Technical basis for quantifying phosphorus transport to surface and groundwaters[1]

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ABSTRACT: Decreasing the potential delivery of phosphorus in fertilizers or animal waste to surface and groundwater requires a knowledge of phosphorus's fate and transport mechanisms. The USDA Natural Resources Conservation Service has recently mandated each state to develop an assessment tool to estimate P transport to water bodies. The objective of this paper is to describe the processes involved with P transport to surface and groundwaters that must be accounted for in practical methods used to quantify the potential for P loss. Mechanistic models to assess P loss should account for: 1) P adsorbed to eroding sediments, 2) soluble P in runoff water, 3) soluble P in leaching water, and 4) P losses related to specific P sources. With sediment-bound P, the adsorbed P content in the eroded soil mass at the field edge must be quantified, whereas runoff volume and P concentration are needed to estimate soluble P loss in runoff water. Estimating P leaching potential requires calculation of drainage water volume and P concentration. When P is applied in animal waste, the specific source influences both soluble and particulate P loss because of differences in P solubility between waste types. In addition, the effects of conservation practices and other technologies on decreasing sediment, soluble, and leached P need to be included. Using these methods, a practical, quantitative P loss assessment tool can be developed that will enable technical service providers and other practitioners to estimate potential P loss and design best-management practices for land-applied waste management systems in order to minimize P transport to surface and groundwater.

Key Words: Environment, Erosion, Leaching, Manure, Runoff, Water Quality

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Introduction

Over the last several decades, agricultural sources of P have been suggested as a contributing factor to water quality degradation (U.S. EPA, 1996). Although industrial wastes, municipal wastes, and urban runoff contribute to P loading in certain watersheds, P applied directly to cropland through fertilizers and animal wastes can be contaminant sources of P in watersheds with predominately agricultural land uses (USDA-NRCS, 1994). Whereas fertilizer P use has increased slightly in the last decade (Terry and Kirby, 2000), substantial growth of confined animal feeding operations (**CAFO**) has occurred (USDA-NASS, 1999).

In general, <25% of P applied is recovered by the crop. With P applied at recommended rates, soil test P generally remains the same or increases slightly with time depending on P rate, soil type, and crop removal (Havlin et al., 1999). Because manure rates are typically based on crop N requirement, concomitant P rates are two- to fivefold greater than crop needs. Continued overapplication of P will increase soil test P and subsequent risk of P loss to surface and groundwater. In North Carolina, P applied in animal waste exceeded P requirements of all agronomic crops in nearly 40% of counties, representing about 85% of CAFO in the state. (Crouse et al., 2001)

Because of recent research results regarding P use and water quality, the USDA-NRCS revised the *Nutrient Management (590) Field Office Technical Guide* to reflect impacts of P use on water quality (USDA-NRCS, 1999). The revised standard required each state to develop a method to quantify potential P loss from agricultural fields.

The objective of this paper is to describe primary P transport mechanisms and factors essential to include in developing relatively simple and practical tools to quantify P loss in agricultural fields (Figure 1).

Phosphorus Loss Assessment

Potential P transport to surface and groundwater increases when waste P application rates exceed crop

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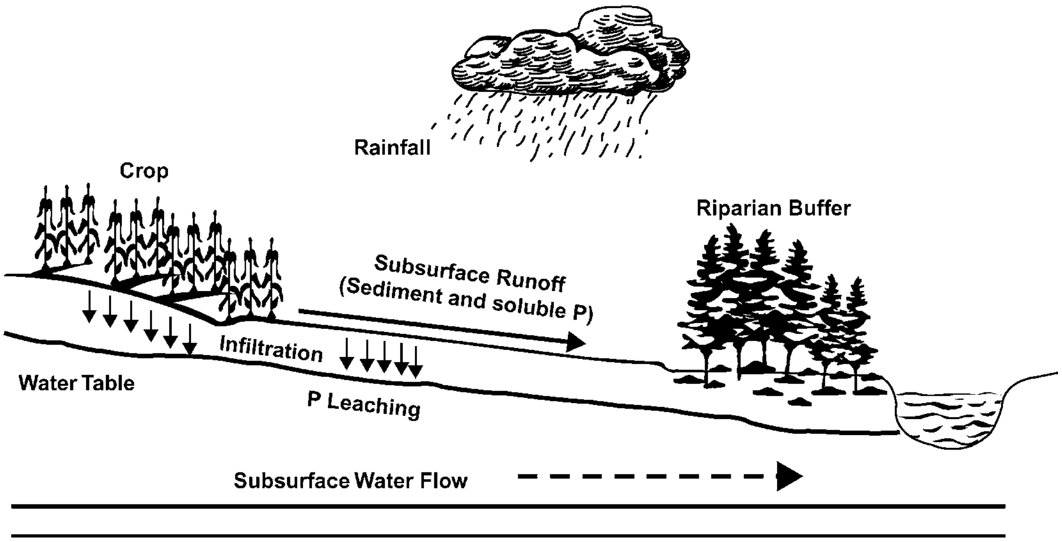

Figure 1. Diagram of major pathways contributing to P transport to surface and groundwater.

requirement (Sharpley et al., 1996; Sims et al., 2000). The primary factors responsible for potential P delivery from an agricultural field to surface or groundwater include 1) P adsorbed to eroding sediments, 2) soluble P in runoff water, 3) soluble P in leaching water, and 4) P losses related to the type of waste P applied. Practical P loss assessment methods or tools must include all four components to accurately assess the risk of P loss from agricultural fields (Lemunyon and Gilbert, 1993; Sharpley et al., 1996). Interestingly, the first P loss assessment tool provided by USDA-NRCS as a framework from which states could build individual P loss assessment tools did not include P leaching potential or waste source effects on P loss (USDA-NRCS, 1994). Under the revised USDA-NRCS standard, P application rates can be based on the following: 1) an agronomic interpretation of soil test P levels, 2) an environmental interpretation of soil test P, or 3) a P index (site-specific assessment of potential P delivery). In the case of using an agronomic soil test interpretation, the critical soil test P level represents that level above which crop yield is not increased with additional P applied, and, therefore, no fertilizer or manure P would be recommended. Using an environmental interpretation requires establishing the relationship between soil test P level and runoff P concentration. Soil test P levels that produce, for example, >1 mg/L dissolved P in runoff (or some other level), would be established for various soil types. Once this level is attained, no fertilizer or manure P would be recommended. The P index approach should assess the primary P loss pathways as influenced by soil properties, land forms, and management parameters and provides an estimate of the potential risk of P to water bodies (USDA-NRCS, 1994). Based on the P loss assessment, nutrient management plans would be adjusted to reduce P rates and the potential for P loss to surface and groundwater. Adjustments would include animal waste rates ranging from those entirely based on P requirements of the crop to cessation of waste applications. The most reliable method must account for all sources and mechanisms of P transport off a field to the edge of the water body.

While several states decided to use an agronomic or environmental P soil test interpretation to assess risk of P loss, most states recognized that this approach does not adequately reflect P transport, P source, and P management factors primarily contributing to P loss. Although more difficult to develop, a robust P index tool that accurately accounts for P transport pathways and inherent variability in environment, crops and cropping systems, soil types and management, and many other factors should provide accurate assessments of potential P delivery to surface and groundwater.

Several process-based models have been developed to quantify P transport (Sharpley et al., 2002). For example, AGNPS (Agricultural Nonpoint Pollution Source) assesses nutrient loading on watershed scales of >20,000 ha using individual runoff events (Young et al., 1995) or annual runoff (Croshley and Theurer, 1998). The Soil and Water Assessment Tool (**SWAT**) was developed to predict the impact of land and water management on sediment and chemical transport in large watersheds (Arnold et al., 1998). The SWAT uses daily time steps capable of 1- to 100-yr simulations. The Erosion-Productivity Impact Calculator (**EPIC**) has also been used to quantify sediment and nutrient transport (Sharpley and Williams, 1990). Although these and other process-based models have been adapted to assess P transport and loss on a field basis, all require detailed data on soil properties, hydrology, crop management, and many other parameters. However, the greatest limitation to adapting these models for use in a P index tool is the skill required by the user. The USDA-NRCS mandated that a P loss assessment tool was intended to be used by public or private

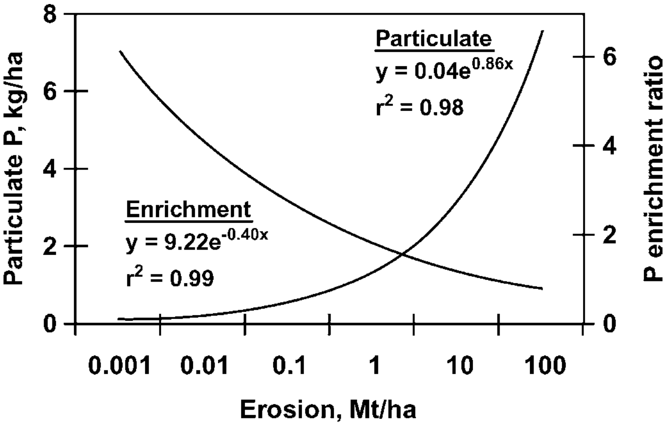

Figure 2. Effect of soil erosion rate on particulate P loss and P enrichment ratio.

county-level technical service providers on a field basis. Thus, states electing to develop a P index approach for P loss assessment must utilize routine methods and models that are well established to facilitate their use by practitioners.

Phosphorus Adsorbed to Sediments

Phosphorus adsorbed to soil clays and organic matter leaves the field during and immediately after storm events through sheet and rill erosion processes, and may constitute up to 90% of runoff P from cultivated fields (Sharpley et al., 1992). Sediment P is eventually redeposited downslope within the field, deposited in a riparian buffer or other sediment-trapping practice, or is delivered to a surface water body (Figure 1). The quantity of sediment P delivered to and beyond the field edge is a function of 1) soil erosion rate, 2) amount of sediment deposition within the field, and 3) the quantity of P adsorbed to the eroding soil particles. Beyond the field edge riparian buffers, sediment basins, and water control structures may reduce particulate P delivery to surface water.

Estimating Soil Erosion Rate

Soil erosion involves detachment, transport, and deposition of soil particles (Foster, 1982). Detachment occurs as a result of raindrop impact on and over the soil surface (Renard et al., 1997). Estimating sediment loss is essential because of its influence on particulate P loss (Figure 2). The Revised Universal Soil Loss Equation (**RUSLE**) is a reliable and accepted method to estimate average annual soil erosion rates within a field (Wischmeier and Smith, 1965; USDA-NRCS, 1995; Renard et al., 1997). The RUSLE is an empirical method that estimates the effects of rainfall energy or erosivity (quantity, intensity, and duration); soil erodibility related to selected soil physical and biological properties (clay and organic matter content, permeability, and soil structure); landform or topography (slope length and steepness); crop cover management (crop canopy and residue cover) as influenced by crop growth stage and tillage practices; and supporting practices (terraces, contour tillage, and contour cropping) on annual soil loss. Erosivity, erodibility, and slope steepness are factors that do not change with management, whereas,

Table 1. Soil and crop management factors that reduce annual estimated sediment loss

RUSLE parameter[a]	Conservation practice	Relative reduction[b]
Slope length (SL)	Terraces	High
	Vegetative filter strips[c]	Medium
Crop cover (CC)[d]	Permanent pasture	High
	No tillage (standing residue)	High
	No tillage (residue removed)	Medium-Low
	50% residue incorporation	Medium
	75% residue incorporation	Low
Practice factor (PF)[e]	Terraces	High
	Contour tillage	Low
	Contour conservation tillage	Medium-Low
	Contour cropping (conventional tillage)	Medium-Low
	Contour cropping (no tillage)	Medium-High
	Contour strip cropping (conventional tillage)	Medium
	Contour strip cropping (no tillage)	High

[a] Revised Universal Soil Loss Equation. The abbreviation SL represents the slope length factor, CC is the crop cover management factor, and PF represents supporting practice factor.

[b] Relative reduction sediment transport to the field edge as influenced by conservation practices.

[c] Narrow (<1 m) permanent grass strips planted on the contour. As with terraces, the steeper the slope the smaller the interval between strips.

[d] Crop cover relates the amount of previous crop residue left on the surface and growing crop canopy available to protect the soil surface from raindrop impact. Crop cover factor varies greatly with previous and growing crop. Increasing tillage intensity decreases surface crop residue cover and increases potential sediment loss.

[e] Practice factor reduces estimates of sediment loss with adoption of contour tillage, contour cropping, and contour strip cropping relative to tillage and cropping parallel with the slope direction. Terraces are also included in this factor because terraces are installed on the contour.

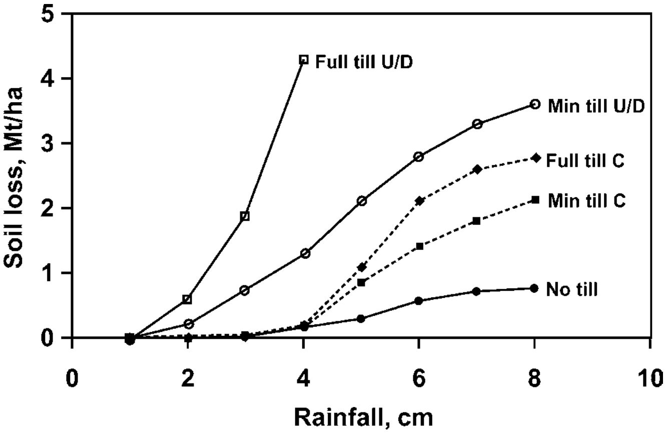

Figure 3. Influence of rainfall, tillage direction, and residue level on soil loss. Treatments shown include the following: full till U/D = full tillage (no residue cover) up and down or parallel to the slope direction; min till U/D = minimum till (30% residue cover) up and down the slope; full till-C = full tillage on with the slope contour; min till-C = minimum tillage on with the slope contour; no till = no tillage (100% residue cover).

slope length, crop cover, and supporting practices depend on soil and crop management to influence sediment loss. Table 1 illustrates relative effectiveness of selected conservation practices in reducing sediment transport. Because there are many interacting factors that influence soil erosion control, it is difficult to generalize. For example, terracing is one of the most costly conservation practices to implement and, thus, is not commonly used unless slopes are >3 to 5%. However, when properly designed and maintained, they can reduce effective slope length and the erosive kinetic energy associated with runoff compared with longer slopes. Conservation practices also have cumulative effects. For example, contour cropping and/or contour strip cropping combined with terracing can be more effective than contour cropping or strip cropping alone. If these systems are implemented with no-tillage management (all crop residues left on the field) the potential sediment loss is further reduced. Figure 3 illustrates the combined benefit of increasing surface residue cover and contour tillage on soil loss (Dickey et al., 1986).

Table 2. Effect of receiving slope length on the sediment delivery ratio and proportion of particulate P delivered to the field edge[a,b]

Receiving slope length, m	Sediment delivery ratio	Edge of field particulate P delivery
0	1.00	1.00
15	0.45	0.58
30	0.35	0.46
60	0.15	0.20
90	0.10	0.13

[a] Ratio of soil transported by erosion to sediment delivered to the receiving slope.

[b] Edge of field particulate P delivery reflects an increase in sediment delivery ratio of 1.3 to account for enrichment of more finely textured particles in sediment delivered to the field edge.

All NRCS technical specialists at the county level and many other public and private technical service providers are familiar with the use of RUSLE for estimating annual sediment loss. While most process-based models quantify sediment detachment and transport, the RUSLE is routinely integrated into soil and natural resource management programming throughout the United States. Implementation of RUSLE into a practical P loss assessment tool would be relatively easy because technical specialists would be familiar with its use.

Estimating Sediment Delivery

Quantifying particulate P delivered to the field edge also requires estimating the sediment yield or the amount of soil detached (soil erosion rate) minus soil deposited along the erosion slope. In this process, sediment delivery ratios (**SDR**) are established for typical receiving slopes and slope lengths to compute a SDR. The most critical factor influencing sediment deposition is the length and steepness of the receiving slope. Table 2 illustrates an example where estimated sediment P is reduced 80% with a 60-m receiving slope length. Soil texture also influences SDR where sediment yield increases with decreasing particle size (Foster et al., 1985). Thus, fine-textured soils exhibit greater sediment delivered to the field edge than course-textured soils.

Because the P content is higher in clay-enriched sediment, an enrichment factor should be applied to SDR to compensate for differential deposition by particle size (Menzel, 1980; Ongley, 1982; Sibbesen, 1995). The preferential transport of soil clays occurs as coarse soil fractions settle out of the runoff water, resulting in greater sediment-bound P at the field edge compared with the source soil. Sharpley et al. (2002) suggested a 1.5 sediment P enrichment factor where enrichment decreased with increasing erosion rate (Figure 2). As erosion rate increases, particle size separation decreases, resulting in decreased enrichment. Sibbesen (1995) reported an enrichment factor of 1.3 (Table 2).

Estimating Phosphorus Adsorbed to Soil Particles.

To quantify the sediment P delivered to the field edge, the amount of P adsorbed to soil must be estimated. Estimates of adsorbed P are site specific because P adsorption varies with soil pH, clay content, and other properties influencing P buffering potential or the amount of P (milligrams per kilogram) required to raise soil test P by 1 mg/kg (Mehlich, 1984). In North Carolina, for example, Cox (1994) correlated P buffering potential (measured by Mehlich-3 P soil test) to average clay content to estimate the quantity of P adsorbed to

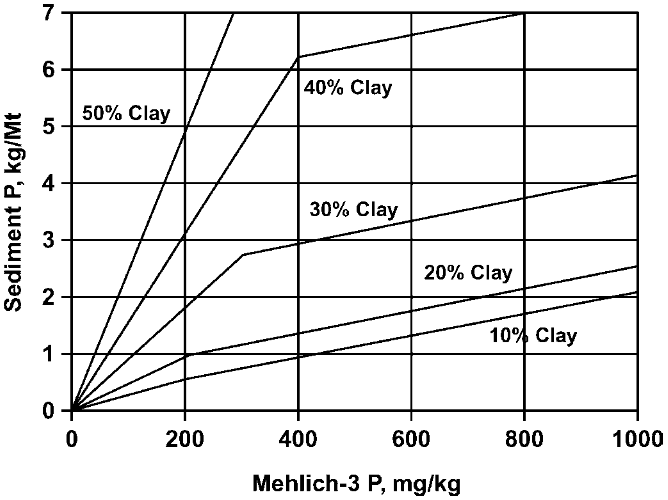

Figure 4. Sediment P determined by clay content and the Mehlich-3 P soil test for P (Mehlich-3 P level). Inflection point represents threshold Mehlich-3 P that produces ≥1 mg/L of soluble P.

sediment (Figure 4). Some have suggested that total soil P in the eroded sediment can be used to estimate particulate P loss (Sharpley et al., 2002). However, plant or bioavailable P forms, as estimated with routine P soil test methods, may be more sensitive relative to water quality impacts (Sims et al., 2002). Again, a practical P loss assessment tool or index should use as many existing methods and procedures as possible. Measurement of total P, compared with soil test P, in soil samples collected from agricultural field is not a routine method for most soil testing laboratories.

Sediment Phosphorus Trapping Practices

Vegetative buffers between the field and stream edge (Figure 1) have been shown to remove from 20 to more than 90% of sediment P in surface runoff, depending on buffer width (Daniels and Gilliam, 1996; Dillaha et al., 1989). Increasing buffer width increases sediment P removed. In contrast, buffers are generally considered ineffective in reducing soluble P in runoff. Sediments must be evenly distributed within the buffer to maintain long-term effectiveness in reducing sediment P. Using data from North Carolina, approximately 60% of sediment P can be retained in a 5-m-wide buffer (Figure 5).

Several in-field conservation structures can also help reduce sediment P delivery to the stream edge. These include controlled drainage structures, sediment basins, and ponds. Whereas ponds will generally maintain standing water throughout the year, sediment basins do not. Based on numerous field studies, controlled drainage can reduce sediment P by 35% (Evans et al., 1991). Although limited data are available for sediment basins (Bhaduri, et al., 1995; Borden et al., 1998; Steegen et al., 2001; Verstraeten and Poesen, 2002), sediment P reductions of 10 to 50% have been observed.

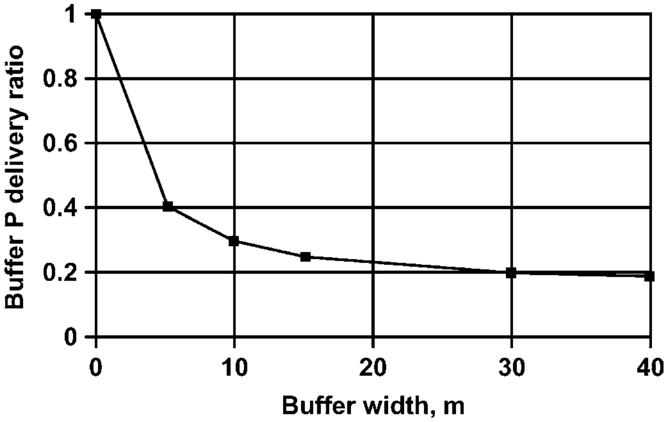

Figure 5. Effect of riparian buffer width on P delivery ratio in the buffer, which represents the fractional decrease in sediment P with increasing buffer width (data summarized from Peterjohn and Correll, 1984; Magette et al., 1989; Daniels and Gilliam, 1996; and Dillaha and Inamdar, 1997).

Phosphorus in Surface Runoff

Total soluble P in surface runoff water delivered to the stream edge is determined by estimating runoff volume (hectare centimeters) and P concentration in the runoff (milligrams per liter).

Runoff Phosphorus Concentration

Initially, the U.S. Environmental Protection Agency suggested 1 mg/L dissolved P as a guideline for agricultural runoff to protect surface water quality (U.S. EPA, 1986). However, processes controlling runoff P concentration and ultimate transport to surface water are influenced by many factors within the field and between the field edge and edge of the stream or water body. It is more appropriate to evaluate contributions of dissolved P in runoff relative to factors specific to each site. For example, in one watershed or field, 0.05 mg P/L could be considered too high whereas in another field 2 mg P/L in runoff may not significantly degrade surface water quality.

One easily measurable parameter that is directly related to dissolved P in runoff is soil test P. Three soil tests are commonly used in the United States and were originally designed to provide indices of plant available P for purposes of making P recommendations for crop production. The Mehlich (Mehlich, 1984) and Bray (Bray and Kurtz, 1945) soil tests are typically used for acid soils in which Al and Fe phosphate minerals control soil solution P concentration and P availability, whereas the Olsen soil test (Olsen et al., 1954) is used in neutral and calcareous soils, in which Ca phosphate minerals predominately control soil solution P concentration. The advantage of the Mehlich soil test over the Bray soil test is its ability to extract other plant nutrients in addition to P.

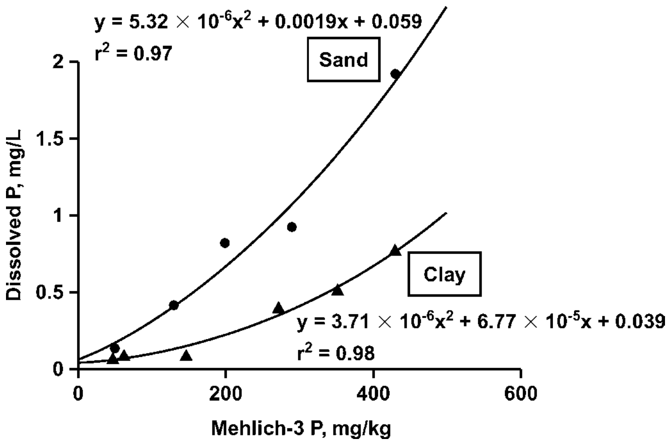

Figure 6. Dissolved P in runoff on North Carolina sand and clay soils as influenced by the Mehlich-3 soil test for P (Mehlich-3 P level).

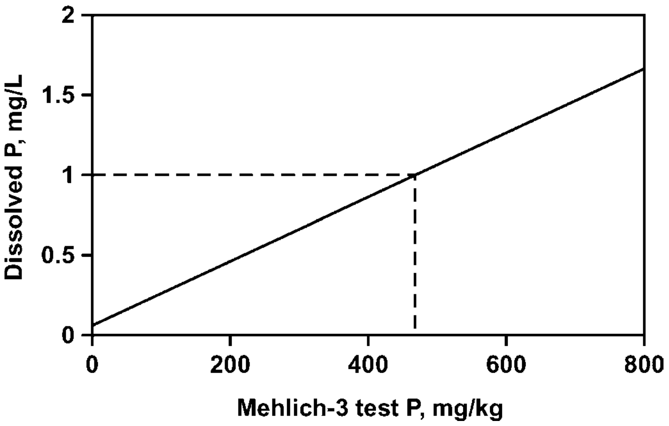

Figure 7. Influence of Mehlich-3 P soil test (Mehlich-3 P level) and dissolved P in runoff in a Pennsylvania watershed.

The Mehlich-3 P soil test has been established as a useful index of the potential environmental risk of increasing P in soils (Sharpley et al., 1996; Khiari et al., 2000; Sims et al., 2002). The soil solution P concentration is related to percentage of phosphorus saturation of the soil, which in turn influences Mehlich-3 P level. Numerous studies have established relationships between soil test P and dissolved P concentration in runoff (Sharpley et al., 1994; Pote et al., 1999; Weld et al., 2001; Torbert et al., 2002). In these studies, the Mehlich-3 P level that results in 1 mg P/L in surface runoff varied from 150 to 625 mg P/kg soil. Because P is adsorbed more strongly in clay soils, higher Mehlich-3 P levels are associated with high clay soils. Therefore, higher soil test P is required for a given runoff P concentration in clay soils compared with sandy soils.

Similar results were obtained from studies on the relationship between Mehlich-3 P and soluble P in runoff in North Carolina soils (Cox, 1994; Cox and Hendricks, 2000). For a given concentration of Mehlich-3 P, there was more dissolved P in runoff in sandy soil than clay soil (Figure 6). A concentration of 1 mg P/L in runoff from a sandy soil was associated with a Mehlich-3 P soil test of 253 mg/kg, whereas it was estimated by extrapolation to be about 500 mg/kg Mehlich-3 P for a clay soil (Figure 6). This difference is related to P being held less tightly in the sand compared with the clay soil because of differences in P adsorption capacity.

McDowell and Sharpley (2001) showed that in a Pennsylvania watershed (clay and clay loam soils) dissolved P in runoff increased with Mehlich-3 P (Figure 7). These results also showed that a Mehlich-3 P level of 500 mg/kg resulted in a dissolved P concentration of approximately 1 mg/L. Based on many studies, the effects of Mehlich-3 P on runoff P concentrations for the major soil groups were established for North Carolina (Figure 8). Mehlich-3 P levels producing 1 mg/L soluble P for the organic, sand, loam, and clay soils were 50, 100, 200, and 500 mg/kg, respectively (Novais, 1977; Reddy et al., 1980; Cox and Hendricks, 2000).

Runoff Volume

The quantity of runoff water associated with an individual storm event depends on characteristics of the rainfall (quantity, intensity, and duration), surface soil conditions that influence infiltration (residue cover and soil physical properties, including water content), subsoil properties that influence hydrologic conductivity (soil structure, texture, water content), and water table depth (USDA-NRCS, 1989). During rainfall, water enters the soil through large, surface-connected macropores under a positive hydraulic head. Water then diffuses vertically and horizontally into a network of micropores by capillary action or soil moisture tension (**SMT**). Water flow into the subsoil volume is predomi-

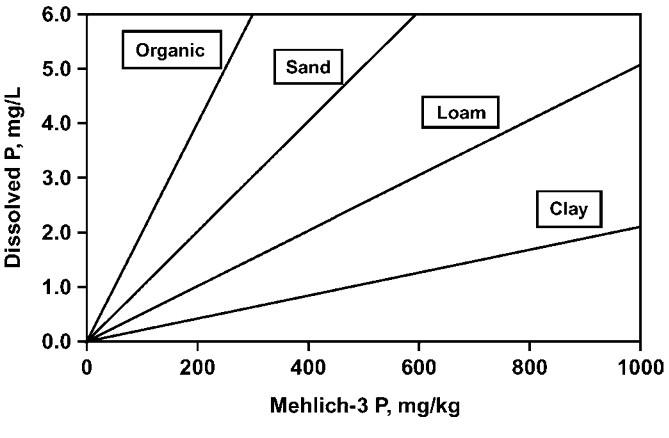

Figure 8. Concentration of dissolved P as a function of Mehlich-3 soil test P level for major soil textural classes. The Mehlich-3 P levels that result in 1 mg/L dissolved P are as follows: organic soils = 50; sands = 100; loams = 200; and clays = 500.

Table 3. Soil Characteristics used in the curve number method of estimating surface runoff

Soil group	Soil characteristics	Minimum infiltration capacity, cm/h
A	Sandy, deep, well-drained soils; deep loess; aggregated silty soils, lowest runoff potential	>0.76
B	Sandy loams, shallow loess, moderately deep and moderately well-drained	0.38–0.75
C	Clay loam soils, shallow sandy loams with clay subsoil layer, low-organic-matter soils	0.13–0.37
D	Heavy clay soils with swelling potential, water-logged soils, certain saline soils, shallow soils with impermeable subsoil layer	<0.12

nately governed by SMT. After soil macropores are filled, water moves through micropore areas toward the highest SMT. The water infiltration or transport rate is governed by the number, size, and continuity of the pore network. The presence of old root channels, earthworm borrows, and natural subsoil structural macropores can substantially increase water infiltration and transport of dissolved P through the profile; however, their presence and influence are difficult to quantify.

Presence of a shallow water table can reduce water transport in the subsoil and increase runoff. In regions where shallow water tables are prevalent, artificial drainage is commonly installed to improve infiltration and soil productivity. Models designed to estimate runoff volume should account for both drained and undrained soil conditions (Skaggs et al., 1982; Evans et al., 1995). In addition, whatever runoff model is incorporated into a practical P loss assessment tool, it must be based on annualized runoff instead of individual runoff events, and should be capable of routine use by field technicians.

These criteria are met for well-drained, upland soils, with the empirical curve number approach used to estimate runoff volume (USDA-NRCS, 1989). The curve number method is also used in several process-based simulation models to predict daily runoff (Knisel, 1993; Sharpley and Williams, 1990; Arnold et al., 1998). The curve number method relates runoff potential to land use and soil characteristics. The runoff depth is determined from the total rainfall adjusted for estimated infiltration determined from surface and subsoil physical characteristics. The method assumes that an accumulated rainfall depth of $>0.2 \times$ maximum soil water retention must occur before generating runoff. The maximum soil water retention and curve number are determined for major soil hydrologic groups based on soil physical properties, vegetation, land use, and antecedent moisture conditions or soil water content at time of runoff producing rainfall (Table 3). Curve numbers are established to reflect variations in soil properties and antecedent moisture that influence infiltration and runoff. For a specific rainfall amount, runoff would increase with increasing curve number (Figure 9). In practice, relationships similar to Figure 9 are developed for each county using long-term rainfall records, various cropping systems, and hydrologic soil conditions to estimate runoff volume.

For artificially drained, shallow-water-table soils, the curve number approach does not accurately estimate runoff volume. Most shallow-water-table soils similar to the coastal plain regions of the United States are not suitable for agricultural production unless artificial drainage is installed. Calculation of runoff for drained soils requires information on the drain depth, the estimated distance between drains, and the soil transmissivity (square centimeters per hour), which represents the ability of the soil profile to transmit water laterally when the water table is 30 cm below the soil surface. These values have been established for all soils in areas where artificial drainage is commonly used. The model widely accepted for this purpose is DRAINMOD, which has been thoroughly field-calibrated and is widely used to estimate runoff volume and water transport through the profile in high-water-table soils (Skaggs, 1999; Skaggs and Chescheir, 1999).

Soluble Phosphorus Retaining Practices

Unlike sediment P, there are few conservation practices that can reduce soluble or runoff P before reaching the stream edge. However, any soil management practice that increases infiltration and decreases runoff (conservation tillage or water control structures) can reduce the transfer of soluble P to surface water. Ripar-

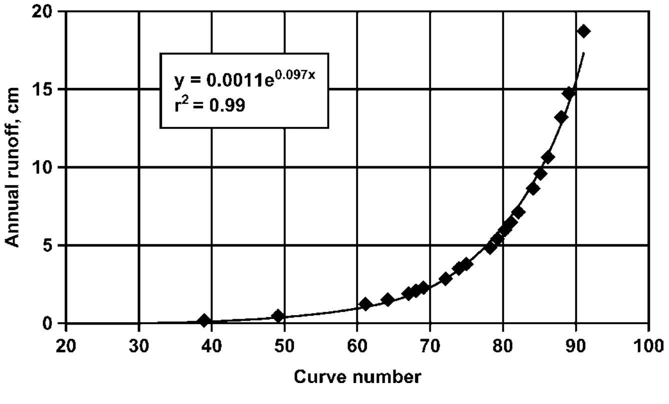

Figure 9. Example relationship between curve number and runoff volume for Wake County, North Carolina.

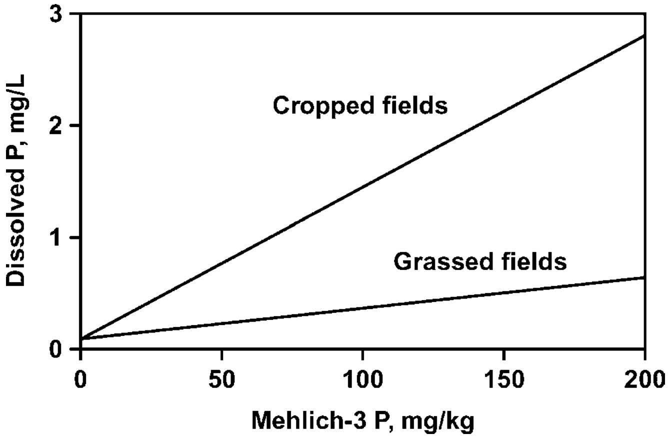

Figure 10. The effect of Mehlich-3 P soil test level (Mehlich-3 P) and cropping system on dissolved P in runoff. Cropped fields represent wheat with residue incorporated with a moldboard plow and grassed fields represent native short grass pasture.

ian buffers, for example, do not reduce transfer of soluble P, but they are very effective in reducing transfer of sediment P to surface water (Nash and Murdoch, 2000). Studies in Oklahoma show that permanent residue cover (pasture) resulted in much less dissolved P in runoff compared with row-cropped fields, predominately due to reduced contact between runoff water and surface soil (Sharpley et al., 1991; Smith et al., 1991) (Figure 10). Sharpley et al. (2002) summarized data over a wide geographical area in the United States and showed that increasing surface residue cover decreased dissolved P in runoff (Figure 11).

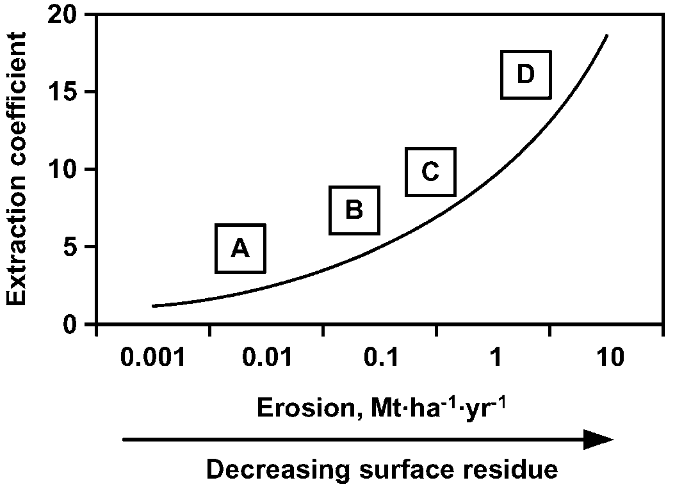

Figure 11. Influence of soil erosion rate, which increases with decreasing surface residue cover, and the P extraction coefficient, which represents the slope of the linear relationship between Mehlich-3 soil test P and dissolved P concentration in runoff water. A = native grass/pasture; B = no tillage; C = reduced tillage; and D = conventional tillage.

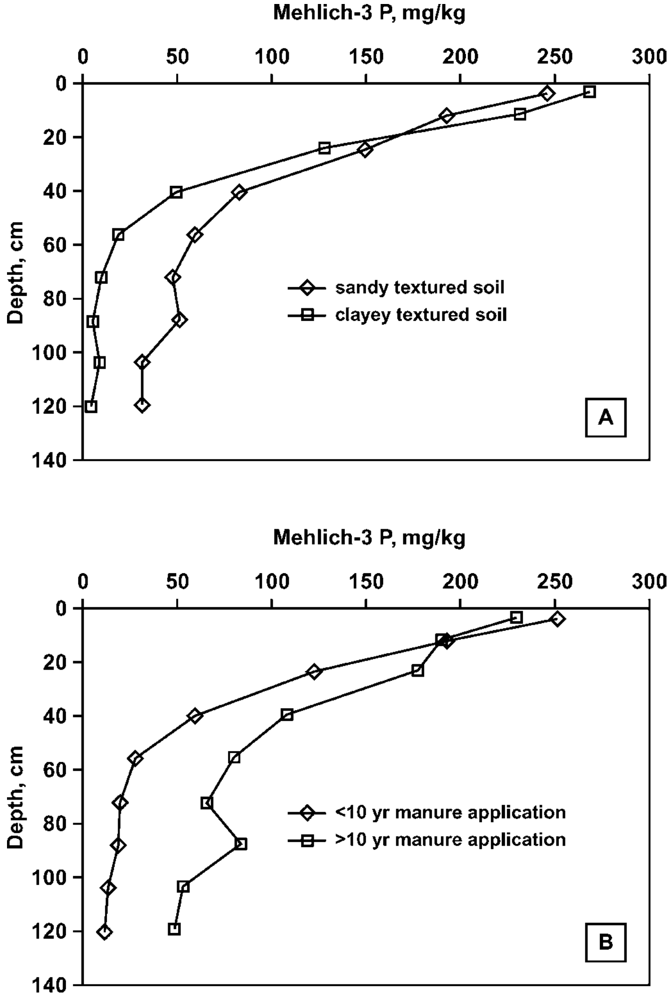

Figure 12. Influence of soil texture (A) and duration after surface waste application (B) on P leaching in a sandy soil.

Phosphorus in Subsurface Drainage

There is potential for P to leach below the root zone and be transported to surface waters through subsurface flow (Figure 1). Because P is strongly adsorbed to soil particles, P leaching would occur only when the percentage P saturation of the soil is increased to very high levels through continued applications of P exceeding crop requirement. Although P leaching is frequently greater in sandy soils (low P adsorption capacity), P leaching can occur in some clay soils through transport in macropores (Djodjic et al., 1999; Laubel et al., 1999). Estimating P leaching potential requires quantifying drainage water volume and the P concentration in the drainage water.

Phosphorus Concentration in Drainage Water

Many recent studies have demonstrated downward movement of P in soils where high rates of manure were applied over extended periods. Ham et al. (2000) demonstrated P leaching in a sandy soil when Mehlich-

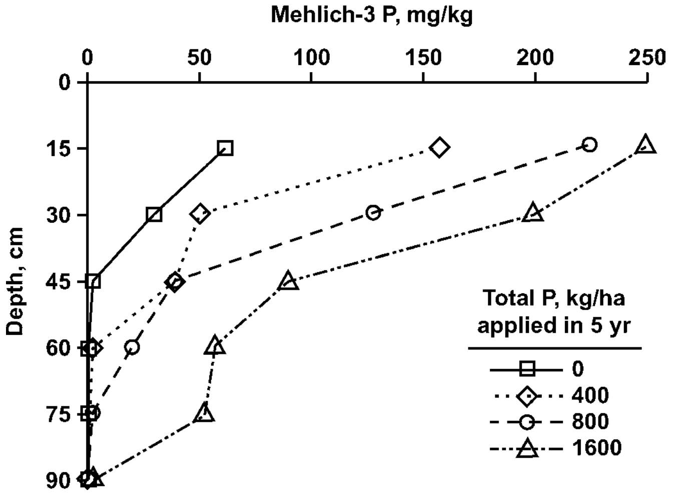

Figure 13. Effect of total P applied as swine effluent during a 5-yr period on distribution of Mehlich-3 P soil test level.

3 P (0 to 20 cm depth) was 250 mg P/kg (Figure 12). At this Mehlich-3 P level, the P adsorption capacity was nearly saturated (100% P saturation at 270 mg/kg). These authors also showed the influence of continued waste application on increasing P leaching potential (Figure 12). Application of 1,600 kg P/ha to a fine loam soil over 5 yr resulted in P leaching to 76 cm in depth (Figure 13). In this soil, the P adsorption capacity was 100% saturated at 250 mg/kg Mehlich-3 P (Reddy et al., 1980).

Based on numerous leaching studies, the P concentration in the leachate can be estimated using Mehlich-3 P levels, similar to the relationship used for estimating soluble P in runoff. For example, in North Carolina, if Mehlich-3 P levels (0 to 20 cm in depth) are less than the threshold values in Figure 8, then P leaching does not likely occur. If Mehlich-3 P levels exceed these threshold values, however, the soil profile to a depth of 75 cm should be sampled to determine whether P leaching has occurred. Depending on the Mehlich-3 P level at a 75-cm depth, the leachate P concentration can be estimated from Figure 14.

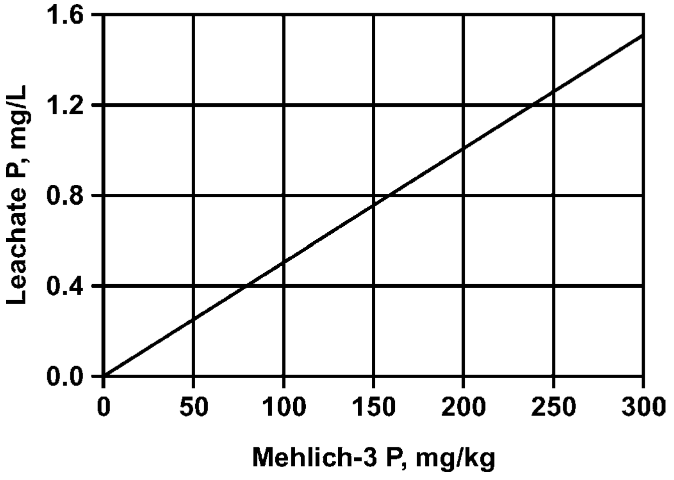

Figure 14. Dissolved P concentrations as a function of Mehlich-3 soil test P levels at the 0.75-m depth.

Using the Olsen soil test, Heckrath et al. (1995) reported that P concentration in the drainage water substantially increased at Olsen soil test P levels >60 mg/kg. These researchers forwarded the *change point* concept, or the soil test P threshold above which dissolved P in the drainage water greatly increases (McDowell and Sharpley, 2001). In similar work, Maguire and Sims (2002) reported greatly elevated dissolved P in drainage water when Mehlich-3 P levels exceeded approximately 200 mg/kg. In development of practical P loss assessment tools, the *change point* or soil test P threshold should be established for the major soil groups in each state.

Drainage Water Volume

When P leaches below the root zone, intensive subsurface drainage will increase potential for subsurface transport (Figure 1). Intensive subsurface drainage also decreases surface runoff that reduces potential runoff P loss (Skaggs et al., 1982; Gilliam and Skaggs, 1986; Evans et al., 1995; Gilliam et al., 1999). The factors affecting subsurface P transport are subsurface drainage intensity (rate of drainage water leaving a field) and P adsorption capacity of the soil. Subsurface drainage occurs when the capacity of surface water storage has been satisfied. Subsurface drainage is influenced by soil hydraulic conductivity, profile depth, water table depth, and characteristics of artificial drainage (drain depth and spacing) if present.

For purposes of estimating P leaching potential, when the estimated leachate P concentration equals 0 mg/kg, calculating leachate volume is unnecessary. For well-drained soils with a calculated leachate P concentration of >0 mg/kg, the average annual precipitation (**PPT**), runoff volume calculated from the curve number method, and evapotranspiration (**ET**) based on site-specific data on climate, crop, and soil type can be used to estimate subsurface drainage volume by the following equation:

$$\text{Average subsurface drainage (cm)} = \text{Annual PPT (cm)} - \text{Runoff (cm)} - \text{ET (cm)}$$

As with estimating runoff volume in drained soils discussed previously, the DRAINMOD model (Skaggs, 1978) is the most accurate method to estimate water balance of shallow-water-table soils. DRAINMOD calculates evapotranspiration, infiltration, surface runoff, subsurface drainage, deep drainage, water table depth, and soil water distribution. Model descriptions and applications are presented by Evans and Skaggs (1989), Skaggs (1999), and Skaggs and Chescheir (1999). DRAINMOD can be readily incorporated into practical

Table 4. Selected sources of animal wastes and their P content[a]

Waste source	Total P[b]	Soluble fraction[c]	Soluble P[d]	Nonsoluble P[e]
Beef				
Lagoon liquid, kg P·ha^{-1}·cm^{-1}	15.0	0.80	12.0	3.0
Lagoon sludge, g P/L	2.7	0.60	1.6	1.1
Slurry, g P/L	1.2	0.75	0.9	0.3
Dairy				
Lagoon liquid, kg P·ha^{-1}·cm^{-1}	15.0	0.80	12.0	3.0
Lagoon sludge, g P/L	1.2	0.60	0.7	0.5
Scraped, kg P/t	1.4	0.60	0.8	0.6
Slurry, g P/L	0.7	0.75	0.5	0.2
Swine				
Lagoon liquid, kg P·ha^{-1}·cm^{-1}	10.3	0.80	8.2	2.1
Lagoon sludge, g P/L	2.6	0.40	1.0	1.6
Slurry, g P/L	1.2	0.60	0.7	0.5
Broiler				
Fresh manure, kg P/t	3.6	0.25	0.9	2.7
House litter, kg P/t	17.3	0.25	4.3	13.0
Stockpiled litter, kg P/t	17.5	0.25	4.4	13.1
Layer				
Highise manure, kg P/t	12.3	0.60	7.4	4.9
Lagoon liquid, kg P·ha^{-1}·cm^{-1}	8.9	0.80	7.1	1.8
Lagoon sludge, g P/L	4.9	0.50	2.5	2.4
Slurry, g P/L	3.1	0.60	1.9	1.2
Undercage manure, kg P/t	6.9	0.50	3.5	3.4
Turkey				
Stockpiled litter, kg P/t	15.9	0.25	4.0	11.9
House litter, kg P/t	11.5	0.25	2.9	8.6

[a] Barker et al. (1994), USDA-NRCS (1998), Dou et al. (2000), and Sharpley and Moyer (2000).
[b] Concentration units vary with waste source.
[c] Weight basis.
[d] Soluble P = total P × soluble fraction.
[e] Insoluble P = total P − soluble P.

P loss assessment tools used by technical service providers and other practitioners. Once drainage water volume and P concentration are estimated, the quantity of P leached can be determined.

Phosphorus in Waste Sources

Generally with fertilizer P applied at recommended rates, P losses are related to sediment transport and to a lesser extent soluble P runoff. With annual applications of animal waste providing 3 to 5 times the P requirement of the crop, runoff P loss generally increases depending on waste source characteristics, application method, soil properties, erosion potential, cropping system, and environmental conditions following P application (Khaleel et al., 1980; Westerman et al., 1983; Edwards and Daniel, 1994). During runoff events, soils with low erosion potential have contributed high losses of soluble and non-sediment bound P. To consider waste source contributions to P transport, we focus on the P in the waste that is transported to the water body edge that does not or has not interacted with the soil or other particulate matter in the field during transport. The waste source characteristics that influence P delivery to a surface water body are total P content and the soluble fraction defined as the solid proportion of the total waste product on a weight basis.

Source Characteristics

Total P content of waste sources varies widely with animal species, diet, and method of waste handling and storage (Table 4). In general, P content in common waste sources follows this order: poultry and turkey > beef > swine > dairy. Increasing P concentration the animal diet will increase P content in the waste. Total P concentration within a waste source varies depending on waste handling system, where the order generally is dry litter > liquid > slurry > sludge (Gilbertson et al, 1979; Barker et al., 1994; Barnett, 1994b; Sharpley and Moyer, 2000). Inorganic P makes up 60 to 90% of total P in animal wastes, where about 20 to 80% of total P is water soluble (Barnett, 1994a; Sharpley and Moyer, 2000).

Waste sources with high P content solubility contain a maximum amount of soluble P (Table 4), which can infiltrate into the soil and reduce potential soluble P loss (Westerman et al., 1983). Several studies have determined that P loss in runoff is highest during the first runoff event following waste application; however, P losses decrease with subsequent runoff events (McLeod and Hegg, 1984; Edwards and Daniel, 1994). Waste materials with a high percentage of solid content will remain on the soil surface until decomposed or dissolved with rainfall. Transport of low-density manure parti-

Table 5. Soluble and insoluble P attenuation factors[a]

Waste source	SPAF[b]	NSPAF[c]
Litter	0.4	0.1
Manure	0.4	0.1
Sludge	0.4	0.1
Scraped	0.4	0.1
Slurry	0.3	0.1
Liquid	0.1	0.1
Fertilizer	0.1	NA

[a] Adapted from Westerman and Overcash (1980), Westerman et al. (1983), Edwards and Daniel (1993a,b), and Tarkalson (2001).
[b] SPAF = Soluble P attenuation factor.
[c] NSPAF = Insoluble P attenuation factor.

cles in runoff is greatest within 3 d of application (Westerman et al., 1983). Solubility of P in the waste is a useful indicator of P loss (Dou et al., 2000), and water-soluble P content is a useful indicator of P in runoff or leaching water (Sharpley and Moyer, 2000).

The potential amount of transportable P that is delivered to the field edge after in-field adsorption and infiltration processes occur is not well documented. Generally, as rainfall intensity increases, greater runoff volume decreases the concentration of total and dissolved P by dilution in runoff (Edwards and Daniel, 1993a,b). For surface-applied swine slurry, increasing rainfall intensity diluted P concentrations but did not increase losses of P mass (Edwards and Daniel, 1993b). However, in a similar study with surface-applied poultry litter, increased rainfall intensity resulted in increased mass loss of P, although P concentration in runoff decreased (Edwards and Daniel, 1993a). Causes for the differences between waste sources were not studied but could result from differences in water-soluble P concentration in the waste or differences in physical characteristics represented by soluble fraction (Table 4).

Recent research has demonstrated the value of metal salts or by-products containing Al, Fe, or Ca added to solid or liquid wastes in decreasing soluble P content in the waste. With animal wastes, aluminum sulfate (or alum) has been used to decrease soluble P content by nearly 70% (Sims and Luka-McCafferty, 2002). Moore et al. (2000) reported 73% reduction in runoff P with alum-treated poultry waste compared with untreated waste applied to pastures. Practical P loss assessment tools should include the use of alum and other products to reduce potential P loss. It is important to remember that although soluble P loss in runoff can be substantially reduced with alum-treated wastes, total P load to a soil is the same as untreated waste. Thus, soil P concentrations will continue to increase with waste applied in excess of crop P requirement.

Soluble and Insoluble P Attenuation

Although both soluble and particulate P are being transported in runoff water to the field or stream edge, a proportion of this mobile P is retained in the field (Table 5). Some within-field attenuation or readsorption of soluble P to eroded sediments occurs. If P attenuation is not accounted for, P runoff is overestimated (Kleinman et al., 2002a,b; Vadas et al., 2002). Because many P loss studies are usually conducted in small plots, the P loss measured overestimates soluble and particulate P delivered to the field edge (Edwards and Daniel, 1994). Vervoort et al. (1998) confirmed that soluble P loss measured on 0.45-ha areas was less than soluble P loss measured on small plots.

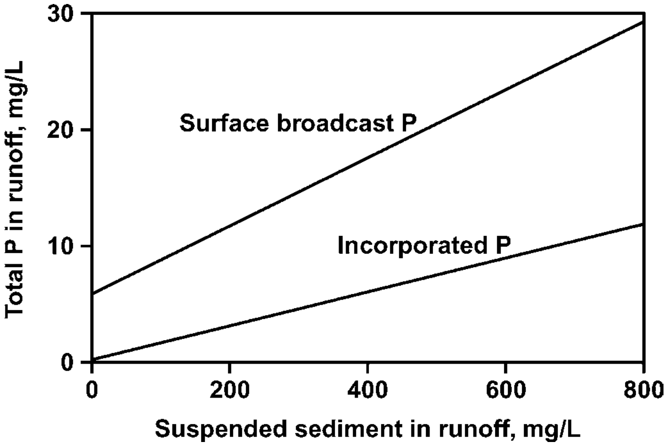

Figure 15. Influence of sediment content in runoff and surface broadcast or incorporated P in manure or fertilizer on total P concentration in runoff.

Application Rate and Method

Regardless of whether P loss is measured as concentration (milligrams per liter) or mass (kilograms per hectare), P transport increases with increasing waste application rate (Westerman et al., 1983; Edwards and Daniel, 1993a). As a result, application rate must be included in any model used to predict P transport to surface and groundwater. It is also well established that incorporated or injected wastes are less susceptible to runoff loss than surface applications (Mueller et al., 1984). Thus, surface applications to pasture result in greater soluble P loss than in cultivated soils regardless of the percentage of residue cover (Daniel et al., 1993). Kleinman et al. (2002b) reported substantially greater P in runoff water when manure or fertilizer P was incorporated compared with surface broadcast (Figure 15). Tarkalson (2001) reported a 95% decrease in loss of soluble P with incorporated broiler litter compared with surface applied litter. Based on these data, P loss assessment tools might use a factor of 0.05 (95% reduction in P loss) for incorporation within 48 h (Table 6). As the time interval between application and incorporation increases, potential runoff P loss increases.

Application of animal waste in relation to probability of rainfall is an important factor in managing loss of waste-derived P to surface water. Sharpley et al. (1994) reported that runoff P can be reduced by applying waste

Table 6. Approximate factors used to adjust P loss estimates for method of P application

Application method	Factor
None applied	0.00
Injected	0.01
Incorporated within 48 h	0.05
Incorporated within 4 wk or less following application	0.10
Incorporated between 4 and 12 wk following application	0.50
All other surface (unincorporated) applications	1.00

during periods of low rainfall probability, with greater reductions occurring on soils with high P adsorption capacity. Waste application just before a rainfall event can lead to significant P losses in runoff (Eghball et al., 2002). It is important to note that in field studies, decreases in P loss over time for periods greater than 1 to 3 d generally result from a combined effect of the extent of prior removal of readily transportable P in previous runoff events and interactions with subsequent rainfall and the soil, which does not always equate with a decrease in the cumulative mass of P loss. Studies also estimate that a period of 12 to 20 mo after waste application may be required to reduce P in runoff to <1 mg/L (Pierson et al., 2001).

Riparian buffers between the field edge and the surface water body can be effective in trapping sediment P, further reducing the P loss. Even though many factors determine buffer effectiveness, buffer width is the most critical (Figure 5), and, whereas buffers are effective in trapping sediment, they are less effective in trapping soluble P.

Summary

Recently, the USDA-NRCS required each state to develop methods or tools to estimate the quantity of P delivered to surface and groundwaters, so it is important for all nutrient managers to understand the fate and transport of P applied to soils. Currently, only those producers participating in federal commodity price support programs are required to incorporate a P loss assessment into their nutrient management plans. Once developed, however, P loss assessment tools will likely be incorporated into state regulations pertaining to both animal waste and fertilizer application to agricultural crops.

Mechanistic tools used to quantify P loss associated with fertilizer or waste P applications will enable producers to identify best management practices that could reduce or minimize P transport off a field. Accurate estimates of P loss require methods to quantify 1) P adsorbed to eroding sediments, 2) soluble P in runoff water, 3) soluble P in leaching water, and 4) P losses related to the specific waste sources. With sediment P, the mass of sediment and the quantity of P adsorbed to the sediment is estimated. Conservation practices including riparian buffers, in-field or edge-of-field water control structures, and other soil and crop conservation management practices reduce sediment P loss between the field edge and the edge of the water body.

Quantifying soluble runoff P requires estimation of runoff volume and P concentration. Besides runoff-reducing practices based on increasing rainfall infiltration (conservation tillage, contour cropping, terraces, etc.), there are few practices that effectively decrease soluble P between the field and water edge. Leaching of P depends on buildup of P in the surface soil at levels that exceed the P adsorption capacity. Generally, coarsely textured soils exhibit greater P leaching potential than finer textured soils. Quantifying P leaching requires estimating leaching water volume and P concentration. Much of the P applied to soil is retained in the field through adsorption processes, but some of the waste P remains in its original soluble or particulate P. Although waste sources vary greatly in soluble P content, it can be used to estimate source-related P losses.

Substantial research and scientific evidence supports these mechanisms of P loss, but several components require additional research. Specifically, our ability to quantify leaching and waste source-related P losses is much less than for sediment and soluble P in runoff. Another important advantage and benefit of developing P loss assessment models is that they can expose areas of research essential to improving the accuracy of P loss estimates. Regardless of inherent weaknesses of a mechanistic modeling approach, assessment tools that enable the user to assess the impact of adoption of best management practices on reducing potential P loss will decrease the risk of P use on water quality.

The accuracy of a P loss assessment tool is important, but the relative ease of use by technical service providers and other practitioners also must be considered. Therefore, the algorithms imbedded into the four essential components discussed above should be based on well-established and user-friendly models to estimate sediment-bound and soluble P losses. A successful and practical P loss assessment tool should not demand that users fully understand the detailed theory behind the calculations, although they should be trained enough to understand and assess the accuracy of the relative P loss estimates, and evaluate the benefit of adopting best management practices on water quality protection.

Implications

Producers receiving federal commodity support payments are required to incorporate P loss assessment into nutrient management planning. Accurate estimates of P loss require methods to quantify 1) P adsorbed to eroding sediments, 2) soluble P in runoff water, 3) soluble P in leaching water, and 4) P losses related to the specific P sources. Successful methods used to quantify P loss associated with P applications must also enable producers to identify best management practices that reduce or minimize P loss. Although the accuracy of a P loss assessment is important, the relative ease of use by practitioners is essential. Thus, methods that estimate P loss related to the four mechanisms discussed should be based on established and user-friendly methods. The intended use of P loss assessment tools by practitioners routinely interacting with land managers will help ensure that P use in agricultural ecosystems will minimize the contribution of P use on surface and groundwater quality.

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