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# The fate and transport of phosphorus in agricultural systems

N.C. Hansen, T.C. Daniel, A.N. Sharpley, and J.L. Lemunyon

**ABSTRACT:** Phosphorus (P) is an important input for economic crop and livestock production systems. Excessive losses of P from agricultural systems to surface waters can accelerate eutrophication and degrade water quality. This paper reviews the behavior of P in agricultural soils and discusses the transport of P from land to water. The forms, measurement, and sorption processes of P in both soil and water are discussed. Soil test P, the most common soil P analysis, is described relative to other forms of soil P and its use for agricultural and environmental purposes is explained. Loss of soil P to water can occur in particulate forms with eroded surface soil and in soluble forms in runoff, soil interflow, and deep leaching. This paper discusses the relative importance of each transport pathway as affected by soil type and management. Soil P dynamics and water quality risks associated with fertilizer and manure application are illustrated with several examples. Finally, the paper reviews management practices that can effectively reduce the loss of agricultural P to surface waters.

**Keywords:** Best management practices, reactive P, P sorption, particulate P, soil test P, soluble P, stable P, water quality

**Reducing the delivery of phosphorus (P) from agricultural sources to surface and groundwater is the focus of water quality efforts in many watersheds (Sharpley et al. 1994; Carpenter et al. 1998).** Water quality problems associated with excess P relate to accelerated eutrophication of inland, estuarine, and coastal waters and include low oxygen levels, reduced aquatic species diversity, turbidity, and undesirable taste and odor in municipal water supplies (Carpenter et al. 1998; NRC 2000; Sharpley et al. 1994; Smith 1998).

Phosphorus is an economically important input in both crop and livestock production systems (Mallarino and Blackmer 1992; Poulsen 2000; Valk et al. 2000). But the management of P for environmental and agricultural objectives is often disjointed because 1) the cost of P lost from agricultural systems is very small relative to the total cost of P inputs, and 2) the costs of water quality degradation from P occur downstream from the source and cross political boundaries. For example, the recommended P application rate for corn (*Zea mays* L.) grown on a soil low in available P in the Midwest can range from 25 to 45 kg $ha^{-1}$ $yr^{-1}$ (22 to 40 lb $ac^{-1}$ $yr^{-1}$) depending upon yield goal (Rehm et al. 1996). In contrast, the amounts of P loss that are associated with

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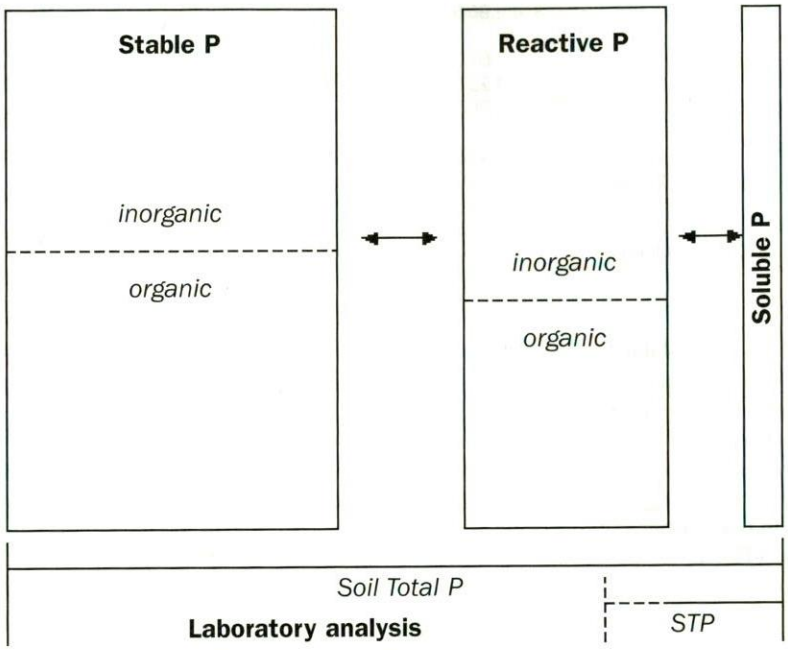
**Figure 1**
A diagram illustrating three pools of soil phosphorus (P) and how they relate to total P and soil test P (STP) laboratory analyses. The relative size of each pool, the composition of organic and inorganic P within each pool, and the concentration of STP vary with soil type.




accelerated eutrophication in lakes can be less than 1 to 2 kg $ha^{-1}$ $yr^{-1}$ (0.9 to 1.8 lb $ac^{-1}$ $yr^{-1}$) (Lennox et al. 1997; Sharpley and Rekolainen 1997). Thus, the challenge of P management in agriculture is to supply adequate P for economical production while minimizing losses to surface waters.

In this paper, we discuss the forms of P in soil and water and the fate and transport of P in agricultural systems. We also discuss management practices that can reduce the risk of P transport from agriculture to surface water.

## Soil Phosphorus

***Forms.*** Soil P is found in various chemical and physical forms, which differ in their plant availability and environmental fate. One way to describe soil P is to classify it into three hypothetical pools of differing reactivity. These pools are soluble P, reactive P, and stable P (Figure 1). The soluble P pool consists of the most reactive and plant available forms of soil P and is predominantly made up of orthophosphate anions (i.e., $H_2PO_4^-$, $HPO_4^{2-}$). Soluble P makes up only a very small portion of the total P in soil, often less than 1% (Brady and Weil 1999). The P in the reactive and stable pools is associated with the soil solid phase and occurs in both organic and inorganic forms (Sharpley and Rekolainen 1997). Phosphorus in the reactive pool consists of forms that are in dynamic equilibrium with solution P. The organic P in this pool is from relatively fresh organic material that is readily decomposed. The inorganic P in this pool is found on soil exchange sites or in relatively soluble minerals. When soluble P uptake or loss occurs, P from the reactive pool can replace it through processes such as desorption, dissolution, and mineralization. The stable P pool, often referred to as fixed, makes up the largest portion of soil P. Stable P is not biologically available and is made up of organic and inorganic compounds that are occluded, insoluble, or tightly sorbed. Inorganic P forms in the stable pool are dominated by crystalline aluminum (Al) and iron (Fe) compounds or by calcium (Ca) compounds (Brady and Weil 1999). In principle, stable P forms are in equilibrium with the other pools, but the reactions of stable P forms are too slow for time periods important in agricultural production (i.e., one growing season).

***Measurement.*** Various laboratory methods have been developed to characterize soil P (Pierzynski and Sharpley 2000). The most common laboratory methods used to quantify soil phosphorus are collectively known as soil test phosphorus (STP) methods. Historically, the goal of STP determination was to provide an estimate of the amount of soil P that would be available to a crop during a growing season. The STP determination involves an extraction of soil P that constitutes all the soluble P and a portion of the reactive P (Figure 1). The amount of reactive P extracted depends on the specific extraction method used and on soil properties such as clay content, organic matter, and pH. Several different extraction methods are in common use including Bray-1 (Bray and Kurtz 1945), Olsen (Olsen et al. 1954), Mehlich-I (Nelson

**Table 1. Common soil test phosphorus (P) methods and the associated chemical extractants, pH values, and primary uses (adapted from Sibbesen and Sharpley 1997).**

| Soil test P method | Chemical extractant | pH | Primary uses |
|---|---|---|---|
| Bray-1 | $NH_4F$ + HCl | 3.0 | Fertilizer recommendations for acid and neutral soils. |
| Mehlich-1 | HCl + $H_2SO_4$ | 1.2 | Multi-element extractant for fertilizer recommendations on acid and neutral soils. |
| Mehlich-3 | $CH_3COOH$ + $NH_4NO_3$ + $NH_4F$ + $HNO_3$ + EDTA | 2.5 | Multi-element extractant for fertilizer recommendations on most soils. |
| Olsen | $NaHCO_3$ | 8.5 | Fertilizer recommendations for calcareous soils. |
| Fe-oxide strip | FeO and 0.01 M $CaCl_2$ | – | Estimation of bioavailable P for agronomic and environmental purposes. |
| Water | $H_2O$ or 0.01 M $CaCl_2$ | – | Primarily for assessment of environmental risk. |

**Table 2. The effect of long term application of phosphorus (P) fertilizer or manure on the concentrations of soil total phosphorus (TP), soil test phosphorus (STP), and the STP as a percent of TP.**

| Soil type, location | Reference | History | TP (mg kg$^{-1}$) | STP (mg kg$^{-1}$) | STP Method | STP (%TP) |
|---|---|---|---|---|---|---|
| Pullman clay loam, Texas | Sharpley et al. 1984 | No P | 353 | 15 | Bray-1 | 4.2 |
| | | Long term manure | 996 | 230 | | 23 |
| Silty clay loam, Rothamsted | Johnston and Poulton 1997 | No P | 670 | 18 | Olsen | 2.7 |
| | | Long term P fertilizer | 1215 | 69 | | 5.7 |
| | | Long term manure | 1265 | 86 | | 6.8 |
| Mexico silt loam, Missouri | Kao and Blanchar 1973 | No P | 418 | 9 | Bray-1 | 2.1 |
| | | Long term P fertilizer | 700 | 57 | | 8.1 |
| Autryville loamy sand, North Carolina | Novak et al. 2000 | No P | 203 | 7.5 | Mehlich 3 | 4.0 |
| | | Long term manure | 404 | 353 | | 87 |

et al. 1953), and Mehlich-III (Mehlich 1984). Each of these methods uses different chemical extractants and laboratory procedures (Table 1) and therefore measures a different amount of soil P.

Because of the wide use of STP methods in agriculture, there is interest in expanding their utility to include assessment of environmental risks. Much effort has been devoted to the development of environmental soil test methods and their interpretation (Sibbesen and Sharpley 1997; Sims et al. 2000). The limitations of using STP methods to evaluate water quality risks include 1) STP may not measure the forms of P that are important for eutrophication, and 2) STP does not evaluate soil properties that control P transport to water.

Soil total P can be measured using a harsh extraction method (i.e., perchloric or nitric acid digestion), but total P analyses are not common in commercial soil testing laboratories because there is not a routine agronomic need for this information. The utility of total P for evaluating environmental risk is also limited because it does not indicate what portion of the P is bioavailable. Soil total P is not directly related to STP concentration, but varies with soil type and management. Table 2 shows typical values for STP and soil total P concentrations with and without long term applications of P. Total P concentration for the soils in these examples without applied P ranges from 203 to 670 mg kg$^{-1}$ (ppm), and STP by various methods represents <5% of the total P. With long term applications of P as either fertilizer or manure, the soil total P concentrations range from 404 to 1265 mg kg$^{-1}$ (ppm) and STP represents a much higher percentage of total P.

***Soil P sorption.*** Sorption of soil P is a general term that describes reactions in the soil that cause soluble and plant available forms of P to form less soluble or available forms. These processes include precipitation and adsorption to mineral and organic surfaces. Soils have a finite P sorption capacity, which varies widely as a function of clay content, clay type, organic matter content, the concentration of exchangeable Al, Fe, and Ca, and soil pH (Tisdale et al. 1985). Phosphorus sorption capacity increases with clay content, while sandy soils have the lowest P sorption capacity (Figure 2A). Soils high in Al, Fe, or Ca also have a high P sorption capacity (Figure 2B). Addition of organic amendments to soils can reduce the P sorption capacity because higher concentrations of organic acids can form complexes with P and limit the extent of P sorption (Figure 2C). The soil P sorption capacity affects how applied P will react and how much P will remain available for crop uptake or transfer into runoff or leachate.

Phosphorus sorption capacity is determined by equilibrating soil with a series of P solutions of increasing concentration and then determining the amount of P sorbed from solution to solid phase when equilibrium is reached. Results are then fit to a model that describes the soil's P sorption capacity. Knowing both the P sorption capacity of a soil and the current level of soil P allows us to estimate the degree of P saturation in a soil. The P saturation expresses the percent of the P sorption capacity that is filled (Breeuwsma and Silva 1992). A larger percentage of applied P remains as STP for soils with high P saturation.

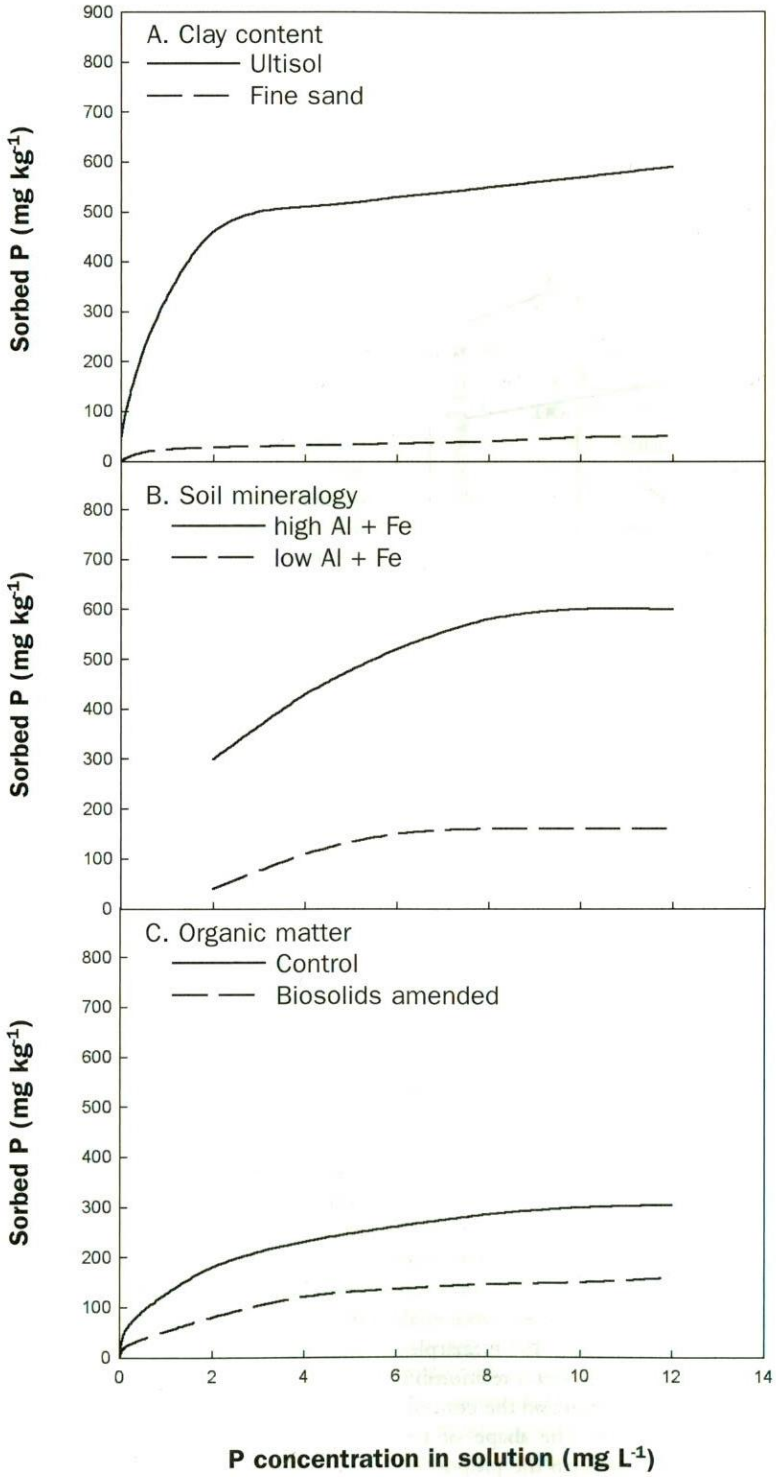
Determination of P sorption capacity and degree of P saturation have advantages for environmental risk assessment over STP methods because they are indicative of soil properties that control the release of P to waters (Hooda et al. 2000; Brooks et al. 1997; Sharpley 1995). But this type of analysis is not commonly performed in commercial soil testing laboratories, and methods of estimating P sorption capacity based on other soil properties have not been widely accepted. As a result, most nutrient management plans and environmental regulations are based on STP.

***Soil reaction to applied P.*** Phosphorus is added in a cropping system as fertilizer or manure to increase the quantity of plant available P forms. In the short term, much of the added P remains plant available and is measurable as STP. Over time, however, added P will be taken up by crops, react with other soil constituents to form insoluble minerals, or be sorbed onto mineral or organic surfaces. Because of these P fixing processes, the addition of fertilizer or manure P does not result in an equal (1:1) increase in STP. The ratio of the amount of P applied (kg P ha$^{-1}$) to the change in STP (mg kg$^{-1}$) for a soil depth of 15 cm (6 in) can range from 3:1 to >20:1 depending upon soil type, initial STP level, P application rate, and soil management history. For a calcareous clay loam soil, with initial Bray-1 P of 10 mg kg$^{-1}$ (ppm), this ratio was 30:1 when P fertilizer was applied at 25 kg P ha$^{-1}$ (22 lb ac$^{-1}$) and was 18:1 when fertilized at 50 kg P ha$^{-1}$ (45 lb ac$^{-1}$) (Randall et al. 1997). A similar soil had a ratio of 12:1 when the initial Bray-1 P was 30 mg kg$^{-1}$ (ppm) (Randall et al. 2000). These studies show that the change in STP with applied P depends upon the initial soil P level and on the application rate. The rate of change in STP also depends upon the degree of P saturation. When the P saturation is high, a larger percentage of the applied P remains in forms extracted by STP methods than for soils with a low percent of P saturation.

Decades of P fertilization at rates exceeding the amount of P removed by crops have resulted in elevated STP levels in many agricultural soils (Sharpley et al. 1994; Sims

**Figure 2**

Phosphorus (P) sorption isotherms illustrating differences in phosphorus sorption capacity. Graph A illustrates the effect of clay content by contrasting a high kaolinitic clay soil (Ultisol) and a sandy soil (adapted from He et al. 1999). Graph B illustrates the effect of soil mineralogy by comparing soils with different concentrations of iron and aluminum oxides (adapted from Brady and Weil 1999). Graph C illustrates the effect of biosolids application to an Iowa Mollisol (adapted from Sui and Thompson 2000).



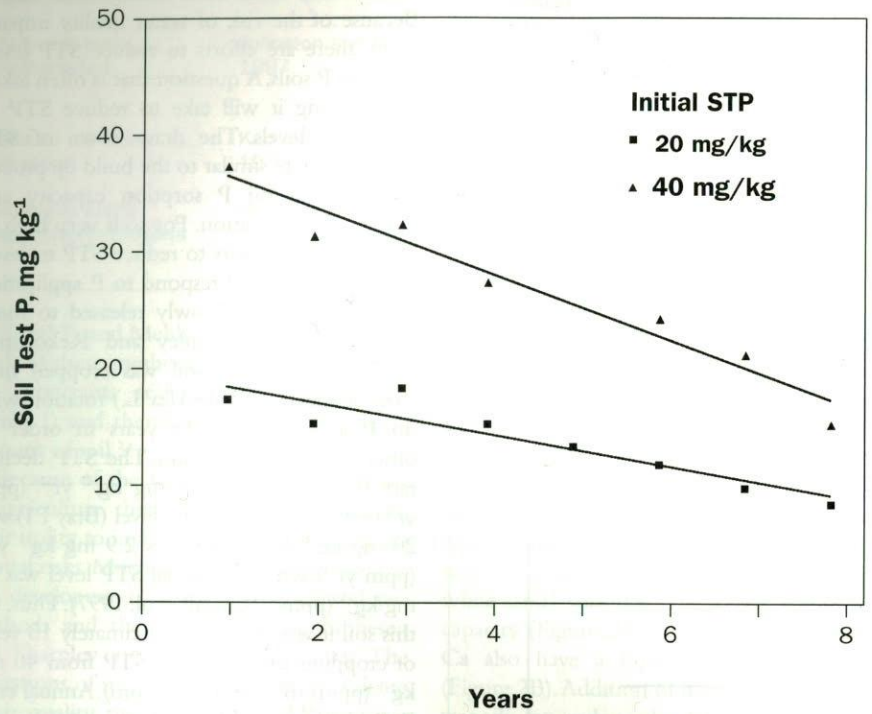

1992). The highest STP levels are generally found in fields where repeated manure applications have been made (King et al. 1990). Because of the risk of water quality impairment, there are efforts to reduce STP levels for high-P soils. A question that is often asked is how long it will take to reduce STP to acceptable levels. The draw down of STP occurs at a rate similar to the build up process and depends on P sorption capacity and degree of P saturation. For soils very high in P, it takes many years to reduce STP to levels where crops would respond to P application because stable P is slowly released to more available forms (Sharpley and Rekolainen 1997). A clay loam soil was cropped in a corn/soybean (*Glycine Max* L.) rotation with no P applied for eight years in order to observe STP decline rates. The STP decline rate (Figure 3) was 1.6 mg $kg^{-1}$ $yr^{-1}$ (ppm $yr^{-1}$) when the initial STP level (Bray P1) was 20 mg $kg^{-1}$ (ppm) and was 2.9 mg $kg^{-1}$ $yr^{-1}$ (ppm $yr^{-1}$) when the initial STP level was 40 mg $kg^{-1}$ (ppm) (Randall et al. 1997). Thus, for this soil it would take approximately 10 years of cropping to reduce the STP from 40 mg $kg^{-1}$ (ppm) to 15 mg $kg^{-1}$ (ppm). Annual crop P removal rates in the study were >8 kg $ha^{-1}$ $yr^{-1}$ (>7 lb $ac^{-1}$ $yr^{-1}$). Thus, removal rates of P in harvested crops (kg $ha^{-1}$ or lb $ac^{-1}$) are not directly equivalent to reductions in STP levels (mg $kg^{-1}$ or ppm).

## Phosphorus in Water

***Forms.*** Phosphorus originating from the soil can be transported in runoff water either in solution (dissolved P, DP) or associated with eroded soil particles (particulate P, PP). The relative proportion of DP and PP in water relates to the availability of that P for biological uptake and the potential for eutrophication. Dissolved P is available for rapid biological uptake, while only a portion of the PP is available. The portion of PP that is bioavailable can vary widely and depends on soil type, degree of P saturation, particle size, management history, and redox potential (Sharpley 1993; Sharpley 1997; Sharpley and Rekolainen 1997). Studies report that the percent of bioavailable PP ranges from 10 to 90%, with typical values around 20% (Sharpley 1993; Gburek and Sharpley 1998; Eghball et al. 2000; Eghball and Gilley 1999; Uusitalo et al. 2001).

The primary laboratory characterization of P in water is based on a separation of DP and PP forms. Dissolved P is operationally defined

**Figure 3**
Decline rates of soil test phosphorus (STP) for an Aastad clay loam soil in Minnesota with differing initial STP concentrations over an eight year period with no fertilizer P added. The site was managed in a corn/soybean rotation (modified from Randall et al. 1997).




as P that will pass through a 0.45 μm (1.8 x $10^{-5}$ in) diameter filter, while PP is retained on a 0.45 μm (1.8 x $10^{-5}$ in) filter. This definition is not physically based, as there is some P associated with solid, colloidal material with an effective diameter smaller than 0.45 μm (1.8 x $10^{-5}$ in). For simplicity, PP is usually calculated by subtracting DP concentration from the total P concentration determined by a harsh laboratory extraction of an unfiltered sample (i.e., acid digestion) (Pierzynski and Sharpley 2000).

Some laboratory methods have been developed to estimate the concentration of bioavailable P (BAP) in runoff. These methods are based on correlations between a chemical extraction of P in a runoff sample and an algal growth bioassay performed on the same water source (Sharpley 1993; Miller et al. 1978). The most common extraction technique used to estimate runoff BAP is insertion of an iron oxide impregnated strip into the water sample to extract a fraction of the P in the total sample. The iron-coated strips are removed from the water sample and P is extracted and measured (Sharpley 1993). As BAP techniques have extraction times of about one day, estimates represent the P concentration available for rapid biological uptake, and do not evaluate the amount of P that may become available from sediments over longer times. The long-term bioavailability of PP is less well defined and is a function of water body properties, such as residence time, depth, and surface area of the water body. More detailed explanations of P fractionation and terminology for water samples are found elsewhere (Haygarth and Sharpley 2000; Pierzynski and Sharpley 2000).

***Transfer of soil P to runoff water.*** The transfer of soil P to runoff water is a process occurring within a 1 to 5 cm (0.4 to 2.0 in) depth of soil (Sharpley 1985) and is controlled by physical and chemical processes such as desorption, dissolution, and diffusion. The concentration of DP in runoff is directly related to the quantity and reactivity of P near the soil surface. As STP concentration increases near the soil surface, the concentration of DP in runoff also increases (Pote et al. 1996; Romkens and Nelson 1974; Sharpley et al. 1996). Figure 4 illustrates a relationship between STP concentration and the concentration of DP in runoff. The shape of this curve indicates an increase in the proportion of STP being transferred to runoff water with increasing STP concentration. This effect occurs due to an increase in the degree of P saturation with increasing STP concentration. Thus, knowledge of P sorption capacity improves the use of STP for evaluating water quality risk. The relationship between STP and runoff DP concentrations is soil specific and is affected by physical, chemical, and management factors (Sharpley 1995).

The concentration of PP in runoff is not directly related to STP or soil total P concentrations because of its dependence on the concentration of eroded soil particles. The concentration of eroded particles varies widely from event to event, with different soil types and with different management practices. The concentration of P per unit mass of eroded particles (mg P $kg^{-1}$ or ppm) is related to the concentration of soil total P. When compared to the bulk soil, the concentration of total P in eroded soil particles is higher because the erosion process is selective of smaller, lighter, and more easily transported soil particles such as clay and low density organic particles (Sharpley 1985; Wall et al. 1996). These particles have a higher sorption capacity for P than the bulk soil. The increased P concentration in eroded soil particles relative to the bulk soil is called P enrichment, and enrichment ratios are used to represent the increase in P concentration of sediments relative to the parent soils. Enrichment ratios of Bray-1 P averaged 2.5 for various Oklahoma and Texas soils (Sharpley 1985) and ranged from 1.7 to 4.5 for cultivated soils in Illinois (McIsaac et al. 1991).

***Leaching of soil P.*** In many watersheds, runoff and erosion are the predominant pathways of soil P loss (Hanway and Laflen 1974; Randall et al. 2000). In these cases, subsurface P losses may be negligible because of the high P fixing capacity of subsoils. But the movement of P through the soil profile can be significant for soils that have very low P sorption capacity (low in clay, Fe and Al oxides, and carbonates). Leaching of P can also occur when STP is elevated from continuous application of organic wastes to 1) sandy soils, 2) acid organic soils, and 3) in soils prone to macropore flow. Although leaching of P is less likely in fine-textured mineral soils, downward movement in the profile has been reported when STP levels are high (Gächter et al. 1998; Heckrath et al. 1995; Hooda et al. 1999). The environmental implications of P leaching through the soil profile are greatest for shallow water tables or for soils with

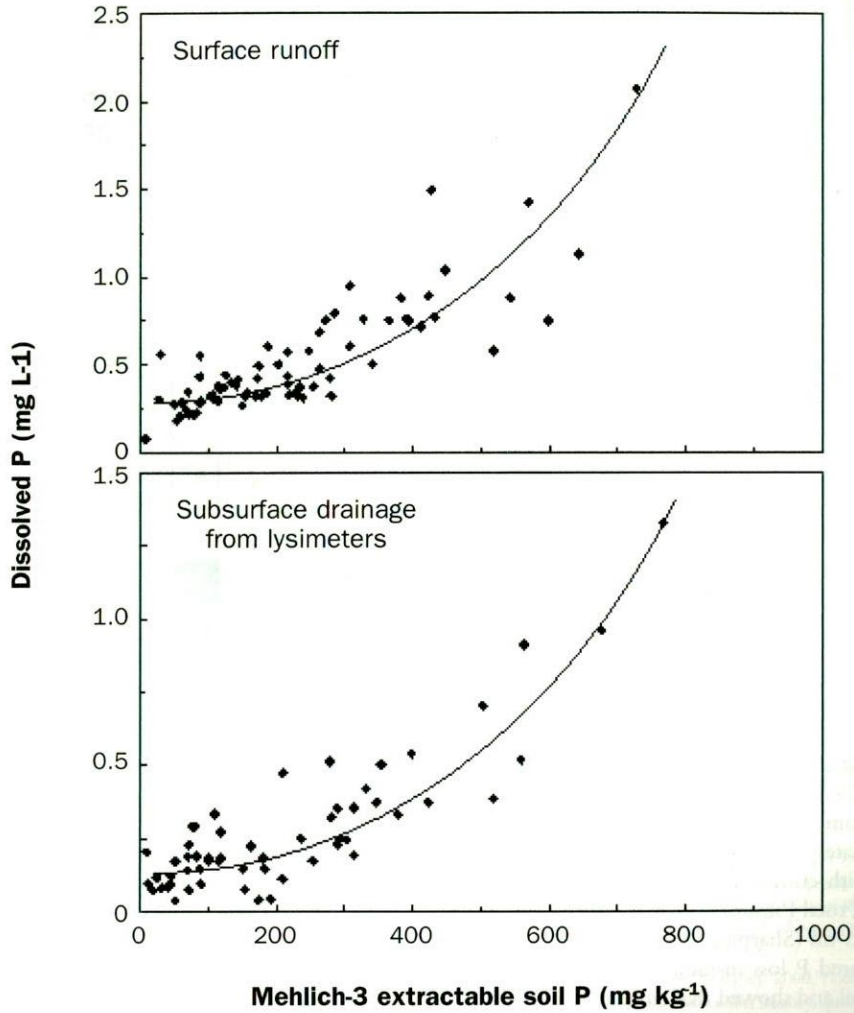
artificial drainage. In a P leaching study, the concentration of DP increased with increasing STP concentrations (Figure 4). This shows that the concentration of STP relates to the risk of P leaching, but its use is improved with knowledge of the P sorption capacity of the soil and the soil hydrology. The loss of P in agricultural drainage was recently reviewed by Sims et al. (1998).

***Direct losses of applied P.*** In some situations, the direct transmission of applied P materials to runoff or loss by preferential flow in leaching can be more significant than the transfer of soil P. Direct P losses are important when runoff follows shortly after surface application of P fertilizer or manure. When such an event occurs, the losses of P can represent up to 10% of the amount of P applied (Sharpley and Rekolainen 1997; Baker and Laflen 1982; Edwards and Daniel 1992; Carpenter et al. 1998). The risk of direct transmission losses for subsequent runoff events is much less than for runoff from the first rain after application (Edwards and Daniel 1994). The risk of direct transmission losses is largely a function of management decisions such as the timing of application, the source of applied P, and the application method and tillage practice. Injection or incorporation of applied P fertilizer or manure dramatically reduces the risk of direct P loss in runoff.

Several studies have documented the loss of applied P through leaching and tile drain flow. Phosphorus from heavy manure applications to an Atlantic Coastal Plain soil accumulated and moved into shallow groundwater (Novak et al. 2000). In Scotland, significant concentrations and losses of P were found in tile-drained pastures after application of cattle slurry. The rapid appearance of P after manure application was indicative of preferential flow (Hooda et al. 1999). Gaynor and Findlay (1995) evaluated P loss in tile drainage from a Brookstone clay loam. Loss of DP in tile drainage averaged 0.9 kg $ha^{-1}$ (0.8 lb $ac^{-1}$) under no-till corn production, representing 7% of applied P. Extensive soil cracking, and lower losses with conventional tillage suggested that preferential flow was responsible for P movement to the tile lines.

**Figure 4**

The relationship between the concentrations of dissolved phosphorus (P) in surface runoff (upper graph) and subsurface drainage (lower graph) from 30 centimeter deep lysimeters and the Mehlich-3 extractable P concentration of surface soil (0 - 5 centimeters) from a central Pennsylvania watershed (adapted from McDowell and Sharpley 2001).

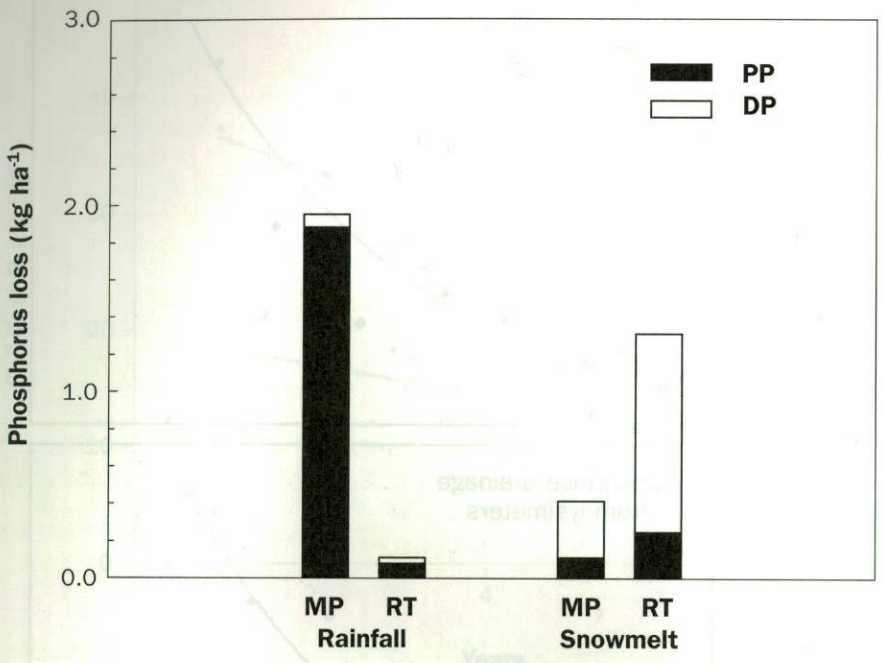



## Management Practices To Reduce P Loss From Agriculture

Reducing the movement of agricultural P to surface waters is best accomplished through the combined management of P sources in the agricultural environment and the factors that control the transport of P in the landscape. Management practices implemented to reduce P movement have the largest effect when they are used at locations that have a high potential of impairing water quality. Site risk assessment tools have been developed to assess the risk of off-site P movement from fields or watersheds (Birr and Mulla 2001; Gburek et al. 2000, Lemunyon and Gilbert 1993). These site risk assessment tools are based on the evaluation of P source and transport factors. In addition to identifying sites with high risk of P movement to surface waters, these tools help identify site-specific management practices that will reduce P loss (Eghball and Gilley 2001; Sharpley et al. 2001).

***Management practices to reduce P transport.*** Management practices that reduce erosion and runoff are effective at reducing P loss from cultivated agricultural soils. Conservation tillage systems reduce sediment and associated PP losses when compared to tillage approaches that leave little crop residue on the soil surface (Ginting et al. 1998; Hansen et al. 2000b). There can be significant concentrations of DP in runoff from reduced tillage systems because P from fertilizers and crop residues accumulates near the soil surface, where it can be transferred to runoff. In many cases, a low runoff volume from reduced tillage soils compensates for the

**Figure 5**
Phosphorus (P) loss in runoff caused by either rainfall (Hansen et al. 2000a) or snowmelt (Hansen et al. 2000b) from a tillage study in Minnesota. The tillage practices compared are fall moldboard plow (MP) and ridge tillage (RT). Each bar is divided into dissolved P (DP) and particulate P (PP) fractions and the combined bar is equal to total P loss. The data represent average annual losses over a two year period.




accumulation of P near the surface. In the U.S. Southern Plains, loss of BAP in runoff from agricultural watersheds was approximately three times less with no tillage than with conventional tillage but the proportion of total P found in BAP forms was higher for no-till (Sharpley 1993). Another study compared P loss in rainfall runoff from a high P soil and showed that ridge tillage resulted in a sixfold reduction in total P loss and a twofold reduction in DP loss compared to soils that were moldboard-plowed, despite higher STP near the soil surface (Figure 5)(Hansen et al. 2000a). These studies suggest that controlling erosion in cultivated systems is a high priority for reducing P losses in runoff. In cold regions, snowmelt runoff can be an important source of P loss from agricultural systems (Hansen et al. 2000b). The effects of tillage on the loss of P in snowmelt runoff can be opposite to the effects observed for rain-induced runoff. When soil frost limits erosion from snowmelt runoff, DP loss predominates. Losses can be higher for reduced tillage systems because they retain more snow and because of the accumulation of P near the soil surface (Figure 5).

Additional reduction of PP transport can be obtained by routing runoff through vegetative buffers, riparian zones, or structures that reduce the concentration of PP through deposition. Routing runoff through vegetated buffers can reduce total P in runoff by as much as 90%, depending upon factors such as buffer length, runoff rate, and the type of buffer (Chaubey et al. 1995). Constructed wetlands and ponds can also reduce the concentration of P in water (Uusi-Kamppa et al. 2000). Buffers and other erosion control practices are more effective at reducing PP concentration than DP concentration in runoff (McDowell and McGregor 1984).

***P source management.*** Managing the sources of P in agricultural systems can reduce losses of agricultural P to surface waters. Comprehensive Nutrient Management Plans (CNMPs) have become an essential component of establishing P management strategies in agricultural production systems (Beegle et al. 2000). Effective CNMPs evaluate all P sources external to the system and the redistribution of P within a system and consider such practices as P application rates, times, and methods. When used together with P site risk assessment tools, CNMPs can minimize build-up of P in sensitive areas. Comprehensive nutrient management relies on soil analysis in order to account for available P in the soil and considers all forms of P imported to the farm operation, including fertilizers, manure, and the quantity and forms of P in livestock feeds. Farm operations that closely balance P imports and P exports will ultimately have a lower potential for creating water quality problems than farm operations that accumulate P on the farm.

Nutrient accumulation is a challenge for confined animal feeding operations because of the concentration of nutrients imported in feeds. When adequate land is not available for manure application, or when hauling and spreading costs limit the distribution of manure, P accumulation occurs and the risk of water quality degradation is elevated. The situation can be improved by formulating animal diets to avoid overfeeding of P and by improving P digestibility (Poulsen 2000; Valk et al. 2000). Manure treatment, mechanical processing, or composting can be used to increase the value of manure and decrease the economic limits on hauling distance. New technologies are also being developed to explore alternative uses for livestock manure such as energy production or as a source for extracting and recycling P (Mikkelsen 2000). These technologies may prove important for balancing P on some farms, but land application will remain manure's most common use and its value should not be overlooked.

Rate, method, and timing of P applications to soils are important P source management decisions from production and water quality perspectives. Surface applied P fertilizer or manure without incorporation can create a large risk of direct P transmission to surface water, especially when runoff follows shortly after application. In this situation, the magnitude of P loss increases with application rate (Figure 6). The proportion of applied P that is lost depends on the form of P applied. In Figure 6, note that P concentrations were lower in runoff when poultry slurry was applied than when poultry litter or inorganic fertilizer were applied. This occurred because the application of liquid slurry allows for more infiltration and reaction of applied P with the soil than for the other sources. The extent of P loss in runoff decreases as the time between application and runoff increases (Sharpley 1997). Thus, runoff P can be reduced by avoiding excessive P application rates in sensitive areas and by avoiding

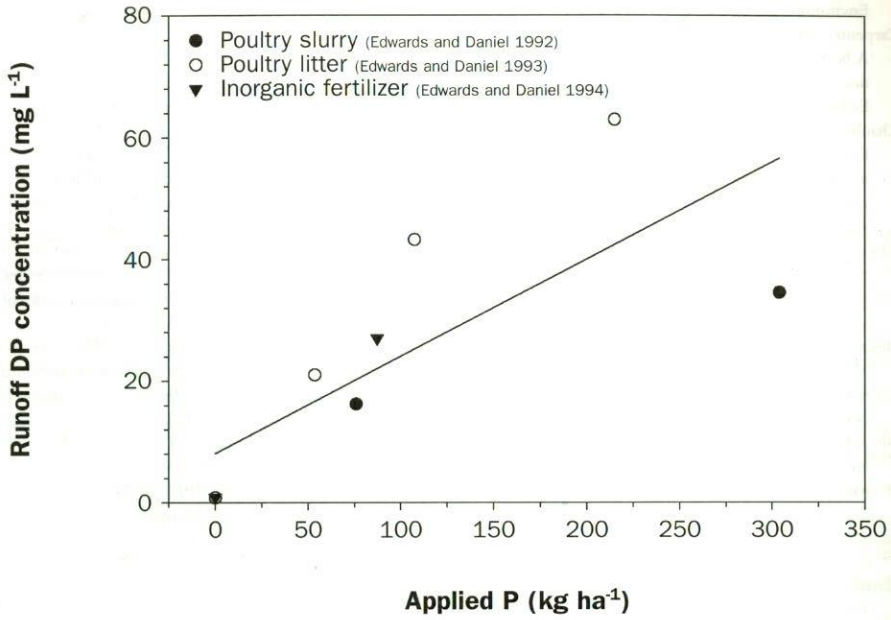
applications during periods of high rainfall probabilities.

Incorporation of applied P fertilizer or manure can reduce the concentration of DP in runoff that immediately follows application by as much as fourfold, compared to broadcast application without incorporation (Bundy et al. 2001). But tillage used to incorporate applied P can increase the risk of PP loss from erosion due to the reduction in residue cover. Thus, there is a conflicting need to incorporate applied P to reduce direct transmission losses while minimizing soil disturbance to reduce erosive losses of PP. For erosion prone soils, practices that control erosion should be highest priority. Injection of fertilizers or liquid manures is a means of maintaining residue while placing P below the soil surface. Specialized equipment can be used to apply P fertilizer in a band below the soil surface in the fall or as part of the seeding operation. In situations such as permanent pasture, where neither incorporation nor injection of applied P fertilizer or manure is feasible, other management practices should be used to reduce the risk of off-site transport. Runoff loss of P from recently applied manure can be reduced by chemically treating the manure to reduce P solubility. Phosphorus concentration in runoff from fields after application of poultry manure was reduced by a factor of three when the manure was treated with aluminum sulfate ($Al_2(SO_4)_3 \cdot 14H_2O$) compared to untreated manure (Moore et al. 2000).

Soil sampling and CNMPs are useful tools in identifying appropriate site-specific P management strategies. Special attention should be paid to sites identified as having a high risk of P movement to surface waters. Combinations of P source and transport management will improve water quality and help maintain farm profitability.

**Figure 6**

Comparison of the concentration of dissolved phosphorus (DP) in runoff to the rate of applied P from different sources. All sources were broadcast applied to pasture 24 hours prior to simulating a 5.0 cm $hr^{-1}$ rainstorm.




## Summary and Conclusions

Soil P is both an important crop nutrient and a primary water contaminant. The STP extraction methods commonly used for crop fertilizer recommendations are now being used to help accomplish water quality goals. But STP concentrations do not directly relate to the soil total P concentration or to the degree of P saturation. Knowing these additional parameters about soil P will allow us to better understand the soil specific environmental risks and to recommended P management practices. Incline and decline rates of STP concentration depend on the P sorption capacity of the soil. It can take many years to reduce P concentration in soils with a high STP concentration.

Runoff and erosion from agricultural soils are important mechanisms of P loss from land to water. While DP is immediately available to aquatic organisms, only a portion of PP is available in the short term. Dissolved P in runoff originates from soil P near the surface, from crop residues, and from direct transmission of surface applied fertilizer or manure. Particulate P in runoff originates from eroded soil particles and is enriched in P concentration relative to the parent soil.

In many watersheds, subsurface losses of P can be a significant concern. Leaching of P is most likely to be a concern on soils with a low P sorption capacity and with shallow groundwater or artificial drainage. Phosphorus is most likely to move downward in soils with a high degree of P saturation, usually as a result of frequent application of organic wastes. Recently applied P can also move in soil through preferential pathways.

Water quality improvement is best achieved by identifying sites with a high potential of P movement to surface water and then implementing management practices to reduce P losses from those sites. Site risk assessment tools like the P index concept are effective at site identification and can help identify site-specific management practices to reduce P loss. Management practices implemented to reduce P loss from agriculture should address both P source and transport factors.

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# Nitrogen modeling for soil management

M.J. Shaffer

**ABSTRACT:** Simulation models of the nitrogen (N) cycle have been used for well over 20 years to help estimate nitrate ($NO_3$-N) leaching, soil residual $NO_3$-N, fertilizer N requirements, soil organic N status, and gaseous N emissions associated with agriculture. These models have been coupled with simulations of other related processes such as water and solute transport, crop growth, soil chemistry, temperature regimes, and management to make more complete models of cropping systems. At the core of these tools have been databases for soils, climate, model coefficients, and field/farm/watershed management scenarios. This paper reviews the basic types of N models, modeling techniques, and required databases. The accuracy of N models along with their strengths and limitations are discussed in a management context. Tips are provided on initializing N constituent pools, on using N models in Geographic Information Systems (GIS) applications, on developing confidence bands for N model output, and on using web-based N models. Finally, methods are described to analyze simulated $NO_3$-N leached, nitrous oxide ($N_2O$) emissions, and N use efficiencies.

**Keywords:** C/N cycling, fertilizers, manure, $N_2O$ emissions, nitrate leaching, nutrient management, water quality

**Management of nitrogen (N) in agricultural settings has become critical for maintaining crop yields while controlling environmental pollution from nitrate ($NO_3$-N) leaching, nitrous oxide ($N_2O$) greenhouse gas emissions, ammonia ($NH_3$) volatilization, and surface runoff of nutrients.** The increased initial cost of N to producers has further emphasized the need for good on-farm N management. The availability and use of models in N management has increased rapidly in the past 20 years worldwide (Shaffer and Ma 2001). Recent reviews have identified at least ten models in North America and eight models in Europe capable of simulating N cycling in cropped agricultural systems (Ma and Shaffer 2001; Grant 2001; McGechan and Wu 2001). These tools are generally process-based and range in complexity from moderately detailed to highly complex.

## The Modeling Process in Soil N Management

Nitrogen cycling processes in soils are mediated primarily by microorganisms, with the greatest activity taking place in carbon-rich layers such as the A horizon (Shaffer and Ma 2001). Because carbon (C) and N are so closely linked in soils, they are almost always simulated together in process-based models. Carbon/nitrogen (C/N) models simulate and link the major N cycling processes as a function of state (or driving) input variables (Figure 1). Nitrogen cycling processes include mineralization-immobilization of crop residues and other organics, mineralization of soil organic matter (SOM), nitrification of ammonium ($NH_4$), $NO_3$-N leaching, $NH_3$ volatilization, denitrification of $NO_3$-N, and N fixation (Shaffer and Ma 2001). To be effective in real-world applications, C/N cycling models also must be connected with models for crop growth, water and solute transport, soil chemistry, temperature, and management as well as other factors to make a complete soil-crop system (Figure 2).

Because soil C/N processes are complex and are often controlled by several driving variables, this system must be modeled with care and from a sound scientific footing (Ma

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