle impacts of eutrophication relate to four phenomena: increased aquatic plant growth, oxygen depletion, pH variability, and plant species quality and food-chain effects. Decreased water clarity, algal scums, unsightly decaying algal clumps, and discoloration may all occur because of increased phytoplankton population. Large diatoms and filamentous algae can clog water treatment plant filters. In addition, extensive growth of larger plant forms (e.g., rooted macrophytes) can interfere with recreation, navigation, and limit fisheries potential.

Phytoplankton settle to the bottom of quiescent waters and their subsequent decay can deplete oxygen and result in the death of desirable fish species. In highly productive streams, algal activity can lead to large diurnal swings of oxygen and pH in unbuffered waters. The latter can have a deleterious effect on fish by causing a shift of the ammonium ion to the toxic unionized ammonia form.

Eutrophication can lead to a shift of the phytoplankton assemblage to noxious species. Some blue-green algae form mats and gas vacuoles, which allow them to form floating mats. Beyond changes in the phytoplankton themselves, shifts in the species types can influence higher organisms. For example, some blue-greens are not a desirable food source for zooplankton. Consequently, shifts to these organisms can influence the amount of biomass that is available to the zooplankton, which in turn constitute a principle food source for fish. Certain blue-green algae associated with eutrophied waters form potent toxins, cause taste and odor problems and interfere with the treatment of drinking water. Farmers experience economic losses when cattle and other animals die after consuming water containing algal toxins, and these toxins may also pose a serious health hazard to humans (Kotak et al., 1993). In addition to reducing water treatment filter run times, algal blooms can contribute to trihalomethane formation during water chlorination (Palmstrom et al., 1988).

The setting of lake-specific goals is complicated by differing, and often conflicting interests and inherent conditions. Where swimming, boating, aesthetics, and drinking water supply considerations are important, the maintenance of low productivity and clear water is most desirable. Where production of fish biomass is a high priority, a higher level of fertility is sometimes desired. While a number of state resource agencies (e.g., Iowa and Georgia Dep. of Nat. Resour.) have published guidelines for lake fertilization, most states and the USEPA have tended to develop policies designed to restrict increased nutrient loadings. The state of Maine, for example, has implemented a standard requiring "stable or decreasing trophic status" for its lakes (Maine Dep. Environ. Prot., 1986), and British Columbia has established P criteria to protect the most sensitive uses of lakes in that province, particularly drinking water, recreation, aesthetics, and cold water fishing (Nordin, 1986). The state of Florida is in the process of implementing agricultural best management practices (BMPs) to reduce P loads to Lake Okeechobee. Similar practices are followed throughout Florida to reduce P inputs from agricultural

Table 1. Limiting nutrients for various water bodies (adapted from Thomann and Mueller, 1987).

System	N/P ratio	Limiting nutrient
Rivers and streams		
Point-source dominated		
Without phosphorus removal	<<10	N
With phosphorus removal	>>10	P
Nonpoint-source dominated	>>10	P
Estuaries		
Freshwater region		
Nonpoint-source dominated	>>10	P
Point-source dominated	<<10	N
Brackish region	≈10	P or N
Saline region	<<10	N
Lakes		
Large		
Nonpoint-source dominated	>>10	P
Small		
Point-source dominated	<<10	N

and urban activities. In addition to eutrophication of lakes, the Florida Everglades is threatened by P inputs from adjacent agricultural areas. Nutrient inputs to this oligotrophic wetland ecosystem has resulted in the invasion of *Typha* spp. into a *Cladium* sp. dominated system (Koch and Reddy, 1992; Reddy et al., 1993). To abate this pollution, plans are underway to create buffer wetlands between agricultural fields and natural wetlands.

Although increased fertility can, in some cases, be beneficial for fisheries, it can also have detrimental effects on other lake uses. As a result, fertilization as a fisheries management tool is generally restricted to lakes and reservoirs used only or primarily for fishing where nutrient additions will not adversely affect downstream waters. Once a lake is fertilized, returning the system to its original clear water condition can be difficult, expensive, and require a lengthy process, especially for lakes with long water residence times.

Phosphorus or Nitrogen Limitation

Although many factors contribute to the eutrophication process, economically feasible controls generally relate to the supply of P and N (Stumm and Morgan, 1981). Thomann and Mueller (1987) have summarized the general trends of N/P ratios for various water bodies (Table 1). For most inland waters P is the limiting nutrient, whereas in estuaries N appears to be more limiting. As freshwater systems become more dominated by point sources, they tend to move toward N limitation.

Although Table 1 provides a general context for the problems, it should be noted that the situation is more complicated. In particular, there is a temporal aspect to nutrient limitations that relate to the waterbody's capacity to compensate. Schindler (1974, 1977) used evidence from experimental lakes to clearly show P limitation in lakes. However, he also demonstrated that biological mechanisms could act to correct algal N deficiencies. For example, blue-green algae can fix atmospheric N_2. As a consequence, although a sudden increase in P loading might induce a temporary state of N limitation, long-term shifts in the species assemblage can return the system to P limitation. Consequently, most eutrophication control in freshwater has focused on controlling P.

The situation in estuaries is more ambiguous. Although similar temporal adaptation may occur, other mechanisms such as denitrification can keep N levels depressed.

External vs. Internal Sources

It is assumed that reduction in external loads to a water body will provide immediate reduction in eutrophication status. In many cases reduction in external loads may not translate into immediate benefit to the lake, because of steady release of nutrients from bottom sediments. Long-term nutrient loading to lakes has resulted in accumulation in bottom sediments, and thus provides a steady source. For example, in Lake Okeechobee, Florida, internal P release from bottom sediments were in the same order of magnitude as external sources (Reddy, 1992). Similarly, in a shallow hypereutrophic lake in central Florida, more than 90% of the P requirements of algae are met by internal P release by bottom sediments (Reddy and Graetz, 1990).

Modeling Approaches

A variety of modeling approaches are presently available to assess the impact of loadings on natural waters. These fall into four general categories:

Empirical Correlations. These models focus on lakes and reservoirs. They consist of plots and regression equations that predict steady-state trophic status and water quality parameters (chlorophyll, water clarity, oxygen, etc.) as a function of nutrient loadings. Although most focus on P (Vollenweider, 1976; Rast and Lee, 1978), some also include N loads in an attempt to discriminate between N and P limitation (Smith and Shapiro 1981). None of the empirical approaches adequately addresses the assessment of multiple limiting nutrients. However, these eutrophication models, which are driven by three fundamental variables calculated from lake morphometry, water budget, and nutrient budget, are used as tools to describe the maximum nutrient loading desired for a particular waterbody in conjunction with the establishment of management goals and standards for the waterbody.

Budget Models. Again focused on P, these mass balance models attempt to simulate steady-state and long-term temporal trends of lakes (Chapra and Canale, 1991). These models are useful in providing a means to assess how lake sediments contribute to the eutrophication process. They are limited in that they do not simulate seasonal trends and do not address N.

Nutrient–Food Chain Models. These are comprehensive computer models that attempt to simulate seasonal trends of N, P, and the food chain (e.g., EUTROWASP [Eutrophication Water Analysis Simulation Program], QUAL-2E [Stream Water Quality Model]). They require extensive verification data and considerable expertise to operate.

Land–Water Models. These attempt to integrate the drainage basin with the receiving water. Examples include HSPF (Hydrologic Simulation Program–Fortran) and SWRRB (Simulator for Water Resources in Rural Basins).

These have more primitive representations of the receiving waters than the nutrient-food chain models. However, their integration of land and water in a single framework makes them appealing for nonpoint assessment.

A major deficiency in most of these models is their inability to discriminate between available and nonavailable forms of nutrients. Most of the correlations and the budget models focus on total nutrient. The nutrient-food chain models divide the nutrients into different forms. For example, P might be divided into inorganic, dissolved organic and particulate organic compartments, whereas N might be broken into nitrate, ammonia, and dissolved and particulate components. However, there are two flaws to this approach that might hamper these models in analyzing nonpoint-source runoff. First, they do not have compartments for unavailable P forms such as apatite; second, the models do not account for sorption of nutrients onto inorganic particles.

On the positive side, these models provide a considerable capability that could provide a basis for assessment of nonpoint-source controls. For example, as presently configured, they could adequately assess feedlot waste, which could be treated as a point source. In contrast, true diffuse sources such as field runoff would be handled somewhat less effectively. Additional research and development would have to be implemented to create a tool that would meet the specific needs of evaluating agriculturally derived P.

Factors Controlling Phosphorus Loss in Runoff

Forms of Runoff Phosphorus

The loss of P in runoff occurs in dissolved and particulate P forms. The standard procedure to separate dissolved P (DP) and particulate P (PP) in runoff is by filtration through a 0.45-μm pore diam. membrane filter. Dissolved P is comprised mostly of orthophosphate, which is immediately available for algal uptake (Walton and Lee, 1972). Particulate P includes P sorbed by soil particles and organic matter eroded during runoff and constitutes the major portion of P transported from conventionally tilled land (75–95%). Runoff from grass or forest land carries little sediment and is dominated by DP. Particulate P can provide a variable but long-term source of P to aquatic biota. Sharpley et al. (1992) found that from 10 to 90% of PP transported in runoff was bioavailable (BAP) and a function of watershed management. Bioavailable P was determined by extracting a sample with 0.1 *M* NaOH and includes DP and a portion of PP (Sharpley et al., 1991a).

Transport Mechanisms

The main mechanisms by which P is lost from agricultural land is by runoff and erosion. The first step in the movement of DP in runoff is the desorption, dissolution, and extraction of P from soil, crop residues, and surface-applied fertilizer and manure. These processes occur as rainfall interacts with a thin layer of surface soil before leaving a field as runoff (1–2.5 cm) (Sharpley, 1985a). Once P is dissolved in runoff water, sorption or desorption with runoff sediment may occur (Sharpley et al., 1981). The magnitude and direction of P transformation is dependent on the concentration of DP, PP, and sediment in runoff. In runoff from no till or pasture, the sediment load is generally so low that little sorption of DP occurs, and DP losses can exceed those in runoff from fields with higher erosion (Sharpley et al., 1992).

As erosion increases, the PP concentration of runoff increases. Sharpley et al. (1992) found that the relationship between erosion and PP was similar for both unfertilized grassland and fertilized conventionally tilled wheat (*Triticum aestivum* L.) fields, although soil and PP losses were approximately two orders of magnitude greater with cultivation (Fig. 2). In addition to surface runoff, a significant portion of DP can also be transported through subsurface flow as observed in sandy spodosols (Gilliam et al., 1994). Similarly, subsurface P losses can be significant in poorly drained soils high in organic matter, which are artificially drained, such as those in North Carolina Coastal Plains (Deal et al., 1986) and from Histosols (Izuno et al., 1991; Miller, 1979; Duxbury and Peverly, 1978).

Particulate P loss from soils is a complex process, determined both by the nature of rainfall events and the soil and management factors that affect runoff and erosion. During detachment and movement of soil in runoff waters, finer-sized soil fractions (e.g. clays, colloidal organic matter) are preferentially eroded. This results in eroded material having a higher content of P than source soil, referred to as *enrichment.* For example, the enrichment of STP and total P in runoff from several soils under simulated rainfall ranged from 1.2 to 6.0 and 1.2 to 2.5, respectively (Sharpley, 1985b).

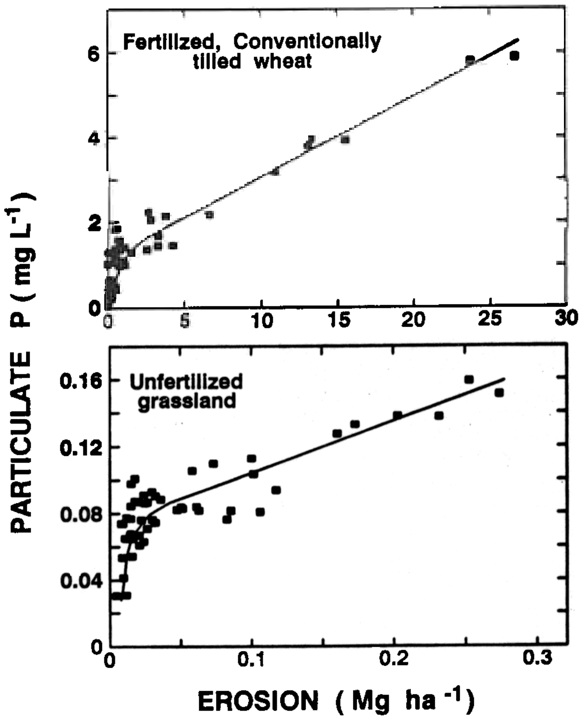

Fig. 2. Effect of erosion on the particulate P concentration of runoff from several Southern Plains watersheds (data adapted from Sharpley et al., 1992).

Source Management

Field research using simulated rainfall has established a relationship between P loss in runoff and rate and method of P application. An increase in P loss in runoff with increasing application rate of fertilizer (Romkens and Nelson, 1974), dairy manure (Mueller et al., 1984), poultry litter (Edwards and Daniel, 1993; Westerman et al., 1983), and swine manure (Edwards and Daniel, 1994) has been reported. It was also demonstrated that the DP concentration of runoff from areas receiving broadcast fertilizer P averaged 100 times more than from areas where comparable rates were applied 5 cm below the soil surface (Baker and Laflen, 1982). Mueller et al. (1984) showed that incorporation of dairy manure reduced TP loss in runoff fivefold compared with areas receiving broadcast applications.

A more difficult management problem is the ubiquitous contribution of surface soil P in runoff, particularly where STP has become elevated. Several studies have reported that the loss of DP in runoff is dependent on the STP content of surface soil. For example, a highly significant linear relationship was obtained between the DP concentration of runoff and STP content (Mehlich-3) of surface soil (5 cm) from cropped and grassed watersheds in Arkansas and Oklahoma (Fig. 3). A similar dependence of the DP concentration of runoff on Bray-1 P was found by Romkens and Nelson (1974) for a Russell silt loam in Illinois ($r^2 = 0.81$), on 0.1 M NaCl extractable soil P ($r^2 = 0.98$) of a Tokomaru silt loam in New Zealand by Sharpley et al. (1978), and on the water-extractable soil P content of several Mississippi ($r^2 = 0.61$; Schreiber, 1988) and Oklahoma watersheds ($r^2 = 0.88$; Olness et al., 1975).

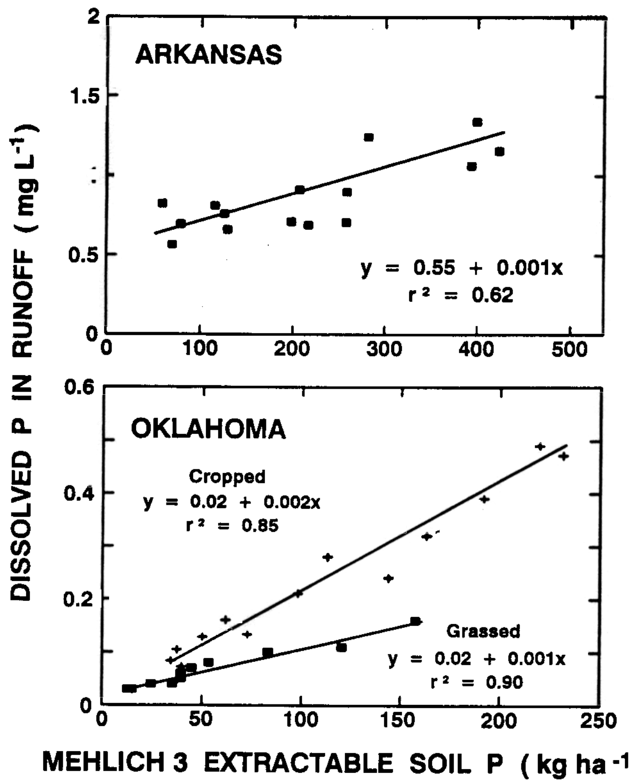

Fig. 3. Effect of soil test P on the dissolved P concentration of runoff from several watersheds in Arkansas and Oklahoma.

Vaithiyanathan and Correll (1992) observed that the loss of P in runoff from forested and cropped watersheds in the Atlantic coastal plains was closely related to soil P content ($r^2 = 0.96$). Other studies have also demonstrated the close dependence of P loss in runoff on surface soil P content (Barisas et al., 1978; Reddy et al., 1978).

Timing of P application relative to the occurrence of intense runoff events is an overlooked factor in management programs that seek to limit P loss in runoff. The major portion of annual P loss in runoff generally results from one or two intense storms. If P applications are made during periods of the year when intense storms are likely, then the percentage of applied P lost would be higher than if applications are made when runoff probabilities are lower (Edwards et al., 1992). Using watershed studies, Burwell et al. (1975) demonstrated that runoff P loss was greatest during the planting season; a time of intense rains, high P application, and minimum crop cover.

The length of time between applying P and the first runoff event also influences P loss, especially in situations involving manure. When simulated runoff was delayed from 1 h to 3 d, Westerman and Overcash (1980) found a 90% reduction in P loss after poultry or swine manure was applied. This reduction in P loss was attributed to increased time for P sorption. Unfortunately, some farmers apply manure to the surface of soils during fall and winter months, when more time is available for this operation or frozen conditions results in less physical damage to soils by application equipment. As the manure is not incorporated and plants are not growing, sorption and plant uptake do not occur, and the potential for P loss during spring rainfall events is increased.

Soil Phosphorus Levels

Decades of P fertilization at rates exceeding the amount removed by crops have resulted in widespread increases in STP levels. For example, an average of 48 mg kg^{-1} of STP (Bray-1) was present in all soils tested in Wisconsin in 1990, compared with 34 mg kg^{-1} in 1967. Coarse-textured soils, reflecting their extensive use in vegetable production, had an even higher average of 72 mg kg^{-1} of STP (Combs and Burlington, 1992). Similarly, the University of Delaware began recording STP values (Mehlich-1) in 1991. Results showed 68% of the samples for commercial crops tested high or very high. Of the very high samples, 64% ranged from 67 to 134 mg P kg^{-1}, and 20% from 135 to 201 mg P kg^{-1}; the threshold STP value in Delaware is 35 mg P kg^{-1}.

Crop yield response to STP is illustrated in Fig. 4, which conceptualizes the process of making fertilizer recommendations based on STP. As discussed earlier, long-term P applications can raise STP above levels required for optimum crop yields. Once STP levels exceed crop P requirements, the potential for P loss if runoff and erosion occur is greater than any agronomic benefits from further P applications.

While the long-term use of commercial fertilizer is primarily responsible for elevated STP levels in the top por-

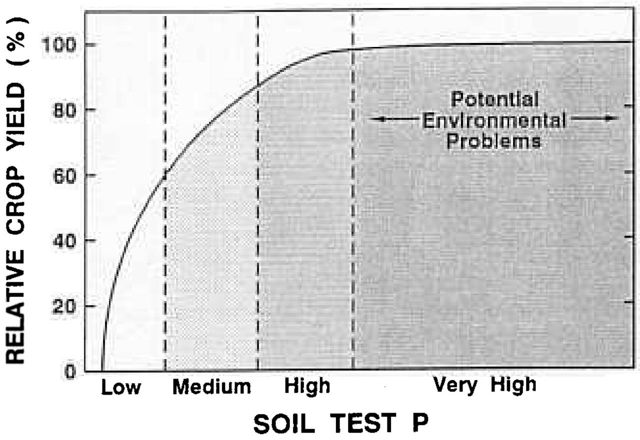

Fig. 4. Relationship between crop yield, soil test P, and the potential for environmental problems due to very high soil test P levels.

tion of soil profiles (0–15 cm), recent changes in farming practices have played a role in increasing the level of STP in the runoff-sensitive portion of the soil surface (1–2.5 cm). Mozaffari and Sims (1994) found that STP (Mehlich-1) values averaged 192, 174, and 56 mg P kg^{-1} in the 0- to 5-, 5- to 20-, and 20- to 40-cm depths of 18 cultivated fields in southern Delaware; values for the same depths in field border areas were 70, 39, and 17 mg P kg^{-1}.

Conservation tillage can increase the STP content of surface soil, if P is broadcast without soil profile inversion. In a long-term tillage study, Griffith et al. (1977) demonstrated the typical stratification of STP level under no-till conditions. Within a few years, the surface layer of soil under no till was six times higher than initial STP levels. In a similar study, Guertal et al. (1991) reported that STP (Bray-1) values in the upper 2 cm of long-term, no-tilled soils were 200 and 290 mg kg^{-1}; values for STP at the 6- to 8-cm depth of the same soils were 60 and 110 mg kg^{-1}.

Levels of STP are also elevated by long-term application of manures and wastes. Application of dairy manure has contributed to 200 mg kg^{-1} STP (Bray-1) levels in Wisconsin (Motschall and Daniel, 1982), and Pierzynski et al. (1990) found levels of 613 mg kg^{-1} of STP (Bray-1) in Illinois as a result of sludge additions. Sharpley et al. (1991b) examined several Oklahoma soils receiving long-term application of poultry litter and found STP (Bray-1) levels of up to 279 mg kg^{-1}.

In many cases, the problem of elevated STP levels has been exacerbated by the fact that manure application rates have been N-based, considering only soil N content and crop N requirements. Also, until recently, because of the time and expense of multiple dilutions and reanalysis of extracts required, many soil testing laboratories did not measure the actual value of STP, once it exceeded an upper limit that was clearly adequate for crop response. Thus, landowners and individuals involved in soil testing and fertilizer management often did not realize exactly how high STP was in many soils. However, with the development of more sophisticated laboratory equipment, such as inductively coupled plasma spectrometers and automatic dilutors, most soil test laboratories can now readily determine and report actual STP values.

After high levels of STP have been attained, considerable time is required for significant depletion. For example, McCollum (1991) estimated that, without further P addition, 16 to 18 yr of cropping corn (*Zea mays* L.) or soybean [*Glycine max* (L.) Merr.] would be needed to deplete STP (Mehlich-3) in a Portsmouth soil from 100 mg P kg^{-1} to the threshold agronomic level of 20 mg P kg^{-1}.

When McCollum's results are compared with data from other states, the magnitude of the problem of high STP levels becomes apparent. For example, the Mehlich-3 soil test has been shown to extract from 1.5 to 2.0 times as much P as the Mehlich-1 and approximately the same amount of P as Bray-1 (Sims, 1989). Because 40% of the commercial soil samples in Delaware in 1991 exceeded 67 mg P kg^{-1} (Mehlich-1), or approximately 120 mg P kg^{-1} (Mehlich-3), it could be several decades before STP levels in a significant percentage of the state's land area approach the current agronomic threshold value (35 mg P kg^{-1}). Phosphorus loading from dairy operations in Lake Okeechobee Basin, Florida, has resulted in significant accumulation of P. In some cases, P accumulation was about 50 times that of unimpacted areas (Graetz and Nair, 1994). Similarly, continuous discharge of P from agricultural area into low nutrient Everglades has resulted in distinct P gradient in the soil as function of distance from inflow (Reddy et al., 1993).

Environmental Soil Testing for Phosphorus

Many farmers rely on soil testing for P, or other nutrients, to rapidly provide them with an indication of the likelihood of an economic return on a fertilizer investment. Farmers assume that soil test values are accurate and have been derived from field calibration studies relating STP to crop yield response. Soil tests also provide farmers, and those that advise farmers, with a means to evaluate the efficiency of their nutrient management programs. However, the large percentages of agricultural soils testing high or very high in P in some areas of the USA (Fig. 1), suggests that soil testing has not been successful at avoiding the buildup of STP to levels that may be of concern for nonpoint-source pollution of surface waters.

Soil testing alone cannot assess the potential for soil P from an individual site or watershed to play a significant role in nonpoint-source pollution. A more comprehensive approach, such as the "Phosphorus Index System" (discussed in more detail in the options section of this article) proposed by the U.S. Soil Conservation Service (Lemunyon and Gilbert, 1993), is needed. The P Index is a field-oriented matrix system that integrates soil P availability, fertilizer and organic manure and waste management, and transport phenomena (erosion, runoff) to rank sites within a given watershed, in terms of their potential to deliver excessive P to surface waters.

A vital component of the P index is the STP value of a site. Soils with extremely high STP levels are logically presumed to be more at risk and require more intensive management. It is appropriate, however, to ask several questions about the use of STP in an index system of this nature. Perhaps most important, do routine soil test extractants, designed to assess plant availability of P, measure the forms of soil P most important to eutrophication? If not, are other types of soil tests for P available and perhaps more appropriate? In a broader sense, can soil testing laboratories play a more comprehensive role, and provide additional analytical or interpretive information to facilitate use of the P index?

Many of the issues related to environmental soil testing for P were summarized recently by Sims (1993). However, the most central question at this time is the need to clearly identify STP levels that are very high and represent situations where only the amount of P that can be removed in crop harvest be applied, because of potential environmental impacts. This situation will be most common where organic manures or wastes are frequently applied.

For years, many soil testing laboratories have recommended no P fertilizers be applied to high P soils, perhaps with the exception of a small amount in a starter fertilizer. However, the economic implications of P-based land application programs for animal-based agricultural and municipalities with sludge disposal responsibilities are highly significant. Several states (e.g., AR, OK, OH, MI, MS, PA, WI) have attempted to identify a soil test level where no P from manures or fertilizers is recommended beyond that which would be removed from the field in the harvested portion of the crop. In many situations, this would require that no manure or sewage sludge be applied and that alternative end-uses be developed. This is a clear example of the need for an integrated approach to the use of soil testing results.

Because environmental concern regarding soil P is eutrophication, the potential for P transport to surface waters must be considered, as well as the bioavailability of sediment and soluble P in runoff. Sampling protocol must also be re-evaluated, as the zone of interaction of runoff waters with most soils is normally less than 5 cm, while most soil samples submitted to soil testing laboratories are obtained from 0 to 20 cm. Given the lack of mobility of P in soils, the soil surface may be highly enriched in P, relative to the entire topsoil, particularly with long-term P applications and reduced tillage situations, where P is rarely incorporated into the soil. The most appropriate extractant to assess the biological availability of P is also of concern, as this is the fraction of soil P most likely to induce surface water eutrophication.

Manure Management

Developing manure management plans that are agronomically, economically, and environmentally sound is a challenge for an increasing number of landowners. In areas where a large number of confined animal producers are located, the amount of nutrients in manure often exceeds local crop requirements and area of land available for application. Thus, the main issues facing efficient utilization of P in manures involve limited area of land available for application, transportation of manure to nonproducing areas, and basing manure applications on P or N.

While row-crop production, as in the Corn Belt, can result in contamination of surface waters with sediments, nutrients, and pesticides (Hussein and Laflen, 1982), Duda and Finan (1983) showed that the greatest potential for accelerated eutrophication occurs in geographic regions of intensive animal production. Vast amounts of dairy manure are produced in states such as Wisconsin, California, Florida, and Texas. Swine manure is produced throughout the Midwest, especially in Iowa and Indiana. Poultry production is the major agricultural industry in Alabama, Arkansas, Georgia, and North Carolina. In Arkansas alone it is a $2 billion industry, producing more than 1 million Mg of broiler manure a year (NASS, 1991).

Regions that coincide with intense animal manure production are especially susceptible to eutrophication for several reasons. Efficiency of operation requires confinement of large numbers of animal units and ultimately the production of vast volumes of manure. Generally, the manure is land-applied as a means of disposal and application rates are N-based with little consideration given to the potential for P to promote eutrophication. For example, if manure is used to meet the N needs for fescue production in northwest Arkansas, an excess of 40, 37, and 17 kg ha^{-1} of P will be applied using either poultry, swine, or dairy manure, respectively (Huneycutt et al., 1988; ASAE, 1991;USDA-SCS, 1992). The P excesses will be even greater if application rates are adjusted for N losses such as volatilization. Thus, the inherent characteristics of animal manures and nutrient uptake by crops promotes P build-up with a corresponding increase in potential P loss.

Land Application of Manure

Most manures are bulky due to liquid volume or incorporated bedding material. Thus, manures have a lower nutrient content than mineral fertilizers (Table 2). As a result, much more manure than commercial fertilizer must be applied to achieve similar nutrient additions. However, the cost of transporting low-density manure more than short distances from the site of its production, exceeds its nutrient value. This has limited the land area available for application of manure. Consequently, most manure is applied in the immediate vicinity of production. Thus, the dominant geology, soils, and topography of the local area often cannot be considered prior to application. This inflexibility may result in application of manure to areas less suited, in terms of elevated STP contents from previous applications and high runoff or leaching potentials, than more distant areas. Unless an infrastructure is developed that can process, market, and distribute manure in nutrient-deficient

Table 2. Average P, N, and K contents (dry weight basis) of animal manures (data adapted from Gilbertson et al., 1979).

Animal	P	N	K
		g kg^{-1}	
Beef	5.6	32.5	2.6
Dairy	11.7	39.6	2.5
Poultry layers	20.8	49.0	2.1
Poultry broilers	16.9	40.0	1.9
Sheep	10.3	44.4	3.1
Swine	17.6	76.2	2.6
Turkeys	16.5	59.6	1.9

areas, this inflexibility will result in continued increases in STP on soils that are sensitive to runoff and erosion.

The limited area of land available for manure application is exacerbated in many parts of the country by the fact that the proliferation of the animal industry has been economically driven. The growth of confined animal operations on small farms has arisen from reduced returns on traditional grain or forage products, regulated price supports, policy changes, and the risk of crop failures. Struggling landowners have turned to confined animal production as a steady source of income with limited cash outlay. In many areas of the southern USA, intensive poultry production has replaced cash crop production. Agricultural lands in these areas are unable to maintain high crop yields, due to erratic weather, sloping topography, and soils that are shallow, coarse-textured, or permeable. Therefore, the local need for P and N additions for crop production will be lower than in areas of intensive crop or forage production.

Further exacerbating this situation, insofar as the poultry and swine industry is concerned, is the fact that many of these farms are part-time operations located on an inadequate land base. Growers often have full-time jobs in nearby businesses, several poultry houses located on small areas (e.g., <5 ha), and little expertise in (or time for) waste management. This again illustrates the need for education and extension programs as well as an infrastructure that can collect, process, and redistribute manures to areas with high local demand for the P and N.

Strategies for Determining Manure Applications

Historically, strategies for application of animal manure have been based on meeting the N needs of the crop to minimize nitrate losses by leaching and the potential for groundwater contamination. In most cases, this strategy has led to an increase in STP levels in excess of crop requirements, due to the generally lower ratio of N/P added in manure than taken up by crops. The animal manures listed in Table 2 have an average N/P ratio of 4, while the N/P requirement of major grain and hay crops is 8 (White and Collins, 1982).

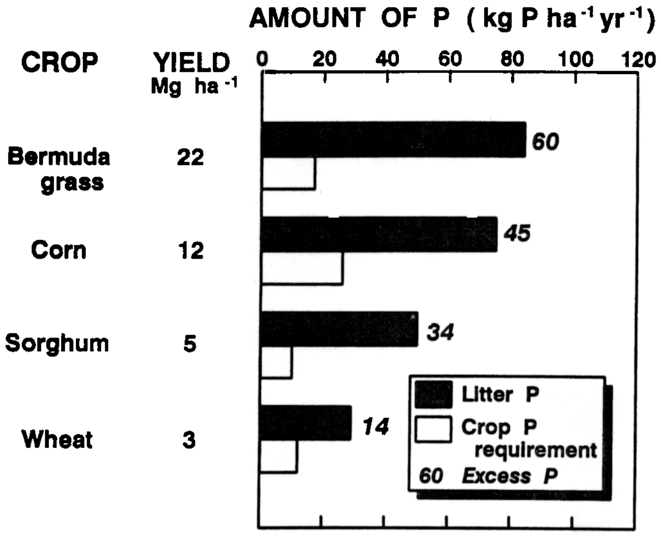

Fig. 5. Amount of P added in poultry litter compared with crop P requirements, if litter application rates are determined by crop N requirements.

The potential for surface soil accumulation of P is illustrated in Fig. 5. If the N requirement of several crops is met by poultry litter application, the amount of P added in excess of annual crop P uptake ranges from 14 to 60 kg P ha^{-1} (Fig. 5). Similarly, potential fertilizer and manure P inputs to intensive crop and livestock production areas of the Po region of Italy are 40 to 60 kg P ha^{-1} yr^{-1} and in central and southern Netherlands are 100 to 200 kg P ha^{-1} yr^{-1} (Breeuwsma and Silva, 1992). These inputs are much greater than average annual crop removal rates of 28 kg ha^{-1} yr^{-1} for these areas.

A P driven manure management program may mitigate the excessive build up of soil P and at the same time lower the risk for nitrate leaching to groundwater. However, basing manure applications on P rather than N management, could present obstacles to many landowners. A STP-based strategy would eliminate much of the land area with a history of continual manure application, as many years are required to lower STP levels once they become very high. This would force landowners to identify larger areas of land to utilize the generated manure, further exacerbating the problem of local land area limitations.

In addition, landowners relying on manure to supply most of their crop N requirements may be forced to buy commercial fertilizer N, to supplement manure N not used. Using a STP-based strategy may resolve potential environmental issues, but could place additional economic burdens on landowners. Clearly, the development of environmentally sound management systems utilizing P originating from manure, is a challenge from both agronomic and economic standpoints. Several BMPs are currently being implemented or evaluated in areas north of Lake Okeechobee, Florida, where large number of dairies are present. Some strategies are (i) management of application rates based on soil test values, (ii) manage cattle grazing density, (iii) feed ration management, (iv) limit cattle access to flow-ways, and (v) create buffer areas near flow-ways (Bottcher and Tremwell, 1994).

OPTIONS

Minimizing Phosphorus Loss to Runoff

Phosphorus loss from agricultural land can be minimized by erosion and runoff control and P-source management. Loss by erosion and runoff may be reduced by increasing vegetative cover through conservation tillage (Fig. 6). However, DP and BAP losses can be greater from no till than from conventional till practices. Accumulation of crop residues and added P at the soil surface provides a source of P to runoff that would be decreased during tillage. Such water quality tradeoffs must be weighed against the potential benefits of conservation measures in assessing their effectiveness.

Additional measures to minimize P loss by erosion and runoff include buffer strips, riparian zones, terracing, contour tillage, cover crops, and impoundments or small reservoirs. However, these practices are generally more efficient at reducing PP than DP. For example, several studies have indicated little decrease in lake productivity with reduced P inputs following implementation of conservation measures (Gray and Kirkland, 1986; Young and DePinto, 1982). The lack of biological response was attributed to an increased bioavailability of P entering the lakes as well as internal recycling. Clearly, effective remedial strategies must address the management of P sources and applications as well as erosion and runoff control.

Efficient management of P sources on soils susceptible to P loss involves fertilizer placement and the use of STP recommendations based on eutrophic rather than agronomic considerations to determine P application rates. Wherever possible, subsurface placement of P away from the zone of removal in runoff will reduce the potential for P loss. It may also be necessary to periodically plow no-till soils to redistribute surface P accumulations throughout the root zone. Both practices may indirectly reduce the loss of P by increasing crop uptake of P and yield, which affords a greater vegetative protection of surface soil from erosion.

However, conflicts within BMPs between Soil Conservation Service residue management guidelines and recommended subsurface applications of P may exist. In compliance with residue conservation programs, landowners may be required to maintain a 30% residue ground cover. Under this BMP, subsurface application or knifing of P fertilizer or manure, which may be recommended to minimize P loss in runoff, could be unacceptable if it reduces residue cover below 30%. Thus, BMPs should be flexible enough to enable modified residue and P management plans to be compatible.

Although P losses in runoff are generally <5% of applied P, DP, and TP concentrations often exceed critical values associated with accelerated eutrophication (0.05 and 0.1 mg L^{-1}; USEPA, 1976; Vollenweider and Kerekes, 1980). This is true even for unfertilized native grass watersheds (Sharpley et al., 1986). Also, P inputs in rainfall can contribute to freshwater eutrophication (Lee, 1973; Schindler, 1977). Thus, the above recommended measures may not reduce P loss in runoff from cultivated land to critical values. This emphasizes the need to target remedial measures on source areas where the potential for P loss is greatest. Further, the critical level approach should not be used as the sole criterion in quantifying permissible tolerance levels of P loss in runoff as a result of differing management. A more flexible approach advocated by limnologists considers the complex relationships between P concentration and physical characteristics of affected watersheds (runoff and erosion) and water body (mean depth and hydraulic residence time) on a site specific and recognized need basis.

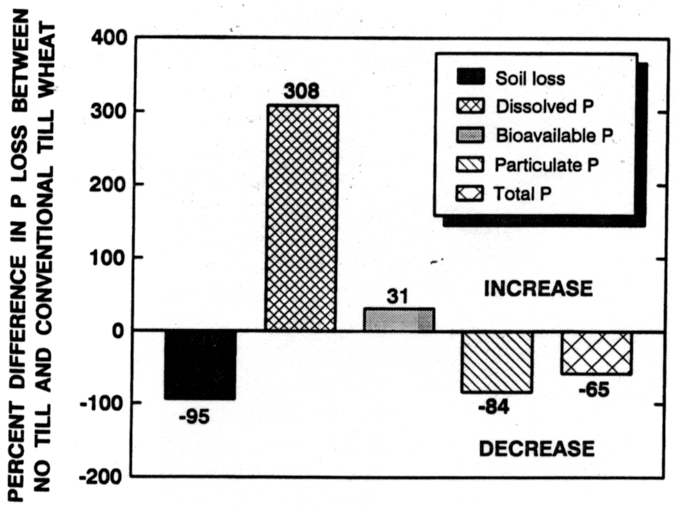

Fig. 6. Percent difference in soil and P loss in runoff from no-till and conventional-till wheat in the Southern Plains.

Environmental Soil Testing for Phosphorus

By necessity, soil testing programs must measure P through the use of rapid chemical extraction procedures if they are to provide recommendations in a timely and cost-effective manner. From an environmental perspective, however, other tests for soil P may be more appropriate. For example, if the environmental issue is eutrophication, an assessment of the desorption of soil (or sediment) P and subsequent bioavailability to aquatic organisms such as algae is of greatest interest. Conversely, for a wastewater irrigation system, estimates of the long-term capacity of a soil profile to retain P against leaching will be needed. While it is unrealistic to expect that routine soil tests can provide the information needed for all environmental management programs, recent research has shown that STP is well correlated with several parameters needed to assess nonpoint-source pollution. Additionally, a number of alternative tests for soil P are available that, while not as easily conducted as a routine soil test, can provide supplemental information on P bioavailability, desorption, and sorption.

Although the loss of P in runoff is dependent on surface soil P content (Fig. 3), watershed variability in runoff and erosion processes and soil management and topographic factors render such relationships site specific. Even so, current soil test information can be used along with soil series and geomorphological information to estimate the potential for P loss in runoff. Soil scientists must get involved in such decision-making processes or else others will, who have less knowledge of soils and soil testing. However, approaches must be developed to more reliably determine the potential for P-related environmental problems to occur with agricultural management.

Biologically available P has been operationally defined as "... the amount of inorganic P, a P-deficient algal population can utilize over a period of 24 h or longer" (Sonzogni et al., 1982). Algal uptake of P has been shown to be closely related to amounts of P extracted from soils or sediment by iron oxide–impregnated paper strips (Sharpley, 1993). While the Fe-oxide strip procedure presents some difficulties for soil testing laboratories because of the extraction time (16 h) and need for a two-step extraction to remove P sorbed on the strip, other research has shown that routine soil tests are well correlated with this measure of BAP. Consequently, in areas where nonpoint-source pollution by BAP in agricultural runoff is important in eutrophication of surface waters, soil-testing laboratories could use routine soil tests to provide preliminary rankings of the BAP level of soil (or sediment) and use Fe-oxide strips as a special test for more intensive management programs.

Wolf et al. (1985), also demonstrated that routine soil tests were related to the equilibrium P concentration at zero sorption (EPC_0) value of soils. Soils with high EPC_0 values have a greater tendency to desorb dissolved P into runoff waters. This is another example of the use of a routine soil test as a surrogate for a more extensive procedure that is not well adapted to a soil testing laboratory.

The potential for soils to sorb P is also important in the design of wastewater irrigation systems, or in areas where leaching and lateral flow of P in drainage waters may be important. The long-term capacity of soils to retain P is commonly estimated by adsorption isotherms that can be used to derive adsorption maxima for soil horizons. As in the determination of EPC_0 values, these isotherms require equilibration of soil with a series of P solutions of increasing P concentration, normally for 24 h, and are not well adapted to routine soil testing laboratories. Bache and Williams (1971), however, suggested that a single-point index could be used to estimate the P sorption maxima of soils with reasonable accuracy. This was recently confirmed by Mozaffari and Sims (1994) for surface and subsoil horizons of four Atlantic Coastal Plain soils.

Current soil test methods represent, for the most part, plant-available P levels in soil. Because of the high organic P content of manure, soil test recommendations must account for mineralization of organic P during the growing season. In addition, manure can provide plant-available P for several years after application. Thus, soil tests must also give credit to the residual effects of manure, possibly resulting in a reduction in application rates in years following initial applications.

In summary, advances in the interpretation of soil test P results for environmental problems will require continued innovation. It is essential that the long-term, proactive role of soil testing laboratories in the development of sampling procedures, analytical methods, and practical recommendations for efficient P management for crop production be applied to environmental management of soil P. Soil testing programs have a unique opportunity to coordinate the efforts of many of the participants needed to develop effective, environmentally sound P management programs.

For example, soil test laboratories could report STP levels that exceed crop P requirements as excessive rather than high or very high. Farmers receiving a report saying excessive STP levels are present, are more likely to stop or reduce fertilizer P additions than a report saying levels are very high.

In many states, the traditional interactions of soil testing laboratories with university-based research and extension soil scientists and agronomists have expanded into closer working relationships with advisory agencies (Soil Conservation Service, local conservation districts), crop and environmental consulting firms, and state and federal regulatory organizations. Soil testing programs, through sustained and creative efforts, can contribute greatly to the development of conceptually sound and technically feasible solutions to the complex problems of nonpoint-source pollution by soil P.

Manure Management

Animal manure can be a valuable resource integrated in cost-effective BMPs. In some areas, animal manure applied on hilly land has increased vegetative cover, thereby reducing runoff and erosion potential. Though these unproductive soils would not normally receive mineral fertilizer, the careful use of manure can reduce environmental degradation. However, greater amounts of P and N are often produced in manure than required by crops in the immediate vicinity. Under these situations, options for efficient utilization of manure can present greater challenges to landowners than mineral fertilizer management. Options include manure analysis, basing application rates on soil vulnerability, tillage operations available, and formation of cooperatives.

Manure Analysis

There are many variables associated with animal production systems that can affect manure quality at the time of application. These include the type and amount of bedding material used, accumulation time, amount and quality of water used to flush the house, location in a storage pit from which the waste is removed, and length of storage before land application. The variability in these management factors can result in a wide range in nutrient content of manure applied (Edwards and Daniel, 1992).

Farm advisors and extension agents in several states are recommending that the P and N content of both manure and soil be determined by soil test laboratories before land application of manure. There is also a tendency among landowners to underestimate the nutritive value of manure. Thus, manure analyses are a constructive educational tool showing landowners that manure represents a valuable source of P as well as N.

A cautionary note to basing application rates on manure analyses must be sounded because of the wide variability in nutrient contents that can be obtained. Igo et al. (1991) estimated N applied using analysis of stockpiled broiler litter on 176 farms and compared this with the actual N loading rate based on analysis of samples collected during application to field corn. When desired application rates were applied to large field plots using commercial manure spreaders, overapplication of 10 to 20 kg N Mg^{-1} of litter commonly occurred, as did underapplication of 5 to 10 kg N Mg^{-1}. Therefore, the accurate application of a recommended litter rate for corn ($\sim$5 Mg ha^{-1}), based on analysis of the litter, commonly resulted in the application of excess manure N approaching the total N requirement of the crop ($\sim$100 kg N ha^{-1}). Similar variabilities are associated with P, often ranging from 5 to 15 kg P Mg $manure^{-1}$. Thus, manure analyses should be conducted as soon as possible before any manure application and be used only as guidelines in determining application rates.

Manure Application Strategies

Because of the high P content of manure relative to N and rapid build up of STP with continual manure applications in excess of crop requirements, management strategies based on P rather than N will present greater restrictions to landowners. As many soils still have low or medium STP levels (Fig. 1), a P-based manure management program is not always appropriate. However, given the large excesses of P accumulating in many states and regions, a national P management strategy should be considered, where P produced in animal manures is redistributed to areas dominated by soils testing low or medium in P. Flexible management systems must consider several site characteristics to determine if manure application rates should be based on P or N. If STP is categorized as high or very high, then P should be the driving element.

Generally, N should be the priority management consideration until STP becomes high. At this time, the potential for contamination of surface waters with P must be considered, to determine what is the most important factor on which application rate must be based. For example, if runoff and erosion potential to P-sensitive waterbodies far exceeds the potential for N leaching, then P should be the main element driving application rates.

Preliminary research in Arkansas by Phillip Moore (USDA-ARS) indicates that poultry litter amendments, such as slaked lime or alum, can reduce P solubility and ammonia volatilization from the litter by several orders of magnitude. Thus, these amendments may have several beneficial effects on litter management. In addition, reducing in-house volatilization of ammonia will be of economic benefit to the farmer by increasing weight gains by the birds. By reducing the solubility of litter P, the transport of bioavailable DP, which accounts for up to 90% of P transported in runoff from litter-treated soils (Edwards and Daniel, 1994), may be reduced. A decrease in ammonia volatilization from in-house litter will increase litter N content. The resultant increase in N/P ratio of the litter will minimize potential accumulation of P in soils, to which litter is applied to meet crop N requirements. Also, the increased nutritive value may increase the distance amended litter can be economically transported, reducing the current severe local land area limitations facing many farmers.

Soil Tillage

Application of manure before or during tillage will reduce surface soil accumulation of P and increase its distribution in the root zone. If a ground cover can be maintained during times of the year when runoff-producing rainfall are most common, environmental risks will be reduced while crop utilization of P will be increased.

Use of tillage to control P losses, however, means that the time frame for manure application would be restricted to the time of tillage operations. As mentioned earlier, farmers face significant economic and labor-related constraints that often result in manure applications at undesirable times of the year from an environmental perspective.

The use of manure on grassland without tillage can be reasonably efficient, especially in areas with a humid climate (Tveitnes, 1979). This is probably because grass species can utilize manure throughout the whole growing season. Crop type and yield will affect the amount of P removed from the production systems, if the crop is harvested. Obviously, the accumulation of P within an agricultural system will be reduced if it is removed from the farm in the harvested crop.

Economic Options

One of the major obstacles facing more landowners is overcoming the economic restrictions of moving manure to a greater land area, where it could supplement or even replace mineral fertilizer requirements. The recent trend in the formation of cooperatives that can more cost-effectively compost and compact manure should be encouraged by cost-sharing programs. Neighboring landowners and private industry are also developing manure-processing alternatives. Examples of this include centralized storage and distribution networks, regional composting facilities, and pelletizing operations that can produce a value-added processed manure for distribution to other areas. Pelletization is a particularly attractive option because it results in a dried, lightweight material that can be handled, transported, and applied in much the same manner as commercial fertilizer.

By composting and compacting, the bulk density of the manure is reduced, as is the cost of transportation. If the consumer is not willing to bear a part of the financial support, then it may be necessary to recommend producers and landowners to take part in cooperative manure treatment programs. The level of involvement could be linked to the number of animals per farm.

Storage of manure will allow more flexibility in timing applications. A wide range of storage methods and costs are available to landowners (Brodie and Carr, 1988). Inexpensive plastic sheeting can perform well with very low cost for some solid manures. However, all storage methods must be managed carefully to fully realize their potential in an agronomically and environmentally sound BMP.

Another economic option is the buying and selling of pollution credits within a given watershed, similar to that recently adopted for air quality control. Farmers able to limit P loss below recommended levels could sell credits to a farmer unable to meet these levels. The number of credits a farmer has could be linked to the number of animals and area of farm. As a result, P export from a watershed may be kept within predetermined limits by sharing the responsibility among farmers. Education and extension programs should highlight the nutritive and mulching value of manure to nonproducing farmers. In effect, increasing the demand for this nutrient resource.

Even so, it is clear that our current technology will not permit an unlimited number of animals in a region, without impacting water quality. Thus, it may be necessary to redistribute animals or to limit animal numbers within an area. Several states now require that new animal facilities that exceed a certain size have an appropriate waste management plan. Thus, it is essential that we develop and transfer technology to implement environmentally sound recommendations for manure management.

Prioritizing Watershed Management

Identification of Phosphorus-Sensitive Waters

While most freshwaters are P limited, there are notable exceptions where P controls will have marginal to no benefit. To optimize control activities there is, therefore, often a need to prioritize management actions to those watersheds where the control of P will provide the greatest benefit. Secondly, management agencies are also often required to further target limited financial and human resources to those P-sensitive lakes having the highest public or ecosystem value.

Management of P inputs to lakes can be ineffective and may not be dictated in some instances. In some lakes and in many streams, plant productivity is limited by high turbidity either from anthropogenic or natural sources. In others, such as the high-elevation lakes in the western USA, N is limiting (Eilers, 1991). Uttormark and Hutchins (1978) highlighted the importance of lake flushing rates as a management factor, noting that those lakes that had flushing rates (*flushing rate* is the number of lake volumes per year that are replaced by inflowing water) higher than 6 times per year generally did not exhibit eutrophication problems, because of the continual washout of plants that require quiescent conditions to utilize incoming nutrients. Shallow lakes, i.e., those that do not stratify, also tend to exhibit different characteristics than deeper lakes and often respond less directly to P inputs. Generally, watershed P control strategies will have the greatest benefit when applied to the deeper, stratified lakes having low flushing rates where natural background P concentrations are not so high as to limit the effectiveness of the control of anthropogenic sources.

Targeting Source Areas

Strategies to minimize P loss in runoff will be most effective if sensitive or source areas within a watershed are identified, rather than implementation of general strategies over a broad area. Long-term field studies that reliably evaluate P movement are costly, lengthy, and labor intensive. Also, use of models simulating the effect of agricultural management on P loss in runoff often require detailed soil information and computer experience to run them. Thus, a P indexing system was developed to identify soils vulnerable to P loss in runoff (Lemunyon and Gilbert, 1993).

The index is outlined in Tables 3 and 4. Each site characteristic affecting P loss is arbitrarily assigned a weight, assuming that certain characteristics have a relatively greater effect on potential P loss than others. The P loss potential is given a value (Table 3), although each user must establish a range of values for different geographic areas. An assessment of site vulnerability to P loss in runoff is made by selecting the rating value for each site characteristic from the P index (Table 3). Each rating is multiplied by the appropriate weighing factor. Weighted values of all site characteristics are summed and site vulnerability obtained from Table 4.

A hypothetical site is used as an example, where soil erosion is 15 Mg ha^{-1} (weighting × value; 1.5 × 2 = 3), runoff class is high (0.5 × 4 = 2), soil test P is high (1 × 4 = 4), 50 kg P ha^{-1} of fertilizer (0.75 × 4 = 3), and 50 kg P ha^{-1} as animal manure (0.5 × 8 = 4) are broadcast in early spring prior to planting (1 × 4 = 4; 1 × 4 = 4). The sum of these weighted values (3, 2, 4, 3, 4, 4, and 4) is 24, which has a high site vulnerability (Table 4). In this hypothetical situation, conservation measures to minimize erosion and runoff as well as a P management plan should be implemented to reduce the risk of P movement and probable water quality degradation.

The index is intended for use as a tool for field personnel to easily identify agricultural areas or practices that have the greatest potential to accelerate eutrophication. It is intended that the index will identify management op-

Table 3. The P indexing system to rate the potential P loss runoff from site characteristics.†

Site characteristic (weight)	Phosphorus loss potential (Value)				
	None (0)	Low (1)	Medium (2)	High (4)	Very high (8)
			Transport factors		
Soil erosion (1.5)¶	Negligible	<10	10–20	20–30	>30
Runoff class (0.5)	Negligible	Very low or low	Medium	High	Very high
			Phosphorus source factors		
Soil P test (1.0)	Negligible	Low	Medium	High	Very high
P fertilizer application rate (0.75)§	None applied	1–15	16–45	46–75	>75
P fertilizer application method (0.5)	None applied	Placed deeper than 5 cm	Incorporated immediately before crop	Incorporated >3 mo before crop or surface applied <3 mo before crop	Surface applied >3 mo before crop
Organic P source application rate (0.5)§	None applied	1–15	16–30	31–45	>45
Organic P source application method (1.0)	None	Injected deeper than 5 cm	Incorporated immediately before crop	Incorporated >3 mo before crop or surface applied <3 mo before crop	Surface applied >3 mo before crop

† The P indexing system was developed by the following team of scientists; J. Lemunyon, D. Goss, G. Gilbert, J. Kimble, T. Sobecki, USDA-SCS; A. Sharpley, USDA-ARS; T. Daniel, Univ. Arkansas; T. Logan, Ohio State Univ.; G. Pierzynski, Kansas State Univ.; T. Sims, Univ. Delaware; and R. Steven, Washington State Univ.

¶ Units for soil erosion are Mg ha^{-1}

§ Units for P application are kg P ha^{-1}.

tions available to land users that will allow them flexibility in developing control strategies.

Targeting Watershed Management

While many states are pursuing some statewide control of nonpoint sources, such as construction site erosion, much attention is being placed on the concept of targeting controls of nonpoint sources using a watershed-based approach (USEPA, 1993). Recognizing the magnitude of agricultural related nonpoint-source problems, Water Quality 2000, a coalition of 80 environmental scientific, professional, governmental, and academic organizations, highlighted the pressing need to target limited financial and technical resources at the most critical and valuable water resources and to impacted watersheds, with additional priority directed toward projects having strong local support (Water Quality 2000, 1992). Some states have also targeted management of nonpoint sources on the basis of individual watershed and waterbody needs.

To select watersheds for implementation of comprehensive nonpoint-source controls, the Wisconsin Department of Natural Resources uses five criteria (Wisconsin Dep. of Natural Resour., 1986). Priority lakes and watersheds are selected based on the following factors:

1. The water quality impairment or threat to the use of the lake, stream, groundwater, wetland, or any other water of the state and the practicability of alleviating the impairment or threat.
2. The practicability of achieving a significant reduction in the amount of pollutants from the nonpoint sources in the watershed.
3. The public use of the watershed's lakes, streams, groundwater and other waters of the state.
4. The capability of the governmental unit to carry out the project considering commitments to ongoing projects.
5. Unique or endangered environmental resources.

Additionally, lakes targeted first for P watershed controls are those especially sensitive to P loadings and which have high public or environmental values. Phosphorus-sensitive lakes are those which:

1. Are greater than 10 ha (25 acres) in size.
2. Stratify during the summer under normal conditions.
3. Have water flushing rates of less than six times per year.

CONCLUSIONS

The accelerated eutrophication of most U.S. freshwaters is limited by P rather than N inputs. Nonpoint sources of P in agricultural runoff now contribute a greater portion of freshwater inputs, due to the easier identification and recent control of point sources. There are many complex and interdependent factors affecting the source, fate, and management of agricultural P in the environment. Thus, options available to landowners to minimize P-stimulated eutrophication of surface waters often require agronomic, economic, and environmental considerations and trade-offs. For example, conservation tillage may reduce total P losses in runoff but increase the loss of bioavailable P and occasionally nitrate leaching. Also, linking manure applications to P may reduce STP levels but pose economic burdens to some landowners having to transport manure greater distances and purchase N fertilizer to supplement crop N requirements.

Generally, the loss of agricultural P is not of economic importance to a farmer. However, it often leads to the deterioration of water quality from accelerated eutrophication, that can have significant off-site economic impacts. By the time these impacts are manifest, remedial strategies are often difficult and expensive to implement; they cross political and regional boundaries; and it can be several years or decades before an improvement in water quality occurs. Thus, identification of sources of P in runoff within a watershed or basin area is of prime importance in targeting cost-effective remedial strategies to minimize P loss. A P-indexing system to rank soils as to their vulnerability for P enrichment of runoff may provide a field tool to fill this need.

Once a water body has been identified as being sensitive to P inputs, source fields and soils vulnerable to P loss in runoff must be carefully managed. Options include recommending that further P applications be made on an environmental rather than agronomic basis. For soils with a high or very high STP level, options may involve applying no more P than removed annually by the crop. Clearly, unlimited P inputs and numbers of animals within an agricultural system, will lead to impaired water resources.

Fertilizer and manure applications based on environmental considerations to reduce potential P loss in runoff, have been practiced in many parts of Europe since the mid-1970s. After initial resistance to adoption of these guidelines, landowners are now aware of the need for this practice and are receptive to suggestions for its implementation. In the USA, soil scientists must get involved in the transfer of research information to the development of environmentally sound management guidelines. Otherwise, scientists with less knowledge of soils and agronomy will do the job.

Judicious applications of P can reduce P enrichment of agricultural runoff via increased crop uptake and vegetative cover. Nevertheless, it is of vital importance that we implement management practices that minimize STP build up in excess of crop requirements; utilize alternative P sources and residual soil P levels; and improve methods to identify soils capable of enriching bioavailable P loss in runoff. This will likely result in a decrease in agricultural P loss to surface waters. Otherwise, the perception by the public that agriculture cannot manage itself for the good of the environment will increase. Unfortunately, the benefit of remedial measures on water quality improvement will not be immediately visible to a concerned public. Consequently, future research and policy should emphasize the

Table 4. Site vulnerability to P loss as a function of total weighted rating values from the index matrix.

	Total index rating value
Low	<10
Medium	11–18
High	19–36
Very high	>36

long-term economic and environmental benefits of these measures.

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