Mechanisms for release of sediment-bound phosphate to water and the effects of agricultural land management on fluvial transport of particulate and dissolved phosphate

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Abstract

Sediment-bound phosphate includes organic and inorganic forms, but the inorganic fraction contains most of the P that can be released into water. The non-apatite inorganic P (NAIP) fraction of sediment-bound phosphate varies considerably with geochemistry and anthropogenic additions (fertilizer, livestock wastes, P adsorbed from municipal wastewater discharges). A small fraction of the NAIP is sufficiently labile to desorb into water, and this release can be described by dissolution or adsorption/desorption mechanisms. Agricultural practices, such as phosphate fertility management and conservation tillage, which affect the levels of phosphate and sediment leaving the land, will determine the partition of sediment-bound P and dissolved P in water draining into lakes, with implications for the availability of that phosphate to phytoplankton.

Introduction

In most instances, the bulk of the phosphate load to lakes is from land drainage, and includes varying proportions of sediment-bound P and dissolved P. P sources which may enter the lake through land drainage include native soil phosphate, fertilizer P, and livestock and municipal wastes. These sources exist in various physical and chemical states (sediment-bound and dissolved, organic and inorganic, etc.), but in the process of transport to the lake, they interact physically and chemically to produce, at the tributary mouth, a discharge which contains some ratio of sediment-bound P to dissolved P. The size of this ratio, or partition coefficient, and the lability of the sediment-bound P will greatly influence the response of phytoplankton to the P load.

This review will examine the mechanisms by which inorganic phosphate partitions, between particulate and dissolved phases; and the impact of agricultural land management practices, on dissolved and particulate phosphate in drainage waters.

Forms of phosphate in soils and sediments

Phosphorus exists in many forms, and there is much confusion among scientists as to the proper use of terms describing these forms. Much of the confusion results from the methods used to measure P forms and the interpretations that are made about P analyzed by different procedures.

Phosphorus can be classified by physical state and chemical type (Logan 1980). Although P can exist in gaseous form, this is not a common occurrence in nature, and the solid and liquid states are the only ones that need to be considered.

Phosphorus in the solid state has been referred to as: particulate, sediment-P and sediment-bound P, especially in relation to fluvial systems. Since some of the particulate matter in streams and lakes may be detrital, the term particulate P may be less misleading than sediment-P or sediment-bound P which commonly carry the connotation of mineral matter. Particulate P can be composed of many chemical forms, and these may be expressed in a number of mineralogical and sorbed states. The

following organization is not inclusive, but does include those forms most likely to be environmentally significant. Since much of the sediment in fluvial systems is of pedologic origin, these forms of particulate phosphate are consistent for soil and sediment systems:

Adsorbed: labile P; exchangeable P

Organic: various forms including phytins (inositols)

Precipitates: primarily in soil; fertilizer reaction products with Ca, Fe, Al and other cations; varying water solubility

Minerals: amorphous, short-range order and crystalline minerals with Ca, Fe, Al and other cations

Researchers in soil science (Chang & Jackson 1957; Williams *et al.* 1967) and, more recently sedimentology (Williams *et al.* 1971a, b, 1976a, b), have attempted to use chemical extraction to quantify the different forms of particulate P in soils and sediments. Early fractionation schemes (Chang & Jackson 1957) partitioned particulate P into: Fe-P, Al-P, Ca-P, occluded-P and organic P, but recent revisions (Williams *et al.* 1971a, b) separate particulate phosphate into: (Fe + Al) bound-P, Ca-P (apatite), reductant soluble-P and organic P. The latter scheme has been used for sediments (Williams *et al.* 1971a, b), and a simplified version has been proposed (Williams *et al.* 1976a, b), which separates particulate P into: non-apatite and apatite inorganic P, and organic P.

Implicit in the development of the various sequential chemical extraction schemes is the assumption that a specific chemical extractant can selectively remove a particular phosphate form. Given the great complexity of phosphate chemistry and mineralogy in soils and sediments, it is not surprising that this assumption is highly tenuous, and in recent years, workers (Williams *et al.* 1971a, b) have begun to use operational definitions which require no judgment as to the selectivity of the extractant, but yet provide significant information about the chemistry and mineralogy of particulate phosphate.

The separation of particulate and liquid phases of water samples is somewhat arbitrary and rarely complete. The standard separation procedure is filtration through 0.45 μm pore diameter membrane filters (APHA 1975), at which point P in the liquid phase is operationally defined as *dissolved.* However-

er, some fine colloidal materials are smaller than 0.45 μm. In the determination of orthophosphate, by the reduced phosphomolybdate colorimetric procedure, the sample comes in contact with strong acid which can extract or dissolve P from particulate forms and, therefore, dissolved orthophosphate may be overestimated.

Much water quality monitoring involves determination of *total P* by digestion of an unfiltered sample. Rigorous digestion with strong acid (perchloric, nitric-perchloric, sulfuric, aqua ragia) will give quantitative recovery of P from mineral and organic forms in soils, sediments and water samples, but these procedures require special laboratory facilities and must be carefully performed. They are unnecessarily rigorous for water samples containing low concentrations of mineral particulate matter. Persulfate digestion (APHA 1975) is routinely used for total P analysis of water and wastewater; but, several researchers (O'Connor & Syers 1975; Logan *et al.* 1979) have found that persulfate digestion can underestimate total P by as much as 20%, compared to strong acid digestion in water samples containing appreciable mineral sediment.

Figure 1 schematically describes the partition of phosphorus in a tributary load during a storm event, in a watershed with a significant proportion of agricultural land use. It is based on a number of research studies in the Lake Erie drainage basin (Corps of Engineers 1979), as well as established principles of the nature of soil and sediment phosphate. The size of the boxes correspond to the relative size of the phosphate pools in this generalized view, and the partition of particulate and dissolved P will depend greatly on the extent of erosion and sedimentation in the drainage basin. The values given in Fig. 1, for bioavailability of P forms, are based on a broad range of research in recent years and these have been reviewed by Lee *et al.* (1980).

Labile phosphate in soils and sediments

As indicated in Fig. 1, the labile P fraction of soils and sediments constitutes a small part of the total particulate P. It includes phosphate that is adsorbed, easily hydrolyzed or easily dissolved. In sediments in contact with water, most of the labile P is adsorbed with varying binding energies as dis-

P Fraction		Chemical Characteristics	Percent Bioavailability
Total P	Total dissolved P	Dissolved inorganic P	> 90
		Dissolved organic P	< 50
	Total particulate P	**Labile P** Includes adsorbed, exchangeable, easily dissolved, easily hydrolyzed	Potentially — 100 Immediately — ?
		Inorganic P Relatively stable, primary and secondary minerals with Fe, Al, Ca.	Potentially — ? Immediately — 0
		Organic P Relatively stable humus compounds such as inositols	Potentially — ? (<50) Immediately — 0

Fig. 1. Particulate and dissolved phosphate forms in soils and sediments.

cussed later. Definitions of labile P in soils or sediments are not rigorous, and are usually experimentally derived. Isotopic exchange with ^{32}P has been used in soils (Watanabe & Olsen 1954) and sediments (Li *et al.* 1972), while anion exchange resin (Amer *et al.* 1955) and Al-saturated cation exchange resin (Huettl *et al.* 1979) have been used to estimate labile P. Water extraction has been used by Sharpley *et al.* (1981), and dilute electrolyte by Oloya & Logan (1980) and Sharpley *et al.* (1978). McDowell (1980) and Johnson *et al.* (1976) used extrapolation from adsorption/desorption isotherms to estimate labile P. Because of the empirical nature of these procedures, they must be used in statistical models to determine their relationship to P release in water.

Phosphate adsorption/desorption

The adsorption/desorption of phosphate by soils and, more recently, by sediments has been the subject of extensive research. Much of this work has involved batch equilibration of soil or sediment with solutions containing varying P concentra-

tions. The resulting adsorption/desorption isotherms have been described by existing models such as the Langmuir or Freundlich equations, or fitted to empirical functions. Adsorption/desorption models can either be equilibrium or nonequilibrium (kinetic).

Equilibrium models

The simplest equilibrium model states that there is, at equilibrium, some partition between dissolved and labile P. This can be expressed as:
as:

$$S = KC^n \qquad (1)$$

where S = P content of soil or sediment (μg P g^{-1}),

C = dissolved inorganic P concentration (μg P ml^{-1}), and

K, n = constants, with K equal to the partition coefficient, K_D, when n = 1

Equation (1) is essentially the same as the Freundlich adsorption isotherm. The exponent n for phosphate is usually <1, and this prevents use of the simplified function, $\frac{S}{C} = K_D$, where n = 1.

The Freundlich adsorption isotherm has not been used to the same extent as the Langmuir isotherm, which owes its popularity to the adsorption maximum, S_{max}, which can be derived from it. Use of the Langmuir adsorption isotherm to describe the equilibrium between labile P from soil or sediment and dissolved P has been extensively studied, and a review of that literature is not possible here. Recent reviews by Barrow (1978) and Rao et al. (1980) effectively discuss the uses and limitations of the Langmuir model. Rao et al. (1980) rearranged the Langmuir equation to give an estimate of the equilibrium partition coefficient, K_D:

$$K_D = \frac{S}{C} = \frac{kS_{max}}{1 + kC} \qquad (2)$$

Equation (2) indicates that K_D varies with the equilibrium phosphate concentration, C, and this would preclude use of a single K_D value for modeling purposes. Rao et al. (1980) showed that if $kC \lll 1$, then equation (2) simplifies to:

$$K_D = kS_{max} \qquad (3)$$

where K_D is independent of C. Equation (3) is the same as the linear Freundlich adsorption isotherm.

Some workers (Syers et al. 1973; Holford et al. 1974) have developed two-component Langmuir adsorption isotherms to handle the non-linearity of single isotherms that is often found when the adsorption experiment is carried out over a wide range of initial dissolved P concentrations. For the purpose of predicting dissolved P in runoff, the low end of the concentration range should be used (0-5 μg P ml^{-1}) if a single K_D value is to be derived by using equations (2) or (3).

Use of K_D to model particulate and dissolved phosphorus partitioning assumes that K_D is an intrinsic property of a soil or sediment and, once determined, can be applied wherever that soil or sediment is found. However, such factors as labile P can greatly affect the adsorption/desorption characteristics of a soil or sediment (Bache & Williams 1971; McCallister & Logan 1978). Presently there are no satisfactory equilibrium models which relate K_D to labile P; however, Sibbesen (1981) obtained a good fit of soil P adsorption data to an extended Freundlich isotherm, which contains an exponent related to labile P.

The difficulty in determining the effect of labile P on P equilibrium is due to the difference in rate of desorption of previously adsorbed P and the rate of initial adsorption. The slower rate of desorption compared to adsorption means that K_D values or equilibrium dissolved P concentrations predicted from adsorption isotherms may not accurately predict conditions where previously adsorbed phosphate is desorbed into water.

Another equilibrium model approach is to determine the equilibrium P concentration (EPC$_0$) at zero net adsorption or desorption from an adsorption isotherm. The EPC$_0$ gives an estimate of the dissolved P concentration. This approach does account for the effect of previously adsorbed P, and has been used by several researchers (Taylor & Kunishi 1971; McDowell et al. 1980; McCallister & Logan 1978).

Non-equilibrium kinetic models

The equilibrium modeling approaches just discussed assume that (1) adsorption/desorption reactions are rapid, and (2) that there is sufficient soil or sediment to buffer the dissolved P. Li et al. (1972) found that isotopic exchange of ^{32}P with sediment P was rapid (minutes), and Oloya & Logan (1980) found that there was near-instantaneous removal of a large fraction of P in soils and sediments by dilute electrolyte. They also found, however, that soils high in labile P also released some P more slowly, i.e. desorption could be described by first-order kinetics. Several researchers (Rajan & Fox 1975; Griffin & Jurinak 1974; Ryden et al. 1977) have described an initial rapid (hours) reaction of P with soil or soil materials followed by a slower reaction over weeks and months. In terms of land runoff events, the slower reaction may not be significant. Adsorption/desorption reaction rates are affected by sediment concentration or soil: solution ratio (Hope & Syers 1976) and labile P (Barrow 1974; Oloya & Logan 1980).

A number of kinetic models have been proposed for adsorption/desorption by soils and soil materials, and many of these attempt to describe the slow and fast reactions by various techniques. Multiple first order equations were used by Amer et al. (1955) to describe P desorption from soil to resin, and Evans & Jurinak (1976) described P release from desert soil by three simultaneous first-order

equations. Barrow (1980) has recently developed an empirical relationship for P adsorption based on the Freundlich isotherm. By an iteration procedure, adsorption can be determined as a function of previously adsorbed P, soil: solution ratio, dissolved P concentration and time. This method was successful in describing P adsorption by some noncalcareous U.S. soils. Several of the various kinetic models have been reviewed by Rao *et al.* (1980).

In land runoff, partitioning of particulate and dissolved P is in the direction of desorption, i.e. the chemical potential of phosphate on the soil or sediment surface is usually higher than that in rainfall. However, most of the kinetic models studied have focused on P adsorption by soils or sediments, and often with high initial P concentrations. Desorption rates have been shown to be much slower than initial adsorption rates, and present models do not satisfactorily describe desorption. However, Sharpley *et al.* (1981) have recently developed an empirical kinetic model for P desorption which accounts for labile P, soil: solution ratio and time. Their equation is:

$$P_d = KP_0 t^\alpha WS^\beta \tag{4}$$

where $P_d = $ P desorbed, $P_0 = $ desorbable P at $t = 0$, $t = $ time, WS $ = $ water: soil ratio and K, α and β are constants. They found that experimentally derived values of K, α and β could be used to describe P desorption for a given soil over a range of values of P_0, WS and t. P_0 was estimated by water extraction, but may be correlated with other methods for determining labile P. As Sharpley *et al.* (1981) have indicated, equation (4) is similar to other exponential equations such as the modified Freundlich equation of Barrow (1980) and the Elovich equation as used by Chien & Clayton (1980).

The practicality of using equation (4) will depend on the variation of the K, α and β constants and the need to measure them experimentally for each soil or sediment.

## Transport of particulate and dissolved phosphate in land drainage

In an agricultural watershed, the land surface is a point of concentration of phosphate, as a result of fertilizer and manure applications. As runoff drains from the land to the stream channel and ultimately to the lake, there is usually a progressive dilution of the total phosphorus load through deposition of sediments and water dilution. At points in the stream, concentrated or 'point' discharges of P from municipal or other sources may elevate P concentrations at their point of entry. The interaction of stream suspended sediments with the P load will be discussed later.

In Fig, 2, the processes by which particulate and dissolved P are removed from the land surface are conceptualized. The following events occur during a runoff-producing rain:

1. Precipitation strikes the soil surface and a fraction infiltrates while another portion runs off.
2. Part of the water that infiltrates continues to percolate downwards and part reacts with only a shallow zone of soil, below the surface, before leaving the field in the runoff. This zone of runoff interaction is probably <1 cm (personal communication with Dr. Andy Sharpley, USDA-AR, Durant, Oklahoma).
3. Precipitation striking the soil surface dislodges soil particles and runoff carries some of the eroded soil downslope with only a fraction actually leaving the field as sediment. During the erosion and sediment transport process, there is a selective removal of clay-sized mineral particles and organic matter, and both of these materials contain higher levels of phosphorus than the coarser sand and silt fractions. As erosion is reduced by conservation tillage (e.g. no-till), the sediment that is lost from the field becomes progressively more enriched in clays and organic matter and the phosphorus enrichment ratio increases (Fig. 1a). Therefore, the percentage reduction in total particulate P (TPP) with conservation tillage will be less than the percentage reduction in soil loss.
4. As water reacts with the soil surface (the zone of interaction), dissolved P held in soil pores is removed, water soluble soil phosphorus compounds are dissolved, and some of the inorganic P sorbed on soil surfaces is desorbed into the water. Also, some dissolved P is contributed from decaying crop residues, especially in no till systems.
5. The final concentration of dissolved P in runoff leaving the field will be determined by the equilibrium between dissolved P and the sediment,

PHOSPHORUS TRANSPORT IN RUNOFF
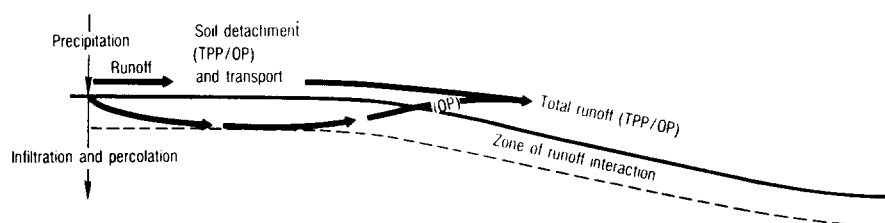
Total Particulate Phosphate (TPP) Transport

Selective transport of clay and organic matter in runoff gives sediment that is enriched in phosphate; proportion of clay in sediment increases as soil loss decreases, which increases phophorus enrichment (Figure a).
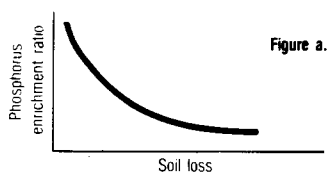
Soluble Phosphate (Orthophosphate, OP) Transport

Soluble P in runoff includes P held in soil pores in the zone of runoff interaction, dissolution of soluble P compounds in the soil, P leached from decomposing surface plant residues and desorption of P adsorbed on soil particles. Soluble P equilibrates with sediment in runoff by adsorption/desorption (Figure b).
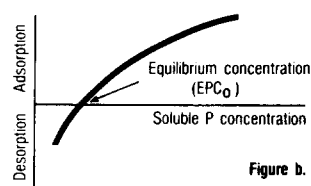
Fig. 2. A conceptual view of particulate and dissolved phosphorus in runoff from agricultural land (Logan 1981).

and the characteristics of the equilibrium iso-therm (Fig. 2b) are dependent on such properties as fertilizer use, soil loss, soil chemical character-istics, etc.

While the equilibrium approach to estimating the particulate and dissolved P partition coefficient has been successfully used by McDowell *et al.* (1980), Sharpley *et al.* (1981) have shown that at low sedi-ment concentrations in runoff, as might be found with no till crop management, equilibrium between the sediment and dissolved P is probably not at-tained during the event, and a kinetic model such as Equation (4) might be more appropriate.

The runoff leaving the field undergoes significant processing during its transport through the stream system to the lake. Sediments may be deposited in the stream for varying periods during which time they may become anoxic (McCallister & Logan 1978), with resulting changes in their chemistry. Upon resuspension and reoxygenation, the chemis-try of the sediments is altered again. Suspended sediments from upland sources may interact with concentrated dissolved phosphate 'point' dis-charges. Taylor & Kunishi (1971) reported a rapid adsorption, by upstream suspended sediments, of a

concentrated P discharge from a swine feedlot in rural Pennsylvania. Verhoff (1982) reported on the deposition and resuspension of stream sediments during a storm event in the Sandusky River above and below a sewage treatment plant discharge. The dissolved P from the discharge appeared to be ad-sorbed by the sediments, but biological uptake could also account for part of the removal from the water column.

McCallister & Logan (1978) and Green *et al.* (1978) studied the phosphate characteristics of agricultural soils and suspended and bottom stream sediments, in the Maumee River Basin of Ohio. Their findings are summarized in Fig. 3. The clay fractions of the surface horizons of the agricultural soils were enriched in total P, and the stream bottom sediments showed a similar enrichment. However, the suspended sediments gave a much higher en-richment than would be predicted from the clay fraction enrichment, and it is assumed that the sus-pended sediments have adsorbed additional dis-solved P from point sources in their passage through the stream system. The EPC_0 and P de-sorbed parameters both reflect the labile P status of the sediments and these are both high for the sus-

Phosphorus adsorption/desorption characteristics of Maumee River Basin soils, their clay fractions and suspended and bottom sediments
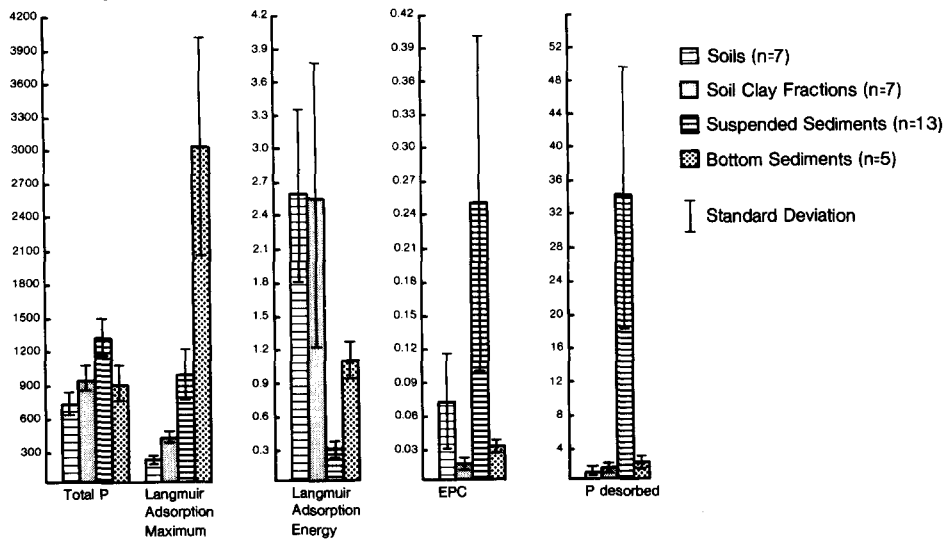

Fig. 3. P adsorption/desorption characteristics of Maumee River Basin soils, their clay fractions, and suspended and bottom sediments (Green *et al.* 1978).

pended sediments. The low Langmuir binding energy parameter of the suspended sediments also indicates that the adsorbed P is weakly held. The high Langmuir adsorption capacities of the bottom sediments were attributed to the fresh iron oxide surfaces formed when these previously anoxic sediments were reaerated during analysis. This condition, however, is not unrealistic for resuspended bottom sediments.

The effects of agricultural land management on the transport of particulate and dissolved phosphate in runoff

Intensive agricultural production is a major source of the tributary phosphorus load to lakes from watersheds with a high agricultural land use (Corps of Engineers 1979). Since particulate, or sediment-bound, phosphate constitutes a large percentage of the total phosphate load from cultivated agricultural land (>75%), erosion control remains the most effective method for reducing the total phosphorus load from this source. Conservation tillage, and especially no-till (the soil is not disturbed, except at planting time where a narrow slot is opened to place the seed, and herbicides are used

exclusively for weed control), can achieve significant erosion control, in many cases >90%, by protecting the soil surface. Logan (1981a) has shown that, because of the selective erosion and transport of clay-sized mineral particles and organic matter, conservation tillage is only about 90% as efficient in reducing the loss of total particulate phosphorus from land as it is in reducing erosion. However, this still provides readily available means of reducing a significant proportion of the total phosphate load.

Logan (1981a), in his review of the effects of conservation tillage on phosphate losses from agricultural land, found that, while no-till was quite efficient in reducing the total particulate phosphate load, no-till increased or had no effect on the dissolved P load. There are several reasons for this effect. In a non-equilibrium soil, such as a fertilized soil where the phosphate added has not reacted completely with the soil minerals, there are significant amounts of P held in soil pores, precipitated as soluble compounds or adsorbed on soil surfaces (Fig. 1). During runoff some of this P is removed and comes to equilibrium (or near equilibrium) with the sediment in the runoff. When no-till is used, the amount of sediment in the runoff decreases dramatically but the runoff volume is changed to a lesser extent (Logan 1981a). There-

fore, the amount of dissolved P, initially removed during runoff, is similar to that from conventional tillage; but there is less sediment in the no-till runoff to buffer the dissolved P load. It, therefore, increases relative to conventional tillage. Another reason for higher dissolved P levels, with no-till, is the buildup of fertilizer P on the surface of no-till soils (discussed in the next section) and the leaching of dissolved P from decaying plant residues. McDowell et al. (1980) examined the equilibrium P concentrations (EPC_0) of no-till soils in Mississippi with corn grown for grain and for silage. EPC_0 as shown in Fig. 2 is a measure of dissolved P concentrations in runoff, and McDowell et al. (1980) reported EPC_0 values of 0.18 and 0.22 μg P ml^{-1} for grain plots and 0.15 and 0.15 μg P ml^{-1} for the silage plots, where much of the corn stalks are removed at harvest. In addition, losses of dissolved P in the fall and winter period from spring-plowed corn grain plots were 0.426 kg ha^{-1} and 0.195 kg ha^{-1} for the silage plots.

These results indicate that, while no-till can be expected to greatly decrease soil loss on land previously tilled, the main effect on phosphorus loads will be to significantly decrease the particulate P. However, there will be no change, or there may be an increase in dissolved P. If this is true, then other management options for controlling dissolved P must be considered, including fertilizer use.

When fertilizer P is added to soil, it is generally 100% orthophosphate, (i.e. it is potentially all crop-available). However it reacts immediately with the small volume of soil, with which it is in contact, and much of it is rendered insoluble and unavailable. Long-term soil fertility research has shown that only about 10–20% of the fertilizer P remains available after field addition. The resulting unavailable forms include highly insoluble precipitates of P with soil-containing cations such as Fe, Al, Mg, Ca and others, and P which has sorbed and diffused into soil mineral surfaces. The available forms include dissolved P in soil pores and labile P sorbed onto soil surfaces. The chemistry and mineralogy of a particular soil will determine the extent to which fertilizer P is converted to available and unavailable forms, and the distribution of P between these forms will affect the amounts of particulate and dissolved P in runoff. But, for a particular soil, any increase in P fertilizer over and above the needs of the crop will increase the levels of available P in the

soil and the levels of dissolved P as well.

Since P fertilization after crop removal will increase the total P level in the surface soil, it should also increase TPP losses in runoff if erosion remains the same. More important, however, is the effect of P fertilization on the loss of 'available' TPP. In the studies reported here, 'available' means available to crops and is determined by a standard chemical extraction such as Bray P1 (Bray & Kurtz 1945). However, it also reflects the pool of TPP that would be available to algae. Lake & Morrison (1975) reported on a simulated rainfall study of fertilized and unfertilized Indiana soils. They showed that while TPP in runoff increased with fertilization, the increase in Bray P1 available sediment P losses was much greater and can be explained by the higher available P levels of the clay fraction of the soils, the fraction that is selectively transported during runoff. Smith et al. (1979) also showed that, where P was applied in excess of crop needs, Bray P1 sediment P losses increased, and the increase was greatest with no-till.

These relationships can be visualized in Fig. 4 which shows the relative enrichment of clay, total P and labile P in runoff sediment, with change in erosion level or sediment concentration.

Fertilizer P increases the available P level in soil and dissolved P in soil pores. Also, at higher available P levels the equilibrium, between adsorbed P and P in solution, is shifted towards higher dissolved P concentrations. If runoff volume and soil loss remain the same, P fertilization in excess of crop needs should increase dissolved P losses in runoff. Several researchers have clearly demon-
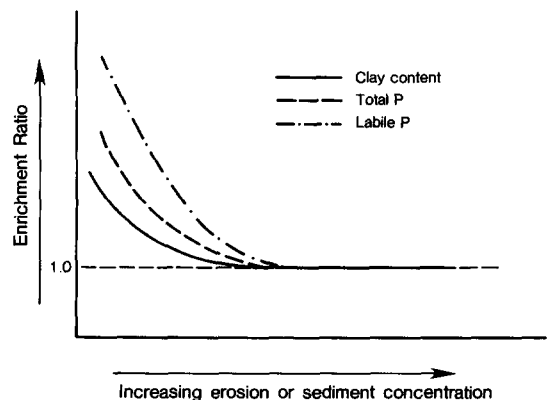

Fig. 4. A conceptual view of enrichment of clay, total P and labile P with sediment load in land runoff.

strated this relationship using different approaches. Romkens & Nelson (1974) found a highly significant positive linear relationship between dissolved P concentrations in runoff and Bray Pl extractable P of the sediment; and McDowell *et al.* (1980) found that several available P tests, including Bray Pl and Olsen's bicarbonate (Olsen *et al.* 1954), were correlated with EPC_0, a measure of dissolved P concentrations of water in equilibrium with sediment.

Oloya & Logan (1980) recently examined the desorption of dissolved P from soils and sediments with a wide range of Bray Pl extractable P levels and found that there was a highly significant positive relationship (Fig. 5); about 5–10% of the available P could be desorbed in a 24-h period.

No-till soils are commonly fertilized by broadcasting on the soil surface, or as a starter application with the planter if available P levels are high enough. Since the soil is not disturbed and since fertilizer P reacts rapidly with the small volume of soil it contacts, P broadcast on the surface of no-till soils will accumulate there. This is illustrated by the

data of Oloya & Logan (1980). A total of 268 kg P ha^{-1} was applied to no-till and conventional plots over a four-year period. This increased the Bray Pl available P from 18 μg P g^{-1}, prior to fertilization, to about 50 μg g^{-1} in the 15 cm depth of the fall-plowed plots, and to 150–184 μg g^{-1} in the 0–5 cm depth of the no-till plots. This increase resulted in increased desorbable P from these soils (Fig. 6) and the monitoring data (Logan 1981b) indicate that soluble P levels in the runoff have also increased steadily, with the greatest increases from the no-till plots.

Romkens *et al.* (1973) used simulated rainfall to study runoff from fertilized and unfertilized plots with different tillage systems. No-till and chisel plowing reduced soil loss, compared to conventional plow systems, but dissolved P losses were higher with the no-till system. The fertilized no-till plot had 4.85 kg P ha^{-1} dissolved P loss, compared to 0.003 kg ha^{-1} with the fertilized conventional plowing. This study has been criticized because the simulated rain was applied only 24 h after fertilizer application, but it nevertheless demonstrates the
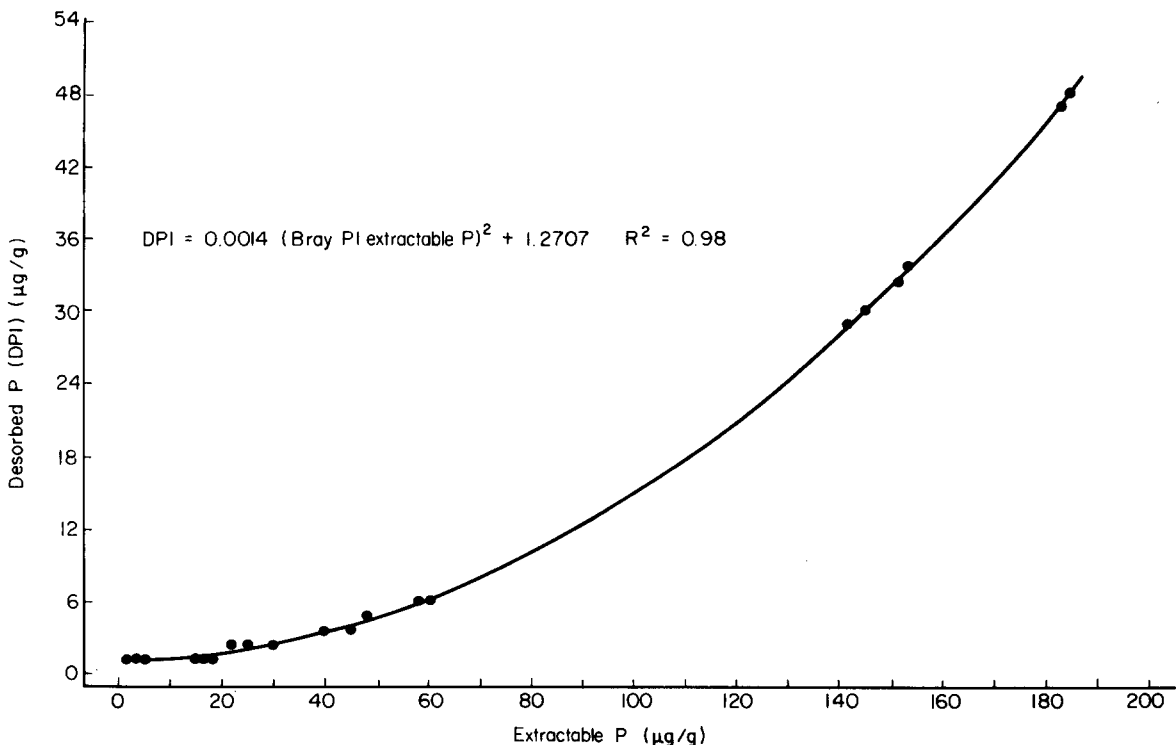

Fig. 5. Regression curve and equation for desorbed P (DPI) vs. Bray Pl extractable P for Hoytville soils from tillage experiment (Oloya & Logan 1980).
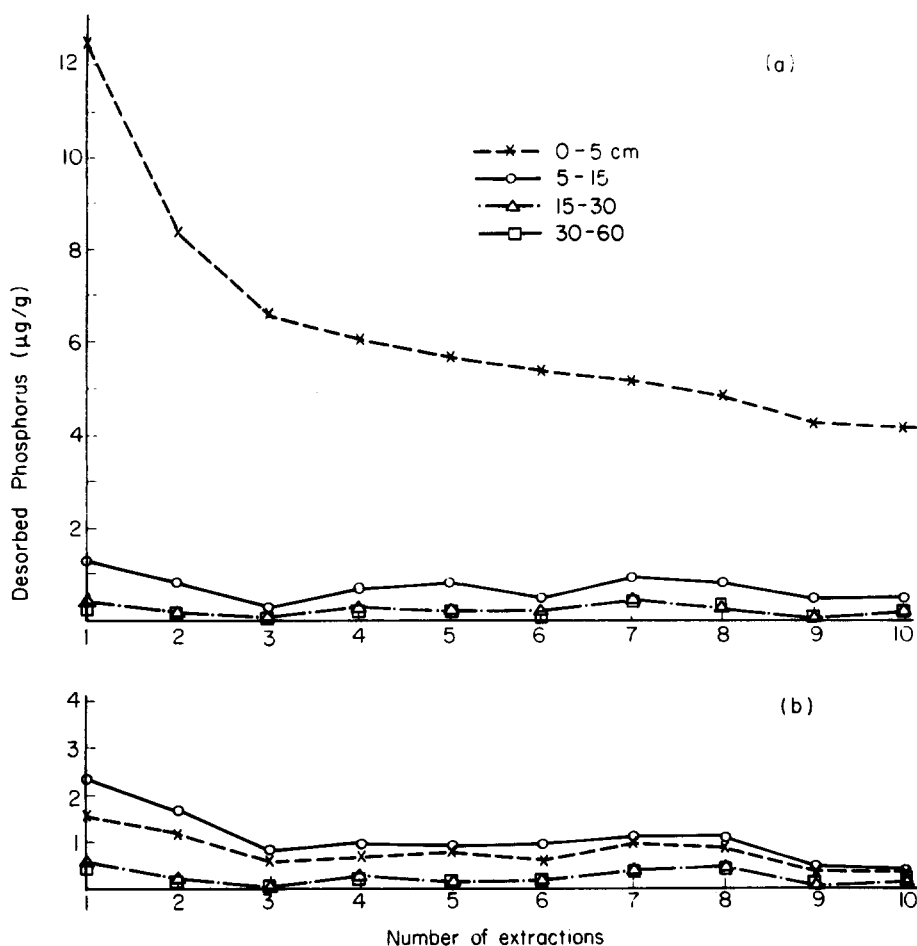
Fig. 6. Sequential P desorption for (a) no-till and (b) fall-plowed Hoytville soils sampled at depths of: 0–5, 5–15, 15–30 and 30–60 m (Oloya & Logan 1980).

potential for loss of dissolved P when fertilizer is broadcast on no-till soil.

Baker & Laflen (1980) also used simulated rainfall to study movement of fertilizer P with residue management. They found that placement of fertilizer P on the soil surface, either above or below the crop residue, gave much higher soluble P losses than nonfertilized plots, or plots where the fertilizer was injected below the soil surface. Their work showed that the major problem was the accumulation of fertilizer P at the surface and not necessarily the degree to which the fertilizer reacted with the soil prior to rainfall. This is probably because, as Oloya & Logan (1980) have shown, surface-applied fertilizer P builds up very high levels of available P at the soil surface and part of this is desorbed during runoff to become dissolved P.

Implications of variable particulate and dissolved phosphate transport to lakes

Present lake eutrophic models (Bierman 1980) do not adequately account for varying availability of P sources, and therefore, there is no way to determine an available P target load for lakes. However, it is obvious that response of algae to dissolved P must be different than to particulate P sources, which may settle out of the photic zone before they can be completely utilized. Phosphorus control strategies for deep lakes, with little sediment resuspension or lake overturn, must emphasize dissolved P sources more than for shallower lakes, where sediments are continuously resuspended and lake overturn can bring P released from bottom sediments to the surface. If the emphasis is to be placed equally on

sediment as well as dissolved P sources then erosion control, by conservation tillage, merits equal consideration with point source reductions. If, however, dissolved P loads are the most important, then control of point sources which discharge directly to the lake should receive the most attention. In addition, agricultural practices which control dissolved P losses will need to be emphasized. These include fertilizer management (Oloya & Logan 1980) and livestock waste management (Porter 1975). Future refinements of phosphate lake response models must account for varying particulate and dissolved P inputs, and their interactions in the lake.

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