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The Management of Soil Phosphorus Availability and its Impact on Surface Water Quality

A.N. Sharpley and A.D. Halvorson

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I. Introduction

An increased public perception of the role of agriculture in nonpoint source pollution, has prompted an urgency in obtaining information on the impact of current and proposed agricultural management practices on surface water quality. The transport of phosphorus (P) to surface waters can lead to accelerated eutrophication of these waters, which limits their use for fisheries, recreation, industry, or drinking. Although nitrogen (N) and carbon (C) are also associated with accelerated eutrophication, most attention has focused on P, because of the difficulty in controlling the exchange of N and C between the atmosphere and a water body, and fixation of atmospheric N by some blue green algae. Thus, P often limits eutrophication and its control is of prime importance in reducing the accelerated eutrophication of surface waters.

Inputs from point sources are easier to identify and have thus, seen more control than diffuse sources. Consequently, agricultural nonpoint sources now account for a larger share of all discharges than a decade ago (US EPA, 1984; Crowder and Young, 1988). In response to this, the Water Quality Act of 1987 (Section 319) increased attention on the need to control nonpoint sources of pollution to achieve the nation's water quality goals, with federal funds becoming available in 1990 for implementation of preventive and remedial control measures. In many states, the pollution control measures will focus on minimizing agricultural P losses through adoption of alternative or improved management practices to control soil erosion. This will require research on the long-term management of soil P availability in alternative or improved agricultural practices, in relation to soil productivity and water quality. In addition, it will be necessary to efficiently transfer to action agencies, existing information on the forms and amounts of P available to both crops and transport in runoff and predict short- and long-term management impacts on crop production and surface water quality.

Profitable crop production depends on many factors, including a sound P management program. Except for sunlight and water, soil fertility most frequently limits crop yields. Even with perfect weather and climatic conditions, a farmer that does everything right except meeting crop nutrient needs, will never reach maximum economic yield potential.

Water, N, and P are generally the dominant yield limiting factors for crops in the United States. Potassium, S, and micronutrients are usually not as limiting and like N and P, their needs can be assessed by soil testing (Halvorson, 1987; Halvorson et al., 1987a). However, soil P deficiency for cereal grains and other crops is common (Potash and Phosphate Institute, 1985, 1987a). Fertilizer P management varies with location and site specific conditions, such as initial soil test P level, soil type, soil pH, available application equipment, crop rotation, and tillage system. Soil testing is the best tool available to assess the need for P fertilization. Accurately assessing soil P availability status and the quantity of

P fertilizer required to alleviate P deficiency is necessary if maximum economic yields are to be obtained.

Fixation and immobilization of soil P in inorganic and organic forms unavailable for crop uptake, necessitates P amendments as fertilizer, animal manure, or crop residue material to achieve desired crop yield goals. Thus, P application has become an integral and essential part of crop production systems in order to provide adequate food and fiber for U.S. consumption and export demands. In addition, proper management of fertilizer P may reduce P enrichment of agricultural runoff via increased crop uptake and vegetative cover. On the other hand, a history of continual P applications via fertilizer and manure has increased soil test P levels in several states, to a point where the majority of tested soils were above sufficiency levels (Figure 1, T. Sims, Univ. Delaware, pers. commun.). A recent survey by Dr. T. Sims of soil test P levels in several mid Atlantic states indicates that the major portion of soils tested either high or excessive in soil test P (Table 1).

It must be remembered that many soils have low to medium soil test P levels which require annual applications of P to sustain profitable crop production (Potash and Phosphate Institute, 1987b). Overall, however, fertilizer P use in the U.S. has been declining since 1980 (Figure 1, from Berry and Hargett, 1989 and Potash and Phosphate Institute, 1987b). This trend reflects action agency efforts to reduce unnecessary applications and farmers' response to high soil test P levels, policy changes, regulating price support, P fertilizer cost, and production control measures.

Several environmental factors affect plant uptake of P from any source, soil or fertilizer (Munson and Murphy, 1986). These include temperature, soil compaction, soil moisture, soil aeration, soil pH, type and amount of clay content, P status of soil, and status of other nutrients in soil. When soil temperatures are low during early plant growth, P uptake is reduced. Soil compaction reduces pore space which reduces water and oxygen content which in turn reduces P uptake. Soil pH greatly affects plant available P, with P being fixed by Ca at high pH and by Fe and Al at low pH. Soils with high clay content tend to fix more P than low clay soils. Thus, more P needs to be added to raise the soil test level of clay soils than loam and sandy soils. The presence of ammonium-N (NH_4-N) enhances P uptake by creating an acid environment around the root when NH_4^+ ions are absorbed. High concentrations of NH_4-N in the soil with fertilizer P may interfere with and delay normal P fixation reactions, prolonging the availability of fertilizer P (Murphy, 1988). Thus, many factors can affect P availability to crops.

In spite of the recent decrease in fertilizer P use, there still exists a challenge to increase the utilization of both on- and off-farm sources of P, by identification and careful management of indigenous and amended forms of available soil P. The placement of P fertilizer or manure in the root zone will provide optimum available soil P levels for plant uptake during periods of maximum crop uptake and will maximize soil productivity and minimize potential P losses in runoff. However, climatic, edaphic, agronomic, and economic factors limit achievement

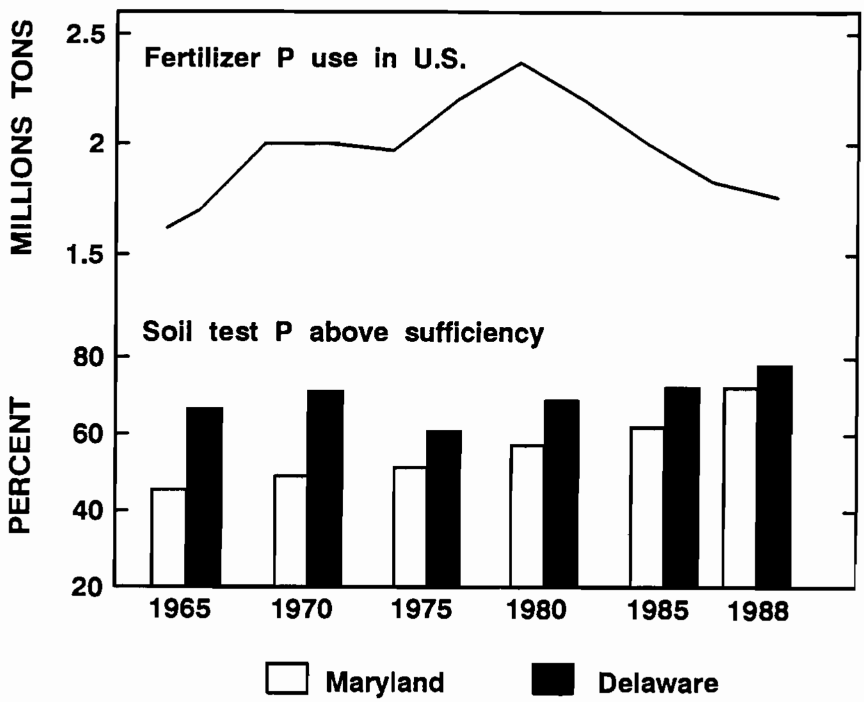

Figure 1. Fertilizer P use in the U.S. and percent soil samples testing above sufficiency level in Maryland and Delaware.

of this situation. In particular, the bulky nature and large amounts of animal manure that are produced throughout the year in localized areas, often limits subsurface placement of the manure. Careful consideration of these factors may lead to the development of more sustainable agricultural management systems that are environmentally sound.

This chapter presents the factors influencing P availability in soil and water systems, as shown in Figure 2. We discuss the role of agricultural management in maximizing soil P availability, while minimizing P losses in runoff. Considering this discussion, future challenges to the development of agronomically and environmentally sustainable P management systems and associated research needs are identified. With the current interest in the development and adoption of efficient and sustainable agricultural systems (Edwards et al., 1988; Francis et al., 1990; Potash and Phosphate Institute, 1989), there will be an increased reliance on efficiently utilizing indigenous soil P forms. Consequently, this chapter emphasizes the effect of soil management on the dynamics of organic and residual P availability.

Table 1. Phosphorus soil test survey for northeast, north central, and mid Atlantic states

State	Soil test method	Summary year	Soil test P level for				Percent samples testing [a]			
			Low	Medium	High	Excessive	Low	Medium	High	Excessive
			kg P ha^{-1}				%			
Northeast										
CT	Modified Morgan	1990	5.5	11	28	>40	25	17	29	29
ME	Modified Morgan	1990	4	11	45	>45	5	34	50	11
NH	Mod. Morgan (pH 4.8)	1990/91	<7	<17	<27	<27	51	21	10	18
NY	Mod. Morgan (ph 4.8)	1988	<4.5	9	45	>224	25	35	40	0
PA	Mehlich-3	1989/90	34	68	112	170	33	23	18	26
North Central										
IA	Bray-I	1983+	34	45	67		31	13	19	0
IN	Bray-I	1988/89	22	34	56	>56	11	11	21	50
KS	Bray-I	1990/91	28	56	112	>112	25	27	24	21
MI	Bray-I	1990/91	<34	67	>90	>224	10	27	40	23
MN	Bray-I	1968-76	11	17	45	>45	12	21	18	49
MO	Bray-I	1990	25	50	78	>150	19	30	23	25
NE	Bray-I	1990/91	5.5	17	27	>27	35	23	22	20
ND	Olsen	1990/91	21	32	44	>44	39	29	13	12
SD	Bray-I and Olsen	1990	<17	28	45		22	30	25	0
Mid Atlantic										
DE	Mehlich-1	1991	38	75	150	>150	9	21	28	40
MD	Mehlich-1	1990	29	50	100	>100	8	18	74	0
NJ	Mehlich-1	1989	30	50	99	>99	12	8	15	65
NC	Mehlich-3	1990/91	60	120	240	>240	6	9	23	62
SC	Mehlich-1	1988	22	44	88	>270	17	25	34	5

[a] In some states, percentages may not total 100 due to incomplete data.

From unpublished data of T. Sims, University of Delaware.

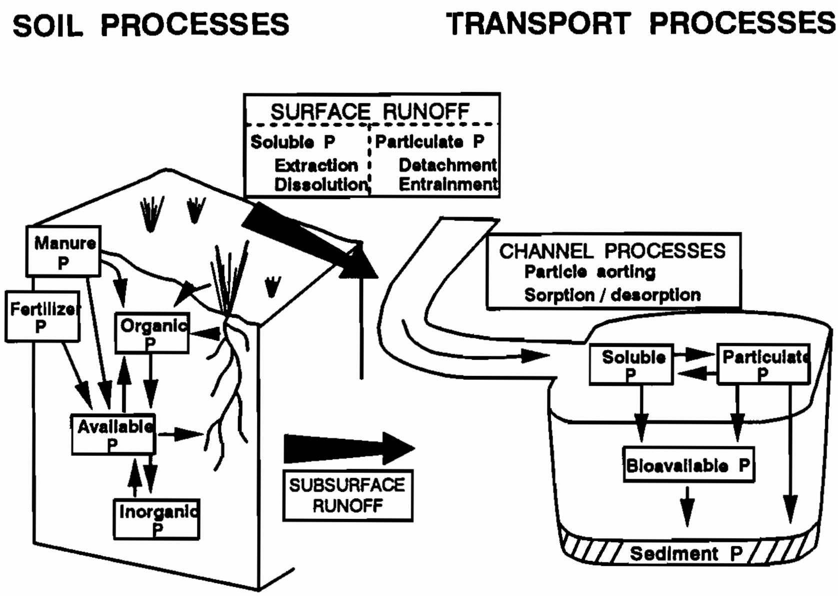

Figure 2. Factors influencing P availability in soils and water.

II. Dynamics of Soil P Availability

Several studies on the effect of agricultural management on the dynamics of P cycling in soils, have found a differential behavior of inorganic and organic P forms (Agboola and Oko, 1976; Harrison, 1978; Sharpley, 1985a; Tiessen et al., 1983). With the application of P, available soil P content increases (Barber, 1979; Khasawneh et al., 1988; McCollum, 1991; Peterson and Kreuger, 1980). This increase is a function of certain physical and chemical soil properties (Barrow, 1980; Larsen et al., 1965; Lopez-Hernandez and Burnham, 1974). The portion of fertilizer P remaining as available P (resin P) 6 months after application, decreased as clay, organic C, Fe, Al, and $CaCO_3$ content increased for over 200 widely differing soils (Table 2: from Sharpley, 1991; Sharpley et al., 1984a; 1989). With an increase in degree of soil weathering, represented by soil taxonomic and other related properties, a general decrease in availability of applied fertilizer P was evident. Clearly, the dynamics of fertilizer P availability differs between soils, and influences the degree of soluble P enrichment of surface runoff.

Where no fertilizer P is added, a net loss of P from the system via removal in the harvested crop is often accounted for by a decrease in soil organic P, while inorganic P generally remains constant. For example, the growth of cotton on a Mississippi Delta soil, Dundee silt loam for 60 yr (1913-1973), with no reported fertilizer P applied, had little affect on inorganic P content (Sharpley

Table 2. Percent fertilizer P available (as resin P) 6 months after application

Related properties	Number of soils	Availability Mean	Availability Range
		%	
Calcareous			
$CaCO_3$	56	45	11-72
Slightly weathered			
Base saturation	80	47	7-74
Available P			
pH			
Moderately weathered			
Clay	27	32	6-51
Available P			
Organic C			
Highly weathered			
Clay	40	27	14-54
Extractable Al			
Extractable Fe			

Data adapted from Sharpley, 1991 and Sharpley et al., 1984a, 1989.

and Smith, 1983). However, a decrease in the organic P content of the cultivated (93 mg kg^{-1}) compared to virgin analogue (223 mg kg^{-1}) surface soil (0-15 cm) was measured. Apparently, mineralization of organic P replenished the inorganic pool and provided adequate amounts of plant available P.

A. Organic P Mineralization

Though inorganic P has generally been considered the major source of plant available P in soils, the incorporation of fertilizer P into soil organic P (McLaughlin et al., 1988) and lack of crop response to fertilizer P due to organic P mineralization (Doerge and Gardner, 1978), emphasize the need to consider organic P in the management of soil P availability, particularly with reduced tillage practices. For more detailed information on the dynamics of soil inorganic P transformations and availability as affected by management, the reader is referred to articles by Khasawneh et al. (1988) and Syers and Curtin (1988). The main variables controlling the dynamics of organic P mineralization

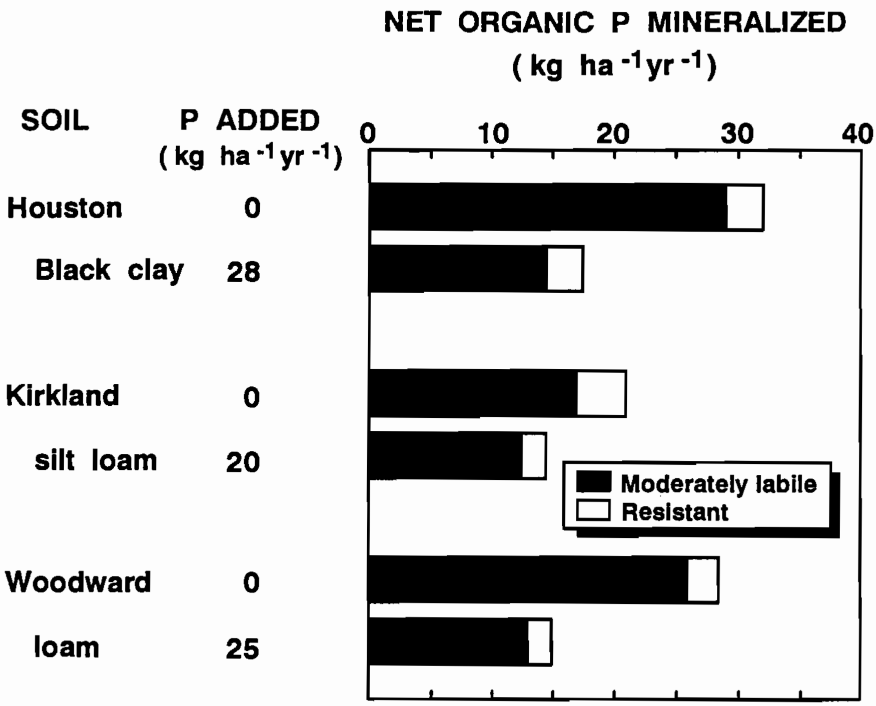

Figure 3. Annual net mineralization of moderately labile and resistant organic P in unfertilized and fertilized soils. (Data adapted from Sharpley, 1985a.)

can be divided into those related to climatic and soil factors and to crop residue factors.

1. Climatic and Soil Factors

Organic P mineralization in several unfertilized and P fertilized soils in the Southern Plains was quantified by Sharpley (1985a) as the decrease in soil organic P content during the period of maximum crop growth (spring and early summer). However, as soil organic P may be formed by plant residue incorporation, the value of net organic P mineralization will underestimate the actual value. Organic P was fractionated into labile, moderately labile, moderately resistant, and resistant pools by the sequential extraction procedure of Bowman and Cole (1978).

Averaged for each soil type, organic P mineralization ranged from 15 to 33 kg P ha^{-1}, with mineralization greater in unfertilized than P fertilized soils (Figure 3). Of this, moderately labile organic P contributed 83 to 93% of that mineralized. As labile and resistant organic P pools remained fairly constant

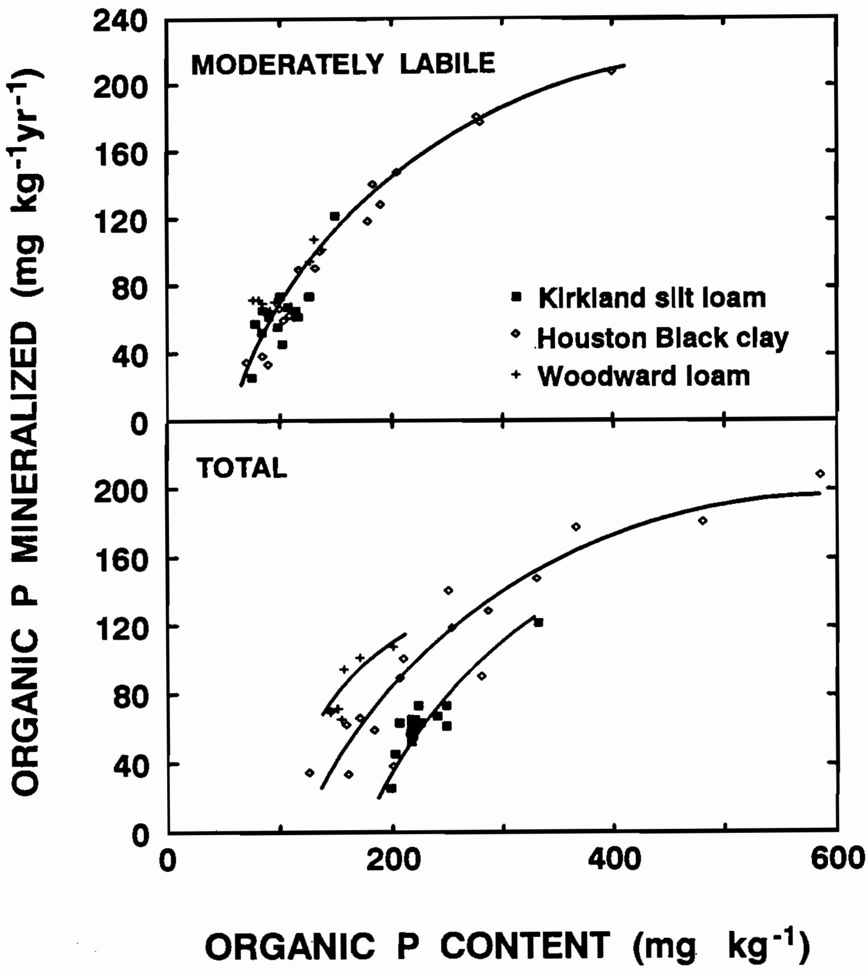

Figure 4. Annual net mineralization of organic P as a function of total and moderately labile organic P for three soils.

during the 2-yr study (Sharpley, 1985a), mineralization of moderately labile organic P replenished the available P pool, when it fell below a critical but as yet undefined level. Tate et al. (1991) also found labile organic P mineralization was an important source of P to pasture in both low- and high-P fertility soils in New Zealand. Both studies (Sharpley, 1985a; Tate et al., 1991), suggest that management practices maximizing the build-up of organic matter during autumn and winter, may reduce external P requirements for plant growth during the following spring and early summer.

Net organic P mineralization was related to total organic P content for both unfertilized and fertilized soil (Figure 4, from Sharpley, 1985a). For a given

Table 3. Average net amount of organic P mineralized for several climatic regions

Region	Fertilizer P applied	Organic P mineralized	Percent mineralized [a]
	------------kg P ha^{-1} yr^{-1}-----------		% yr^{-1}
Southern Plains	0	23	11
	25	17	8
Temperate	0	11	2
	34	6	1
Tropics	0	157	15
	40	67	18

[a] Percent of total soil organic P which is mineralized annually.
data adapted from Sharpley (1985a) and Stewart and Sharpley (1987).

organic P content, mineralization was greater for Woodward than Houston Black and Kirkland soils. As a function of moderately labile organic P, however, no difference between locations was observed (Figure 4). Apparently, organic P mineralization dynamics were a function of moderately labile organic P, the level of which is determined by climatic and soil factors. Further, organic P mineralization (15 to 33 kg ha^{-1} yr^{-1}) was not completely inhibited by fertilizer P application (20 to 28 kg ha^{-1} yr^{-1}), with similar amounts of P contributed by both sources (Figure 3). Amounts of organic P mineralized in the 3 Southern Plains soils studied by Sharpley (1985a) are similar to other temperate soils (Table 3). They are generally lower, however, than for soils from the tropics (67 to 157 kg ha^{-1} yr^{-1}), where distinct wet and dry seasons and higher soil temperatures can increase the amounts of organic P mineralized.

2. Crop Residue Factors

Residue management can affect P cycling and availability as a function of residue amount, type, and degree of incorporation with tillage. A greater amount of residue will increase the amount of P being cycled and, particularly if left on the surface of the soil, will reduce evaporation losses and keep surface soil moist for more days during the growing season, thereby enhancing microbial activity and mineralization (Figure 5, from Sharpley and Smith, 1989a). Little difference in net organic P mineralization and mobility was observed between residue types during the 84-day incubation (Figure 5), even though residue C:P ratio ranged

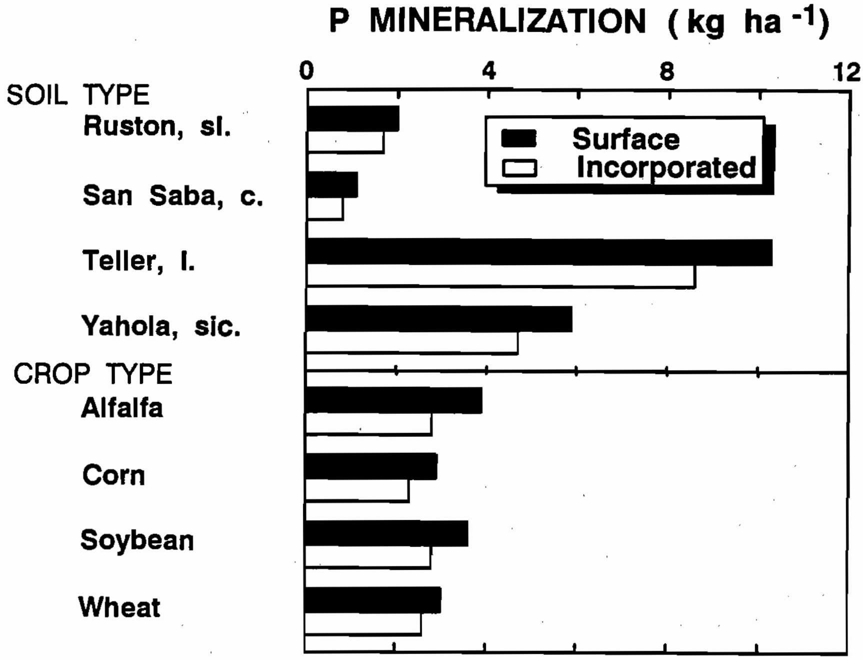

Figure 5. Mineralization of crop residue P in 84 days as a function of residue placement and soil and crop type.

from 260:1 (alfalfa) to 600:1 (corn). This suggests that the effect of C:P ratio may be more evident over a longer period of time as shown for crop residue N mineralization by Power et al. (1986). Net organic P mineralization and mobility was affected, however, by soil type (Figure 5), as a function of available P content and soil C:P ratio. This is consistent with the wide C:P ratio of the residues and resultant immobilization of soil P, prior to mineralization. Although amounts of residual P mineralized and leached are small (Figure 5) compared to normal fertilizer P applications to these soils and crops (30 kgP ha^{-1} yr^{-1}), they may provide an important source of available P during initial crop growth when the residue will be either fresh (i.e., fall planting of winter wheat) or coming out of a relatively inactive microbial period (i.e. spring planting).

Continuously cultivated soils generally decline in organic matter content, such that the major source of P removed by a crop will be supplied by fertilizer. Under reduced tillage, enhanced microbial, faunal, and phosphatase activity may increase immobilization of fertilizer P as microbial organic P and subsequently, rates of organic P mineralization. These processes will also be important under crop rotations, which include a leguminous cover crop such as alfalfa, summer-fallow, or pasture phases, where fertilizer P may be rapidly incorporated into soil organic P, providing a potential source of available P for subsequent crops.

Further research is needed on the dynamics and magnitude of fertilizer P immobilization and mineralization as a function of climate (soil temperature and moisture), soil, and crop management. In addition, preliminary research (Sharpley and Smith, 1989a; Tate et al., 1991; Thibaud et al., 1988), indicates further information is needed on the relative decomposition and mineralization of residual P as a function of soil and crop type under different management practices.

B. Analytical Limitations

One of the main limitations in evaluating the contribution of organic P mineralization to the dynamics of soil P cycling and soil availability under different management practices, has been the difficulty in accurately quantifying and identifying soil organic P. Total organic P is determined indirectly by difference (total - inorganic P) following either extraction or ignition methods (Olsen and Sommers, 1982). However, solubilization or complexation of mineral P during ignition and hydrolysis and/or incomplete removal of adsorbed or occluded organic P during extraction, may introduce errors in organic P estimation between and within soil types. Recent modifications of extraction and ignition methods (Bowman, 1989; Soltanpour et al., 1987) may simplify and increase the accuracy of organic P quantification. Although complete characterization of soil organic P has not been accomplished, application of solid state neutron magnetic resonance (NMR) spectroscopy may improve its identification. Alternatively, sequential extraction procedures may fractionate organic P according to its chemical stability and these fractions then related to bioavailability (Bowman and Cole, 1978; Hedley et al., 1982; Potter et al., 1991).

With the development of fumigation-extraction techniques to measure soil microbial biomass P (Brookes et al., 1982; Hedley and Stewart, 1982), its importance in P cycling has been recognized (McLaughlin et al., 1988; Stewart and Tiessen, 1987). For example, McLaughlin and Alston (1986) showed that microbial biomass can assimilate a similar portion of fertilizer P as taken up by wheat. In a study of P cycling through soil microbial biomass in England, Brookes et al. (1984) measured annual P fluxes of 5 and 23 kg P ha^{-1} yr^{-1} in soils under continuous wheat and permanent grass, respectively. Although biomass P flux under continuous wheat was less than P uptake by the crop (20 kg P ha^{-1} yr^{-1}), annual P flux in the grassland soils were much greater than P uptake by the grass (12 kg P ha^{-1} yr^{-1}).

Not all this biomass P is available for plant uptake each year; some will be directly transferred to subsequent microbes and some released to the soil solution, which can then be fixed by soil or taken up by microbes or plants (Brookes et al., 1984). However, the rate and extent of microbial P flux emphasizes the importance of microbial P in controlling the short-term dynamics of organic P transformations and thereby, management of soil P availability.

Because of the inaccuracies in organic P measurement, which are compounded in estimating organic P mineralization, attention has been given to direct measurement of native and residual organic P mineralization by isotopic dilution (Walbridge and Vitousek, 1987) or addition of ^{32}P labelled organic matter, respectively (Dalal, 1979; Harrison, 1982; McLaughlin et al., 1988). Further research is needed, however, on the validity and relative importance of assumptions concerning achievement of isotopic equilibrium and bidirectional and periodic movement of P between available and microbial pools in estimating organic P mineralization. Although the length of isotopic studies is limited due to the short half-life of ^{32}P and ^{33}P (14.3 and 24.4 days, respectively), they have indicated that the actual rate of organic P mineralization has been underestimated by nonisotopic difference methods. Improvement and application of these methods, will aid future research to elucidate the dynamics of organic P mineralization and role of microbial biomass as a function of soil, residue, and agricultural management.

III. Management of Soil Availability

A. Fertilizer Management

Fertilizer P management strategies that maximize soil P availability while minimizing surface soil accumulations which may increase P loss in surface runoff, must consider fertilizer application rate, timing, type, placement, and residual availability. Fertilizer rate is primarily determined by soil test P levels and desired crop yield goals, discussed in the next section. Because of the immobility of P in most soils, the timing of fertilizer P application is not as critical as its placement. In efforts to efficiently utilize P inputs in sustainable management systems, there has been renewed interest in the estimation and utilization of residual P availability from fertilizer or manure amendments (McCollum, 1991; Pierzynski et al., 1990; Yerokum and Christenson, 1990).

1. Type

Except for the increased acidulation of P fertilizers, producing triple from single super P and partially acidulated rock P (RP), there has been less development of P sources than N. The treatment of fertilizer P to increase its solubility and thereby, crop-use efficiency, increases its cost. Consequently, farm management decisions regarding the type of fertilizer P to be used, have been based more on economic and agronomic considerations than on chemical availability and environmental aspects.

Research has evaluated ways to broaden the use of slow release P fertilizers in the eastern U.S., such as RP, beyond soils which have low pH, Ca, and P content. For example, in soils of neutral pH, it may be possible to apply heavy initial dressing of finely ground RP and include a rotation of fine rooted legumes to generate a low pH rhizosphere with low Ca concentrations and thus increase RP dissolution. Other methods designed to increase acidity in the immediate RP-soil environment, and thereby its dissolution, include addition of elemental S (Muchovej et al., 1989), NH_4^+ fertilizers, or organic matter such as animal manure and crop residues (Hedley et al., 1989). More research is needed, however, to evaluate the effectiveness of these amendments to enhance P availability by root extraction in different soils and cropping practices. An increasing adoption of more efficient management systems and inclusion of forage legumes in crop rotations, along with the development of reactive RP sources, may increase the agronomic and economic value of RP. Thompson (1990) suggests that North Carolina reactive RP may have value for direct application, especially on moderately acid, medium to high organic matter soils low in available P and under forage legumes in low-input systems. Thompson (1990) also suggested that RP may be the best P amendment available for organic farmers and others using only naturally occurring soil amendments.

Less information is available on the effect of fertilizer type on the loss of P in runoff. For example, Sharpley et al. (1978) observed a slightly greater soluble P (SP) loss in runoff following the application of monocalcium P (MCP - main P component of super phosphate) to a permanent pasture in New Zealand (2.80 kg ha^{-1}), compared to that with dicalcium P (DCP) (2.17 kg ha^{-1}), a slow-release fertilizer. This difference was attributed to more rapid dissolution of MCP than DCP at the soil surface. However, an appreciably greater loss of sediment-bound P with DCP (4.92 kg ha^{-1}) than MCP (2.63 kg ha^{-1}), resulted from an increased loss of P by transport of the less soluble DCP particles in runoff. It is expected that RP will affect the enrichment of P in runoff in a similar manner as DCP.

2. Placement

Due to the general immobility of P in the soil profile, fertilizer placement is generally more critical for P than N. Not everyone agrees on the best method of P application. Fixen and Leikam (1989) stated that "contradictory recommendations for method and placement of phosphorus (P) often are due to the fact that conditions influencing P fertilizer response vary among studies." They discussed many factors affecting the effectiveness of P placement methods and addressed the following questions: a) Which is better, band or broadcast P applications?; b) Are all band P applications methods equal in effectiveness?; and c) How much can P recommendations be reduced if P is banded instead of broadcast? They list the following soil and crop factors as influencing fertilizer P response: a) soil test levels; b) P concentration of the fertilized soil solution; and c) root contact with the fertilized soil. Root contact with the fertilized soil is influenced by total root length, volume of soil fertilized (varies with placement method), and location of the fertilized soil in relation to plant roots.

Depending on soil and environmental factors, band applications of P may or may not be better than broadcast incorporated applications of P. In general, if there is a difference in crop response due to P application method, yield response to band applications will be equal to, or better than, broadcast applications. Long-term studies in the northern Great Plains have shown that high rates of broadcast P (90 kg P ha^{-1}) can have long-term effects (17 years) on soil test P, wheat yields, (Bailey et al., 1977; Halvorson and Black, 1985b; Roberts and Stewart, 1987) and profitability (Jose, 1981; Halvorson et al., 1986). Several studies have shown a greater yield response to surface or subsurface band application of fertilizer P at low rates, compared to broadcast or mixing (Alston, 1980; Bailey and Grant, 1989; Lammond, 1987; Yost et al., 1981). In fact, Welch et al. (1966) observed greater P uptake and yield of corn with a combined banded (50%) and broadcast (50%) application (40 kg ha^{-1}).

In addition to agronomics, other factors are equally important in selecting the best P application method. Equipment availability, labor requirements, product availability, and availability of operating capital all affect this decision.

Deep placement of N with P (pre-plant banding) has grown in popularity recently in the Great Plains of the U.S. and in the Canadian Prairie Provinces (Murphy, 1988). The placement of N with P under both conventional and reduced tillage systems has frequently been more effective for wheat than application methods which placed most of the N and P at different positions in the soil (Dahnke et al., 1986; Harapiak and Flore, 1984; Leikam et al., 1983), particularly at low P soil test levels. Generally, yield differences between deep banding or P placement near the seed have been relatively small. Consequently, fertilizer recommendations frequently do not differentiate between seed placement and deep banding in terms of P efficiency. Alessi and Power (1980) reported wheat yield increases resulting from banding P with the seed even when soil test levels were high. Environmental conditions should be considered in addition to soil test results as a part of the management decisions which go into recommending fertilizer rates for higher wheat yields. Cold, wet soil conditions compounded by heavy surface residue may be conducive to P responses, particularly from starter applications, even when P soil tests are high (Murphy, 1988).

a. Soil Test P

Method and rate of application can affect the response of wheat to P fertilization. If low rates of fertilizer P are applied to soils testing "low" in plant-available P, then banding the fertilizer P below or with the seed is generally more efficient and results in greater yield increases than broadcast P applications (Murphy and Dibb, 1986; Peterson et al., 1981; Sleight et al., 1984). However, if sufficient fertilizer P was to be added to attain maximum wheat yields on a soil testing "low" in P, then method of placement may not be as critical. On soil testing medium to high in available P, the difference in effectiveness between broadcast and band applications of any type is lessened (Peterson et al., 1981). The work of Wagar et al. (1986) supports this theory. They found that a single, broadcast P application of 80 kg P ha^{-1} had a greater cumulative yield after 5 years than 20 kg P ha^{-1} applied each crop year with the seed. Thus, the broadcast treatment produced at or near optimum yields each year, whereas the seed placed P treatment produced at less than optimum yield potential during the first several years. They also found that a combination of a residual 40 kg P ha^{-1} broadcast one time plus 10 kg P ha^{-1} applied each crop year with the seed produced near maximum wheat yields. The latter treatment would be desirable from the standpoint of spreading the P fertilizer costs out over a longer time frame and still being able to maintain near maximum yield potential. However, the recommendation of one-time, high P application rates at a particular site, must consider the potential vulnerability for P loss in runoff from the site. Site variables that should be considered include runoff and erosion potential and proximity to P-limited surface waters. The role of these factors in determining vulnerability to P loss is discussed in more detail in Section VI-D.

b. Soil Type

The effect of P application, however, varies with soil type. When P fertilizer is placed in a specific soil volume, root extraction of P depends on the rate of application, which affects soil P adsorption/desorption characteristics and diffusion, and on the stimulation of root growth in the fertilized soil volume. For six soils having a hundred-fold variation in P sorptivity, Holford (1989) found that fertilizer P effectiveness, as measured by yield response in the first crop (wheat), residual effect in the second crop (clover), or cumulative recovery of applied P, was consistently greater for shallow banding at 5 cm depth compared to banding at 15 cm and broadcast applications. The almost equal effect obtained by mixing P throughout the soil, regardless of P sorptivity, suggested that the important factor in maximizing fertilizer effectiveness is its positional availability in the root zone rather than reduction of chemical immobilization by concentration in bands (Holford, 1989).

c. Crop Factors

Positional availability will also be influenced by crop type. For banding or restricted fertilizer placement to increase potential root extraction of P, the rate of P absorption and growth of roots in fertilized soil must increase to compensate for roots in unfertilized soil. Increased root growth and P uptake in the P-fertilized volume of soil compared to unfertilized soil has been observed for corn (Anghioni and Barber, 1980), soybean (Borkert and Barber, 1985), and wheat

(Yao and Barber, 1986). In contrast, several studies have shown that flax does not respond to banded fertilizer due to an inability of its root system to expand and proliferate into and efficiently absorb P from high concentrations in the fertilized zone (Soper and Kalra, 1969; Strong and Soper, 1974). In the case of flax, increased P uptake and yield response was obtained when fertilizer P was placed 2 to 5 cm directly below the seed, ensuring adequate P levels during early growth (Bailey and Grant, 1989).

In the final analysis, P placement may enhance it's availability and increase yields and must be considered in formulating a management plan to maximize both crop yields and associated water quality. Growers attempting to improve crop yields and profitability should maintain recommended rates of P even if more efficient methods of application are used. Cutting back on P rates to cut production costs may result in lost profits (Murphy, 1988).

3. Residual Availability

A need for higher P application rates to optimize crop yield potentials, necessitates that the short- and long-term economic and environmental impacts of P fertilizer management be evaluated. Most research on soil P fertility in the Great Plains has been limited to evaluating wheat response to P fertilizer application from one crop harvest (Dahnke et al., 1986; Fiedler et al., 1987; Follett et al., 1987; Leikam et al., 1983; Peterson et al., 1981; Westfall et al., 1986). Effects of residual fertilizer P in the northern Great Plains have been positive in increasing small grain yields (Bailey et al., 1977; Black, 1982; Halvorson and Black, 1985a; Read et al., 1977; Roberts and Stewart, 1987; Wagar et al., 1986) as well as increasing farm profit potential (Halvorson et al., 1986; Jose, 1981; Roberts and Stewart, 1987). Many of these studies were conducted with conventional dryland tillage systems and a crop-fallow cropping sequence. On a long-term (4 crop years) basis, a single broadcast application of P fertilizer (80 kg P ha^{-1}) may be equally as effective in increasing wheat yields as annual band applications (20 kg P ha^{-1}) (Roberts and Stewart, 1987; Sleight et al., 1984). Long-term P studies conducted by Alessi and Power (1980), Bailey et al. (1977), Black (1982), Halvorson and Black (1985a and 1985b), and Read et al. (1977) in the northern Great Plains indicate that benefits from a single P fertilizer application at rates of 45 kg P ha^{-1} or more may last as long as 16 years, depending on initial rate of application and cropping history. Halvorson (1989) reported that irrigated no-till winter wheat, grown annually on the same land, responded positively to residual broadcast fertilizer P (34 and 67 kg ha^{-1}).

Multiple year responses of alfalfa and grain sorghum to single applications of P fertilizer have been investigated in the central Great Plains (Havlin et al., 1984; Janssen et al., 1985, Schlegel et al., 1986). Halvorson and Black (1985a) suggested that a one-time, high-rate ($>$50 kg P ha^{-1}) application of P fertilizer may be one way to satisfy the P needs of crops grown with reduced and no-till

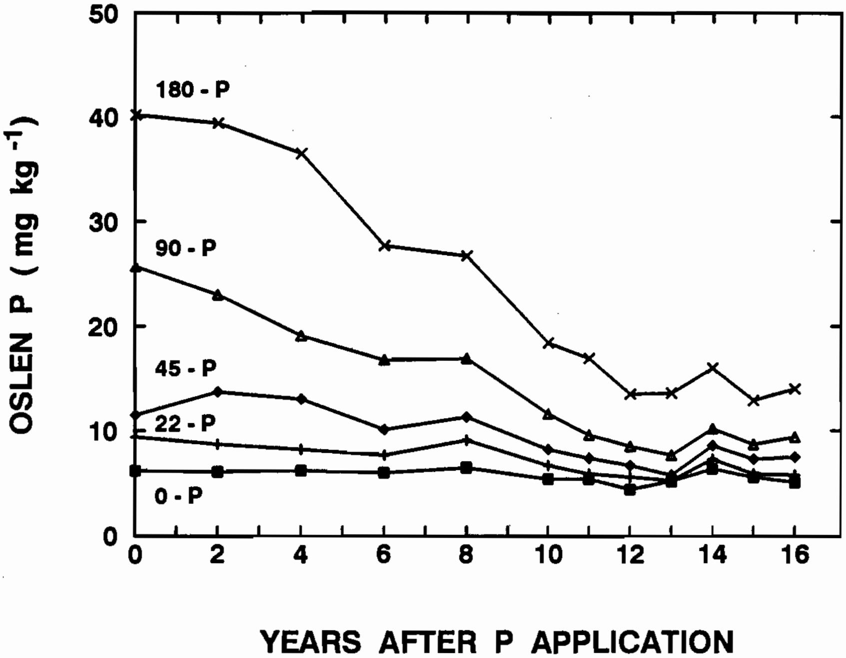

Figure 6. Changes in soil test P (Olsen P) levels with time after P fertilizer application (kg ha^{-1}) to a Williams loam in Montana. (Data adapted from Halvorson and Black, 1985a.)

systems for several years. Data from A.D. Halvorson and J.L. Havlin (1991 unpublished data) in Colorado supports this suggestion.

Phosphorus fertilization changed soil test P levels of several central Great Plains soils for several years (Hooker et al., 1980). Halvorson and Black (1985a) showed that soil test P levels were increased above the initial soil test P level for more than 16 years, by the one-time P applications on a Williams loam in Montana (Figure 6). After the initial increase, soil test P levels declined for about 12 years and then stabilized at a higher soil test level than was initially present, thus establishing what appears to be a new equilibrium level of soil available P. Fixen (1986) reported similar changes in soil test P levels with time. Crop yields reported by Halvorson and Black (1985a) were also improved by the residual P fertilizer for 16 years (Figure 7). Based on soil test P levels for the highest P rates, grain yields would have been increased for several more cropping seasons had the study been continued.

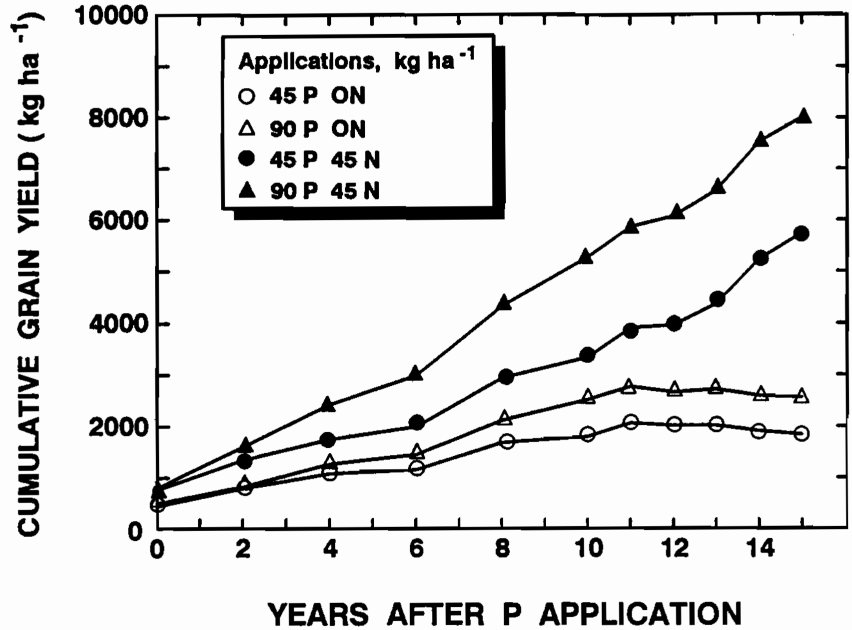

Figure 7. Cumulative wheat grain yield with years after initial P application with or without N applied each crop year. (Data adapted from Halvorson and Black, 1985a.)

B. Manure Management

Manure and organic wastes can be a valuable source of P to crops, improve soil physical properties, and increase soil organic matter content. In fact, by improving vegetative ground cover, manure application can reduce runoff volume and erosion. The concentration of P in manure is highly variable, thereby introducing uncertainty into meeting crop needs. However, the fertilizer value of manures is generally inversely related to their water and carbon contents.

Animal production operations have concentrated in localized areas for economic reasons, which include the close proximity of feed supply and meat processing plants. As a result, manure production often exceeds crop P requirements of both the producing and adjacent farms. Thus, disposal of the concentrated animal waste, that accumulates in confined production systems, is an increasing problem facing the industry. A major part of this problem has arisen from the fact that manure application rates have been based on N management. In most cases, this will lead to an increase in soil P, often well in excess of levels required for maximum yields. The importance to efficient management systems of basing manure application rates on soil P rather than N, are discussed in more detail in Section VII. B.1.

The continual land application of cattle (Sharpley et al., 1984b; Vitosh et al., 1973), poultry (Field et al., 1985; Sims, 1992), and swine (King et al., 1990; Sharpley et al., 1991b) manure, has resulted in an accumulation of P in surface soil. From a survey of several farms in Sussex County, Delaware, one of the most concentrated poultry production areas in the U.S., Sims (1992) found soil test-P levels (Bray I) ranging from 123 to 369 mg kg^{-1}, which exceeded the "high" criteria (50 mg kg^{-1}) established by Delaware soil test laboratories. In addition, a decrease in P adsorption capacity of soil following manure addition (Reddy et al., 1980; Sharpley et al., 1991b), increases the potential for P movement in lateral and vertical soil water flow (Brown et al., 1989; Magette, 1988; McLeod and Hegg, 1984; Westerman et al., 1983).

The increase in soil P availability is related to the rate of manure application. For four loam soils receiving long-term (8 to 35 years) poultry or swine manure, available P content of the surface 50 cm of soil increased an average 27 kg P ha^{-1} for every 100 kg P ha^{-1} added in manure (Sharpley et al., 1991b). In a laboratory incubation study, Field et al. (1985) observed a 12 to 23 kg P ha^{-1} increase in available P (double-acid extractable) for every 100 kg P ha^{-1} added in poultry manure. These values are similar to the proportional increase in soil test P (13 to 28%) following mineral fertilizer P application (Barber, 1979; McCollum, 1991; Rehm et al., 1984).

Organic wastes that are important in localized regions include sewage sludge from municipalities, waste from livestock slaughtering facilities, and wastes from the food processing and other industries. As for fertilizer and animal manure management, the composition, rate, placement, and time of application are major factors affecting soil P availability.

C. Crop Yield Potential

The relationship between the sodium bicarbonate P test and relative yield potential of wheat grown in a dryland wheat-fallow system is shown in Figure 8 (Halvorson, 1986a). These data indicate that a 26 mg P kg^{-1} level in the surface 15 cm of soil is needed to achieve 100% of the wheat yield potential in this semiarid environment. The relationship shown in Figure 8 is useful in estimating potential yield reductions caused by inadequate available P levels. When P fertilizer is added to most soils in the Great Plains, an increase in soil test P levels can be expected. The amount of increase will depend upon soil texture and other soil characteristics. Halvorson and Kresge (1982) used this approach to estimate the amount of broadcast-incorporated P fertilizer needed to optimize yields. If less P fertilizer is applied than recommended, wheat yield potentials are reduced along with N fertilizer needs. Halvorson and Kresge (1982) estimated that 4 to 5 kg P ha^{-1} was needed to raise the soil test P level 1 mg kg^{-1}. In Illinois, application of 9 kg P ha^{-1} is estimated to increase the soil

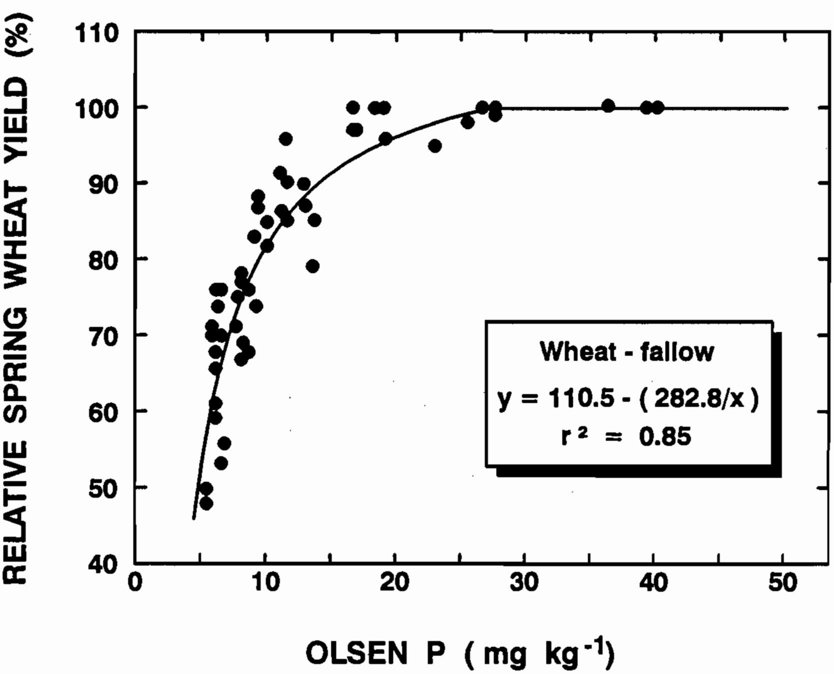

Figure 8. Relative spring wheat yield as a function of soil test P on a Williams loam in Montana. (Data adapted from Halvorson, 1986a.)

test P level 1 mg kg^{-1} (Agronomy Staff, 1989). Based on initial soil test P level and P application rate, Halvorson et al. (1987b) developed a model to predict the change in soil test P level with time on a Williams loam after an application of P fertilizer in a wheat-fallow system. This approach could be used, along with P removal rate by the crop, to predict when future additions of P will be needed. Similar data are needed for other soil types and cropping conditions.

Application of fertilizer P to bring the soil test P level to about 21 mg P kg^{-1} (Olsen P) or about 30 mg P kg^{-1} (Bray-I P) followed by smaller P applications to maintain this soil test level, may result in optimum wheat yields and optimum short- and long-term profitability. This approach to P fertilization would probably provide the potential for optimum wheat yields each crop year. In dry years, a high level of soil P (>20 mg kg^{-1}) will enhance yield potential and in the wet years, a high level of soil P will provide the opportunity to more efficiently utilize available water supplies, provided N is not limiting. For example, Black (1982) showed that in dry years spring wheat yields were increased an average of 417 kg ha^{-1} while in wet years yields were increased by 712 kg ha^{-1} with 180 kg P ha^{-1}.

In summary, P should not be a yield limiting factor for crop production. Phosphorus is a relatively immobile nutrient, not readily subject to leaching

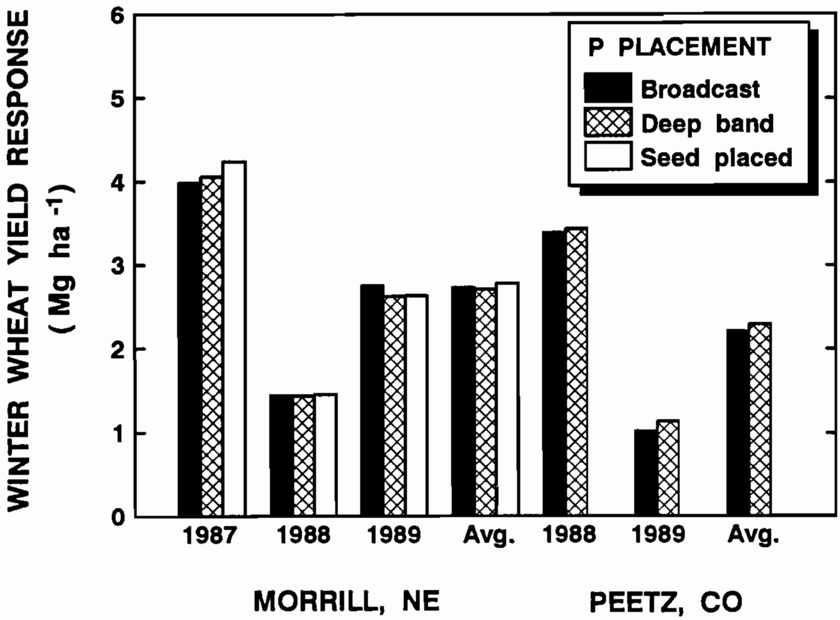

Figure 9. Winter wheat yield response to residual fertilizer P as a function of P application method on loam soils at two locations.

losses. The loss mechanisms are mainly through soil erosion and that removed in the harvested portion of the crop. Phosphorus fertilization is an investment that will pay dividends for several years and should be considered a capital improvement to the land. Therefore, a program to build soil P to a level adequate for maximum crop yield potential and maintain it at this level will probably be the most profitable in the long-term and environmentally sound. Establishing a soil P level adequate to eliminate P as a deficient crop nutrient can be accomplished by one of two methods: 1) by applying a one-time application of P, either broadcast or band, that is sufficient to raise the soil test P level to an optimum level; or 2) by applying smaller rates of P, either broadcast or band, for several crop years.

D. Nitrogen Required for Efficient Residual P Use

Adequate levels of N are essential to get full benefit from residual P fertilizer and efficient P utilization, regardless of the method of P application. Halvorson and Havlin (unpublished data) found that the addition of 45 kg N ha^{-1} increased winter wheat response to residual fertilizer P at Morrill, Nebraska. They also found that initial P placement method had no effect on winter wheat response to residual P (Figure 9). Yield data from Montana (Black, 1982; Halvorson and

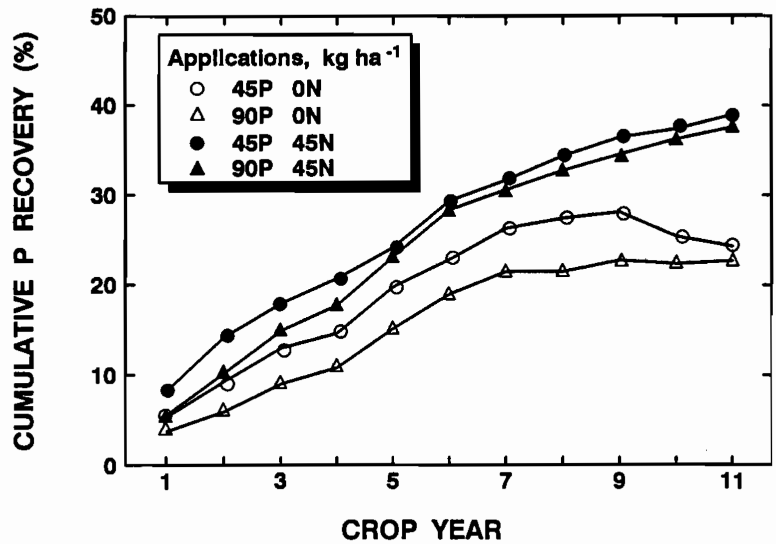

Figure 10. Cumulative P fertilizer recovery in wheat grain with harvest of each additional crop from a single P application with or without N applied each crop year. (Data adapted from Halvorson and Black, 1985b.)

Black, 1985a) also shows that N fertilization was needed to get optimum response of spring wheat to residual P fertilizer (Figure 7). Fertilizer P recovery also improved with each additional crop year (Figure 10). Thus, by having adequate P present and balancing the N needs of the crop based on yield potential, optimum yield and profit potentials can be realized.

Halvorson (1989) found that the presence of adequate levels of P also improved N uptake by irrigated winter wheat. Residual soil NO_3-N levels in the soil profile were significantly less where adequate P was present to optimize yield, thus, reducing the quantity of potentially leachable NO_3-N and ground water quality concerns. Phosphorus uptake and removal with the harvested grain generally increased as the soil NO_3-N plus fertilizer N level increased to an adequate level for maximum wheat yields. Estimated fertilizer P recovery of a single 67 kg P ha^{-1} application in the fall of 1983 in the harvested grain of three winter wheat crops was 7.2, 22.4, 27.6, 26.0, and 23.3% for the 0, 34, 67, 134, and 268 kg N ha^{-1} treatments, respectively. For the single 67 kg ha^{-1} P application, cumulative P fertilizer recovery was 2.1%, 4.9%, and 7.3% without N and 7.6%, 19.1%, and 27.6% with 67 kg N ha^{-1} added for 1984, 1985, and 1986, respectively. Thus, time and N fertilization significantly improved the recovery of fertilizer P in the harvested grain. The positive benefits of residual P fertilizer availability on irrigated and dryland crop yields demonstrate that P fertilizer-use efficiency needs to be evaluated over a longer period than just one

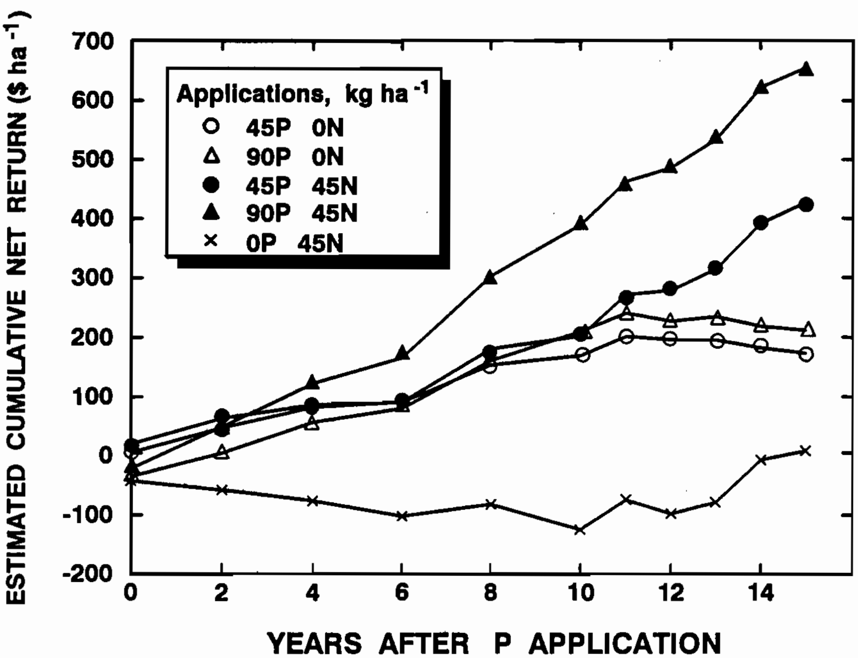

Figure 11. Cumulative net return with time as a function of a single P application with or without N applied each crop year. (Data adapted from Halvorson et al., 1986.)

crop year. It may need to be evaluated for more than 20 years, depending on P rate, soil type, and cropping system.

E. Economics of Fertilizer Phosphorus Management

Many farmers today consider themselves economically stressed as a result of rising production costs while crop prices have remained relatively constant, despite federal price support programs. Current farm management emphasis is on increasing input-use efficiency. Adequate levels of plant nutrients are essential for obtaining optimum economic yields while protecting the environment. By soil testing, more accurate fertilizer recommendations can be made by giving credit for residual N and P in the soil profile, thus helping farmers achieve the required nutrient balance without over- or under-investing in fertilizer. This will require that soils previously receiving banded P fertilizer applications be properly sampled to insure that the soil test accurately reflects the true P status of the soil.

The short- (1 to 2 crop yr) and long-term (>2 crop yr) economics of P fertilization need to be considered (Figure 11). The long-term economics of a

Table 4. Percent of soils testing medium or less in available P in north-central states

State	1984	1986
Illinois	59/70	38/50
Indiana	31	22
Iowa	66	44
Kansas	61	54
Kentucky	75	58
Michigan	31	28
Minnesota	33	33
Missouri	55	55
Nebraska	69	69
North Dakota	72	75
Ohio	42	38
South Dakota	76	76
Wisconsin	38	34
Average	54	48

Data adapted from Potash and Phosphate Institute, 1985 and 1987a.

large one-time application of P fertilizer can be profitable on some soils (Jose, 1981; Halvorson et al., 1986; Wagar et al., 1986). However, the short-term profitability may be marginal for a one-time large P application (>90 kg P ha^{-1}). Jose (1981) concluded there was a long-term economic advantage of a single high rate broadcast P application over an equal quantity of P band applied to several crops at a lower rate at dryland sites in Canada. Crop price, fertilizer cost, and current soil test P level will govern how much P can be profitably applied in a given year. The investment in P fertilizer needs to be amortized over several years, similar to machinery, in order to maximize wheat yields and optimize responses to N fertilization. Fixen and Halvorson (1991) point out that length of land tenure, initial soil P level, expected response to P, and expected lifetime of the investment, influence the level of P fertilizer that can economically be applied.

IV. Soil Testing for Phosphorus Availability

Soil testing is the best tool available to assess the need for fertilizer P. Soil test summaries for 1984 show that on an average, 54% of the soils in the north-central region tested medium or less in available P (Table 4). In 1986, the number had dropped to 48% indicating that some progress has occurred in reducing P deficiency. These data indicate that there are many soils, that if fertilized with P, would probably result in significant increases in crop yields.

In contrast, there are many soils in several eastern U.S. states that would not respond to P for many years (Table 1). Accurately assessing soil P availability and the quantity of P fertilizer required to alleviate any P deficiency becomes very important if maximum economic yields (MEY) are to be obtained. The current emphasis on the need for precise fertilizer rates to optimize grain yields necessitates that the short- and long-term economic and environmental impact of P fertilizer applications be evaluated.

Soil testing is essential if accurate, profitable-fertilizer recommendations are to be made. While soil testing doesn't directly tell us how much fertilizer P will be required for profitable crop production, proper interpretation of the analytical results will guide producers as to the amount of P required. Though questions about the validity of soil testing in predicting fertilizer needs have been raised, the basic problem probably stems not from the analytical results but from the interpretation of these results for making P recommendations (Leikam, 1987). Interpretation of these results and the development of recommended rates of P application should be based on research data (public and private), but the recommendations may need to be refined locally for each situation to obtain maximum benefit from the soil testing/recommendation process.

Soil testing success depends on collection of a representative soil sample and an accurate laboratory analysis. Many soil samples, from the appropriate soil depth, need to be collected from representative areas of the field. A field should be divided, avoiding odd or problem areas, into about 8 to 16 ha lots (40 ha lots may be more practical but less desirable) and 10 to 15 subsamples collected from each lot for an adequate composite sample. Different or special problem areas should be sampled separately. Clearly, a non-representative sample can be misleading and may be worse than no sample at all.

Soil test procedures to estimate plant available P commonly use a variety of chemical extraction methods, which are closely related to plant growth and uptake of P under certain climatic and soil conditions (Fixen and Grove, 1990). Alternative procedures, using anion exchange resin, isotopic P^{32}, buffer capacities, adsorption-desorption relationships, and quantity-intensity curves have been developed to assess plant available P. However, none has found widespread routine application, thus, the basic dilute solution extraction techniques used by soil test laboratories have changed little over the last 25 years. The most commonly used procedures are Bray (Bray and Kurtz, 1945), Mehlich (Mehlich, 1984), and Olsen (Olsen et al., 1954) extractants (Fixen and Grove, 1990).

Despite this lack of change in methodology, improvements in soil characterization, sampling, and fertilizer application techniques have made soil test P methodology and interpretation a weak link in the P recommendation process. Thus, further test development is necessary, particularly in assessing the sustainability of soil P fertility for different management strategies, which rely more heavily on precise fertilizer placement, non-incorporation of crop residues, and utilization of residual inorganic and organic P. These developments should assess both positional and chemical availability of P, soil test P-crop yield relationships, and the use of nonconventional soil test procedures.

A. Positional Availability

The accumulation of P in specific soil horizons with fertilizer P banding, continuous manure applications, and reduced tillage, will present sampling problems to determine subsequent fertilizer P requirements. For example, if location of the fertilizer band is known, what portion of samples should be collected on and off the band and if the band's location is not known, is a random sampling strategy adequate? Collection of 15 (Ward and Leikam, 1986; Shapiro, 1988) to 30 random samples (Hooker, 1976) have been reported to adequately reflect P availability in fields where P bands exist.

Sampling strategies for minimum and no-till conditions would be similar to conventional tillage situations when P has been broadcast applied. However, the sampling strategies described by Kitchen et al. (1990) should be followed for reduced- and no-till situations where P has been banded. When location of the P bands are known, sampling involves one-in-the-band soil sample for every 20 or 8 between-the-band samples for 76- and 30-cm band spacing, respectively (Kitchen et al., 1990). When band location is unknown, paired sampling (approximately 10) where a first random sample and a second sample 50% of the band-spacing distance from the first sample perpendicular to band direction, reduces soil test P variability over completely random sampling.

B. Chemical Availability

As shown in an earlier section, the mineralization of soil organic P can be an important process supplying available P in certain soils. Consequently, it is possible that soil P tests may be improved in certain situations by accounting for or giving credit to mineralizable organic P as well as inorganic P. Several studies have reported that potential soil P supply, as reflected by crop yields, was more closely estimated by including extractable organic P (Abbott, 1978; Adepetu and Corey, 1976; Bowman and Cole, 1978; Daughtrey et al., 1973). Bowman and Cole (1978) used a modification of the Olsen method (Olsen et al., 1954), which measured the total amount of P (inorganic plus organic) extracted by the reagent. Where other soil test P test methods are recommended, a similar adaptation may be used. As conditions of organic P extraction may not replicate the dynamic field conditions influencing organic P mineralization, caution must be used in relating amounts of extractable organic P to expected crop response.

With an increase in residual P levels and potential formation of less soluble P-rich compounds (Adepoju et al., 1982; Pierzynski et al., 1990), it is possible that current soil test P extractants and their interpretation may not adequately reflect residual P availability. It may also be necessary to develop procedures to credit soil test P levels for residual P fertilizer that had been previously banded when making P fertilizer recommendations.

C. Soil Test-Crop Yield Relationships

The relationship between the sodium bicarbonate P test and relative yield potential of wheat grown in a dryland wheat-fallow system (Figure 8; Halvorson, 1986a) was discussed in section III.C. Application of adequate P fertilizer to bring the soil test P level to about 21 mg P kg^{-1} (Olsen P test) on calcareous soils should optimize wheat yield potentials (Halvorson, 1986a, 1986b; Fiedler et al., 1987). The Bray-I P soil test is widely used for acid soils (pH <6.5) with similar soil test P levels (about 30 mg kg^{-1}) required to achieve maximum wheat yield potentials in the north-central Region (Agronomy Staff, 1989; Fiedler et al., 1987; Fixen, 1986; Oplinger et al., 1985). Bray-I soil test values considered adequate for wheat on acid soils range from about 25 to as high as 50 mg kg^{-1}, depending on yield goal and location. Usually, higher soil test values are desirable in northern areas where spring soil temperatures are cooler (Murphy, 1988). As more intensive crop management systems are adapted, particularly under reduced or no-till practices, higher soil test P levels may be required to optimize yields unless stratified P is redistributed within the root zone. In terms of efficient residual P utilization, it may be necessary to develop soil test-yield relationships for different soils by establishing different soil test P levels with one-time applications of variable P rates up to at least 200 kgP ha^{-1}, and then measuring crop response to each residual P level, using optimum farm management practices such as those used with the MEY concept.

D. Development of Nonconventional Procedures

The benefits of several nonconventional soil test methods should be evaluated further. For example, quantity (concentration of sorbed P) and intensity (solution P concentration) factors in soil P test procedures (Kuo, 1990; Moody et al., 1988), as well as the use of resin accumulators (Skogley et al., 1990; Yang et al., 1991) or iron oxide-impregnated paper strips (Fe_2O_3 strips) as a sink for plant available P (Menon et al., 1989a,b) have been proposed. Anion exchange resins more closely simulated soil P removal by plant roots and their action is generally independent of soil type; however, their widespread use in routine soil testing has been limited by slow and cumbersome methodology. Although exchange resins in membrane and spherical forms have been used in biochemical and medical research for some time, they have only recently received attention in soil testing (Sagger et al., 1990; Schoenau and Huang, 1991).

The amount of P extracted by Fe_2O_3 strips (strip P), was more closely related to both dry matter yield and P uptake of maize than Bray-I P for four soils ranging in pH from 4.5 to 8.2 (Menon et al., 1989a,b,c). Sharpley (1991) reported the Fe_2O_3 strips removed primarily physically-bound P (anion exchange resin P) from 203 soils representing all soil orders. As strip P was closely related to different P tests for soils on which the use of the test is recommended and with resin P (Sharpley, 1991), it is possible that the Fe_2O_3

strip may extract amounts of P closely related to plant availability for soils ranging widely in physical and chemical properties.

The close correlation between strip P and soil test P does not in itself justify adoption of the procedure to quantify P uptake. However, it emphasizes the potentially wide applicability of strips to estimate plant available soil P, including residual P, and suggests further evaluation of the method is warranted. For example, P extracted by Fe_2O_3 strips embedded in soil columns, was closely related to Bray-I P for acid and Olsen P for alkaline and calcareous soils (Menon et al., 1990). The potential use of the strips as a nondestructive method to measure *in-situ* soil P availability, may be of value in estimating banded residual P availability. In addition, dry matter yield and P uptake by maize from soils treated with rock P (RP) and partially acidulated RPs, was more closely correlated with strip P ($r^2 = 0.83$ and 0.88, respectively) than Bray-I P($r^2 = 0.53$ and 0.45, respectively) (Menon et al., 1989a). Acid extractants like Bray-I reagent can overestimate P from such soils by dissolving more P than would be available for plant use, whereas Olsen could underestimate P (Chien, 1978; Mackay et al., 1984). Consequently, strip P was more effective than other soil P tests in evaluating P availability from different RPs applied to soil.

V. Transport of P in Runoff

A. Forms

The transport of P in runoff can occur in soluble (SP) and particulate (PP) forms. Particulate P encompasses all solid phase forms, which includes P sorbed by soil particles and organic matter eroded during runoff and constitutes the major portion of P transported from conventionally tilled land (75-90%). Runoff from grass or forest land carries little sediment, and is, therefore, generally dominated by the soluble form. While SP is, for the most part, immediately available for biological uptake (Nurnberg and Peters, 1984; Peters, 1977; Walton and Lee, 1972), PP can provide a long-term source of P for aquatic plant growth (Bjork, 1972; Carignan and Kalff, 1980; Wildung et al., 1974).

In the past, most studies have measured only SP and total P (TP) transport in runoff. However, estimation of bioavailable P (BAP) transport in runoff is needed to estimate more accurately the impact of agricultural management practices on the biological productivity of surface waters. Bioavailable P represents P that is potentially available for algal uptake and is comprised of SP plus bioavailable PP (BPP). Although BPP can be quantified by algal culture tests (US EPA, 1971), these assays generally involve long-term incubations (100 days) and, thus, do not lend themselves to routine analysis. Hence, more rapid chemical extraction procedures have been used to simulate utilization of PP by algae (Hegemann et al., 1983; Sonzogni et al., 1982). Chemical extractants that have been used to measure the BPP content of eroded soil material are NaOH (Butkus et al., 1988; Logan et al., 1979; Sagher et al., 1975), NH_4F (Dorich et

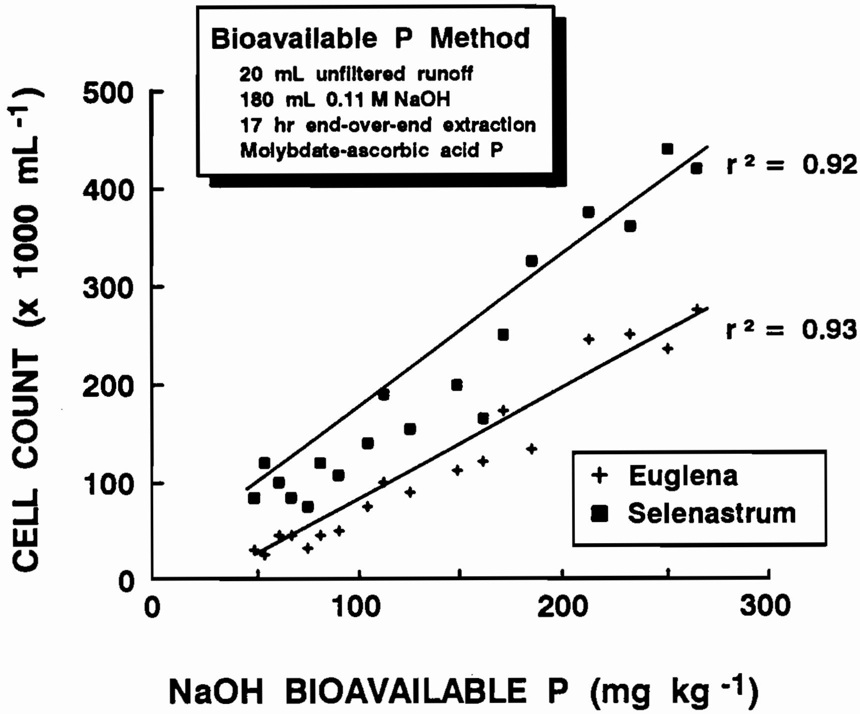

Figure 12. Relationship between the bioavailable P content of runoff sediment, extraction by NaOH and growth of P-starved algae during a 29-day incubation.

al., 1980; Porcella et al., 1970), and ion exchange resins (Armstrong et al., 1979; Hanna, 1989; Huettl et al., 1979).

Use of these methods has shown a wide range in percent bioavailability of P in suspended and deposited lake sediments (Sharpley and Menzel, 1987). However, these extractions require collection of a large volume of runoff to provide an adequate amount of sediment for analysis and are, thus, not applicable to the routine measurement of BPP transport in runoff. As a result there is a lack of information on the effect of agricultural management on BAP transport in runoff. To fill this gap, a chemical extraction procedure to estimate the BPP content of sediment transported in runoff was proposed by Sharpley et al (1991a). This procedure is a modification of the method of Dorich et al. (1985) and involves extraction of unfiltered runoff with 0.1 *M* NaOH for 17 hours (Sharpley et al., 1991a). Using this method, BPP contents of runoff from 9 Oklahoma watersheds, were closely related to the growth of P-starved algae (*Selenastrum capricornutum*) incubated with the same runoff sediments (Figure 12). Consequently, it is suggested that the extraction of P from unfiltered runoff by 0.1 *M* NaOH can be used as a rapid and interference-free method to routinely

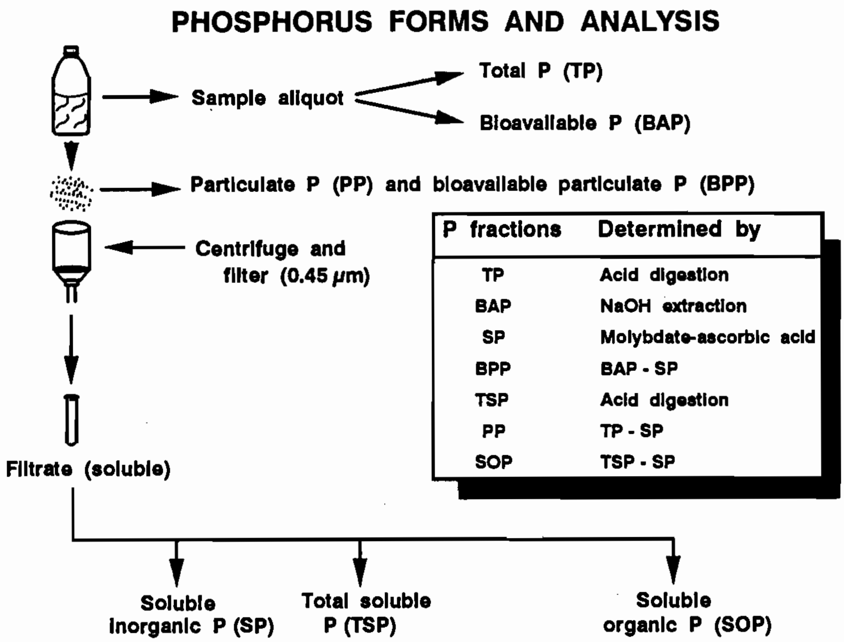

Figure 13. Phosphorus forms and their analysis commonly used in agricultural runoff studies.

estimate the BPP and BAP (BPP plus SP) in runoff. This method has been included in a schematic summary of P forms and analysis commonly used in runoff studies (Figure 13).

B. Processes

1. Soluble P

The transport of SP in runoff is initiated by the desorption, dissolution, and extraction of P from soil and plant material (Figure 2). These processes occur as a portion of rainfall interacts with a thin layer of surface soil before leaving the field as runoff. Simulated rainfall studies have shown this layer to range from 1 to 5 mm in depth (Sharpley 1985b). Although this depth has not been measured under field conditions, it is expected to be highly dynamic due to variations in rainfall intensity, soil tilth, and vegetative cover. The remaining rainfall percolates through the soil profile where sorption of P by P-deficient subsoils generally results in low concentrations of soluble P in subsurface flow. Exceptions may occur in organic or peaty soils, where organic matter may accelerate the downward movement of P together with organic acids and Fe and

Al (Fox and Kamprath, 1971; Singh and Jones, 1976; Duxbury and Peverly, 1978; Miller, 1979). Similarly, P is more susceptible to movement through sandy soils with low P adsorption capacities (Ozanne et al., 1961; Adriano et al., 1975; Sawhney, 1977) and in soils that have become waterlogged, where a decrease in Fe (III) content occurs (Ponnamperuma, 1972; Gotoh and Patrick, 1974; Khalid et al., 1977).

The extraction or leaching of P from plant material in different stages of growth and decay, may account for seasonal fluctuations and differences between watersheds in the transport of SP in runoff (Burwell et al., 1975; Gburek and Heald, 1974). This will be particularly true for watersheds under no till management (Barisas et al., 1978; Langdale et al., 1985; Pesant et al., 1987). Increased SP loss in runoff from alfalfa plots (33 g ha^{-1}) compared to forested (4 g ha^{-1}), corn (11 g ha^{-1}) and oat plots (16 g ha^{-1}), during several simulated rainfall events (7.4 to 12.2 cm^{-1}) over a 2-yr period, were attributed to larger amounts of P leached from alfalfa (Wendt and Corey, 1980). These differences in P loss have been partially explained by studies of nutrient release from vegetation which was cut, decaying, dried, and/or freeze-thawed (Sharpley and Smith, 1989a; Timmons et al., 1970; White, 1973). In a study of growing plants under simulated rainfall (6.4 cm^{-1}), Sharpley (1981) found that cotton, sorghum, and soybean plants could maintain a SP concentration of 0.02 to 0.13 mg L^{-1} in plant leachate (i.e., rainfall intercepted by vegetation). The contribution of plant leachate P to runoff losses was subsequently calculated from the difference in SP concentration of planted and bare soil. For mature plants (40 days after planting), receiving 50 or 100 kgP ha^{-1}, leached SP accounted for approximately 20% of SP transported in runoff for each crop (0.036 to 0.087 mg L^{-1}). However, when the plants were P deficient (no fertilizer P applied) or reached senescence (80 days after planting), plant leachate was the major source of P transported in runoff (up to 90% of 0.024 to 0.154 mg L^{-1}).

Many studies have investigated soil P desorption in relation to soil fertility and water quality, using a wide range in ionic strength and species of extracting medium (Sharpley and Menzel, 1987). However, few have used filtered runoff or lake water (Bahnick, 1977; Barlow and Glase, 1982) as the extracting medium, due to difficulties in preparing large volumes of filtrate of constant chemical composition. As P desorption is a function of the type of extracting medium (Ryden and Syers, 1977) and solution:soil ratio used (Hope and Syers, 1976; Barrow and Shaw, 1979), a standardized method is needed to relate P desorption to its transport and potential bioavailability in runoff. One such method may involve the use of Fe_2O_3 strips as a sink for algal available P. These strips have been used to investigate the desorption kinetics of reversibly adsorbed P in Dutch soils (Van der Zee et al., 1987) and more recently, successfully applied to the estimation of plant available soil P (Menon et al., 1989a, b).

2. Particulate P

As the sources of PP in streams include eroding surface soil, streambanks, and channel beds, processes determining soil erosion also control PP transport (Figure 2). In general, the P content and desorption-adsorption potential of eroded particulate material is greater than that of source soil, due to preferential transport of clay-sized material ($<2\ \mu m$). This has led to the determination of enrichment ratios (ER) for P, calculated as the ratio of the concentration of P in the sediment (eroded soil) to that in the source soil (Knoblauch et al., 1942; Neal, 1944; Rogers, 1941; Stoltenberg and White, 1953). More recently, Sharpley (1985c) observed that the enrichment of available P (Bray-1 - 2.45 and labile P - 2.89) was greater than for other P forms (total, inorganic, and organic) (1.48) for 6 soils using simulated rainfall. The relatively greater enrichment of available than total P forms was attributed to less aggregation of runoff sediment compared to source soil reducing the physical protection of P. Phosphorus reactivity, in terms of the desorption-adsorption characteristics; buffer capacity (1.49), sorption index (1.56), and equilibrium P concentration (EPC_o) (1.80), were also enriched in runoff sediment compared to source soil. The EPC_o is the SP concentration of water due to sediment sorption and desorption processes.

Processes controlling PP bioavailability are more complex than for PP transport, due to particle size sorting during transport and the contribution of less dense organic material to bioavailable P. Thus, the selective transport and enrichment of organic and fine material in runoff will increase PP bioavailability. This will be of particular significance to situations where manure has been applied. Despite a lack of information, increases in P bioavailability in runoff will be expected following manure application due to the increased transport of low density organic material and high solubility of manure P. However, the magnitude of these increases may differ between types of manure as a function of its density, form (wet or dry), organic matter content, and P content. Clearly, the dynamic nature of these interactions must be considered in determining both the short- and long-term potential of runoff to increase the biological productivity of receiving water bodies.

3. Changes in Bioavailablity during Transport

Transformations between SP and PP, occurring during transport in stream flow, can alter both the amount and bioavailability of P entering a lake, compared to edge-of-field losses (Figure 14). These transformations are accentuated by the selective transport of fine materials, which have a greater capacity to sorb or desorb P and will thus be important in determining the bioavailability of P transported. In addition, P may be taken up by aquatic biota and PP deposited or eroded from the stream bed with a change in stream flow (Meyer, 1979; Vincent and Downes, 1980). The direction and extent of P exchange between

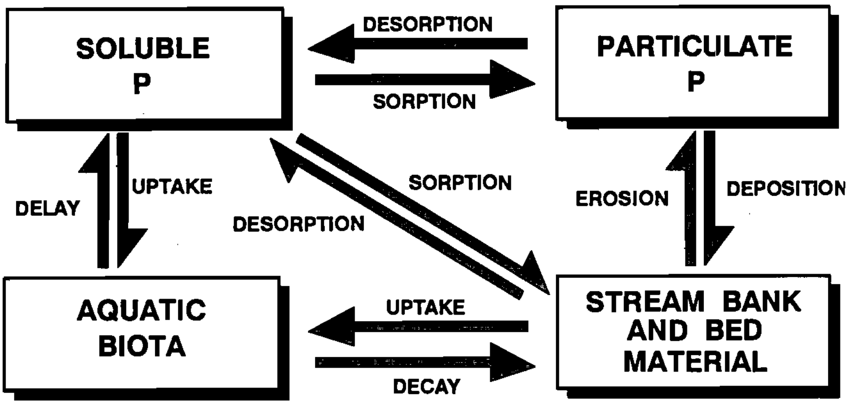

Figure 14. Phosphorus transformations during transfer from terrestrial to aquatic ecosystems.

SP and PP will depend on their relative concentration in stream flow, EPC_0 of the sediments contacted, and rate of stream flow. Several studies have shown that sediments rapidly sorb added P and concluded that the close agreement between EPC_0 and the SP concentration of runoff and stream flow indicated that sediments may determine SP concentration (Klotz, 1988; Meyer, 1979; Taylor and Kunishi, 1971). For example, if the SP concentration of stream flow falls below the EPC_0 of the particulate material contacted, P will be desorbed. However, if SP concentration increases above, the EPC_0, P may be sorbed. Soluble P concentrations of 0.10 to 0.13 mg L^{-1} of runoff from fertilized fields were reduced to 0.009 mg L^{-1} by sorption during movement downstream (Kunishi et al., 1972).

Clearly, changes in P bioavailability can occur between the point where it leaves a field to where it enters a water body. Consequently, the extent to which transformations between SP and PP occur during stream flow must be considered in assessing the impact of P transported in runoff as a function of agricultural management on the potential biological productivity of a receiving lake.

C. Amounts Transported

Amounts of P transported in runoff from uncultivated or pristine land is considered the background loading, which cannot be reduced. These inputs determine the natural status of a lake and, as will be seen later, may be sufficient to cause eutrophication. As we try to assess the impact of agricultural management on P loss in runoff, it becomes clear that little quantitative

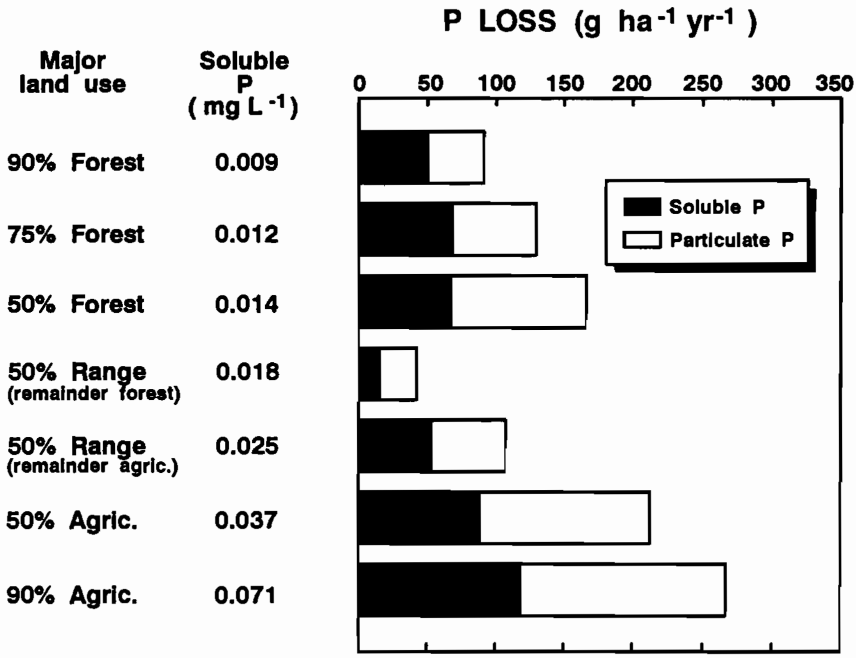

Figure 15. Phosphorus loss in runoff as a function of land use in the U.S. (Data adapted from Omernik, 1977.)

information is available on background losses of P from a given location before cultivation. Consequently, it is still difficult to quantify any increase in P loss following cultivation. These problems result mainly from the expensive and labor intensive nature of water quality monitoring studies, which are site-specific and impossible to replicate, due to spatial and temporal variations in climatic, edaphic, and agronomic conditions. Despite these problems, an investigation of published studies enables generalizations about the effect of agricultural management on P transport in runoff. Several surveys of U.S. watersheds (Omernik, 1977; Rast and Lee, 1978), have clearly shown that P loss in runoff increases as the portion of the watershed under forest decreases and agriculture increases (Figure 15). The loss of P from forested land tends to be similar to that found in subsurface or base flow from agricultural land (Ryden et al., 1973; House and Casey, 1988). In general, forested watersheds conserve P, with P input in rainfall usually exceeding outputs in stream flow (Hobbie and Likens, 1973; Schreiber et al., 1976; Taylor and Kunishi, 1971). As a result forested areas are often utilized as buffer or riparian zones around streams or water bodies to reduce P inputs from agricultural land (Lowrance et al., 1984a, b; 1985). However, the potential loss of P from agricultural land is to a large extent dependent on the relative importance of surface and subsurface runoff in the watershed.

1. Surface Runoff

Increased P loss in surface runoff have been measured after the application of fertilizer P (Table 5) and manure (Gilbertson et al., 1970; Holt et al., 1970; Sharpley and Syers, 1976). These losses are influenced by the rate, time, and method of application; form of fertilizer or manure; amount and time of rainfall after application; and vegetative cover. The portion of fertilizer P transported in runoff for the studies reported in Table 5, was generally greater from conventional compared to conservation tilled watersheds. However, fertilizer P application to no till corn reduced PP transport (McDowell and McGregor, 1984), probably due to an increased vegetative cover afforded by fertilization. Although it is difficult to distinguish between losses of fertilizer and native soil P, without the use of expensive and hazardous radiotracers, the loss of fertilizer P in surface runoff is generally less than 5% of that applied. Often, manure application rates are so large as to cause concern that a relatively greater portion of P will be lost in runoff compared to fertilizer and the capacity of soil to retain P against leaching may be exceeded.

A more detailed evaluation of the impact of tillage management on the amount and bioavailability of P transported in surface runoff is given for an intensive monitoring program of 28 watersheds in Oklahoma and Texas (Sharpley et al., 1992; Smith et al., 1991). These watersheds are representative of agricultural practices in the Southern Plains, with fertilizer P applications recommended by soil test P levels. Mean annual soil and P losses from sorghum, wheat, and native grass are summarized for a 5-yr period (1984-1989) in Figure 16. Soil loss from sorghum and wheat and P loss from wheat where similar amounts of P were added were appreciably lower from conservation than conventional tillage practices (Figure 16). However, P bioavailability increased. For example, mean annual SP concentrations were significantly greater from conservation compared to conventionally tilled watersheds, even though similar amounts of fertilizer P were applied to wheat. Further, the portion of PP that was bioavailable increased with an increase in vegetative soil cover. As a result, a greater portion of P transported from native grass and no till practices was bioavailable (SP plus BPP, 46 to 85%) than that from conventionally tilled practices (21 to 32%) (Figure 16). This difference results from a greater portion of finer-sized particles transported in runoff with reduced soil loss, leaching of crop residue P, and a build up of P in the surface soil with reduced mechanical mixing of the top soil under conservation tillage. Thus, it should be recognized that an increase in SP and BPP transport as a result of conservation tillage practices, may not bring about as great a reduction in the trophic status of a water body as may be expected from inspection of total loads only.

Table 5. Effect of fertilizer P on the loss of P in surface and subsurface runoff

Land use	P applied	Concentration Sol.	Concentration Partic.	Amount Sol.	Amount Partic.	Fertilizer loss Sol.	Fertilizer loss Partic.	Reference
	$kg\ ha^{-1}\ yr^{-1}$	$mg\ L^{-1}$		$kg\ ha^{-1}\ yr^{-1}$		%		
Surface Runoff								
Grass	0	0.18	0.24	0.50	0.67			Sharpley and Syers
	50	0.98	0.96	2.80	2.74	4.6	4.1	(1979), New Zealand
Grass	0	0.01	0.06	0.01	0.20			McColl et al. (1977),
	75	0.03	0.14	0.04	0.29	0.04	0.1	New Zealand
No-till corn	0	0.23	0.46	1.10	2.20			McDowell and
grain	30	0.57	0.51	1.80	1.60	2.3	27.3[a]	McGregor (1984),
								Mississippi
Conventional	15	0.07	3.57	0.30	15.10			
corn	30	0.11	9.71	0.20	17.50	+3.3[a]	16.0	
Contour corn	40	0.19	0.71	0.12	0.45			Burwell et al. (1977),
	66	0.25	1.27	0.15	0.76	0.1	1.2	Minnesota
Wheat-summer	0	0.30	1.80	0.20	1.40			Nicholaichuk and
fallow	54	3.70	7.40	1.20	2.90	1.9	2.8	Read (1978),
								Western Canada
Subsurface Runoff								
Alfalfa	0	0.18	-	0.12	-			Bolton et al. (1970),
(tile drainage)	29	0.21	-	0.19	-	1.0	-	Canada
Continuous corn	0	0.02	0.10	0.13	0.29			Culley and Bolton
								(1983), Canada
(tile drainage)	30	0.11	0.36	0.20	0.42	0.2	0.4	
Bluegrass sod	0	0.02	0.15	0.06	0.09			
	30	1.01	3.29	0.16	0.21	0.3	0.4	
Oats	0	0.02	0.09	0.10	0.19			
	30	0.42	1.10	0.20	0.30	0.3	0.4	
Alfalfa	0	0.02	0.11	0.12	0.20			
	30	0.37	1.03	0.20	0.31	0.3	0.3	

[a] Percent decrease in P loss from fertilized compared to check treatment.

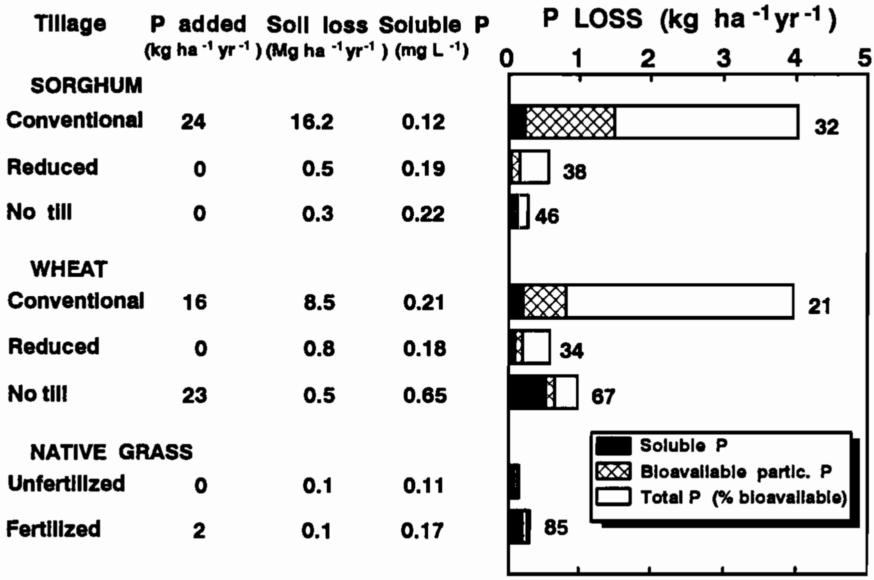

Figure 16. Phosphorus loss in runoff as a function of tillage management of several Southern Plains watersheds, averaged over a 5-year period.

2. Subsurface Runoff

The loss of P in subsurface runoff is appreciably lower than that in surface runoff, because of sorption of P from infiltrating water as it moves through the soil profile (Table 5). Subsurface runoff includes tile drainage and natural subsurface runoff, where tile drainage is percolating water intercepted by artificial drainage systems, such as tile or mole drains, thus accelerating its movement into streams. In general, P concentrations and losses in natural subsurface runoff are lower than in tile drainage (Table 5), due to the longer contact time between subsoil and natural subsurface flow than tile drainage, enhancing SP removal. Increased sorption of P from percolating water also accounted for lower TP loads from 1.0 to 0.6 m deep tiles draining a Brookston clay soil under alfalfa (Figure 17, from Culley et al., 1983). For the shallower drains, TP loads were about 1% of fertilizer P applied, whereas 1 m deep tiles exported about 0.6% of that applied (Figure 17).

Subsurface drainage of a soil can reduce P loss in runoff, via enhanced infiltration and thereby decreased runoff volumes. For example, Bengtson et al. (1988) found annual TP loss in runoff from a Commerce clay loam under corn in the lower Mississippi Valley, was reduced an average 36% in 6 yr following subsurface drainage with 1 m deep tiles (5.0 kg ha^{-1} yr^{-1}) compared to undrained soil (7.8 kg ha^{-1} yr^{-1}) (Figure 18). Of this TP loss reduction, only 8% was exported in tile drainage (0.3 kg ha^{-1} yr^{-1}) (Bengtson et al., 1988). The reduction

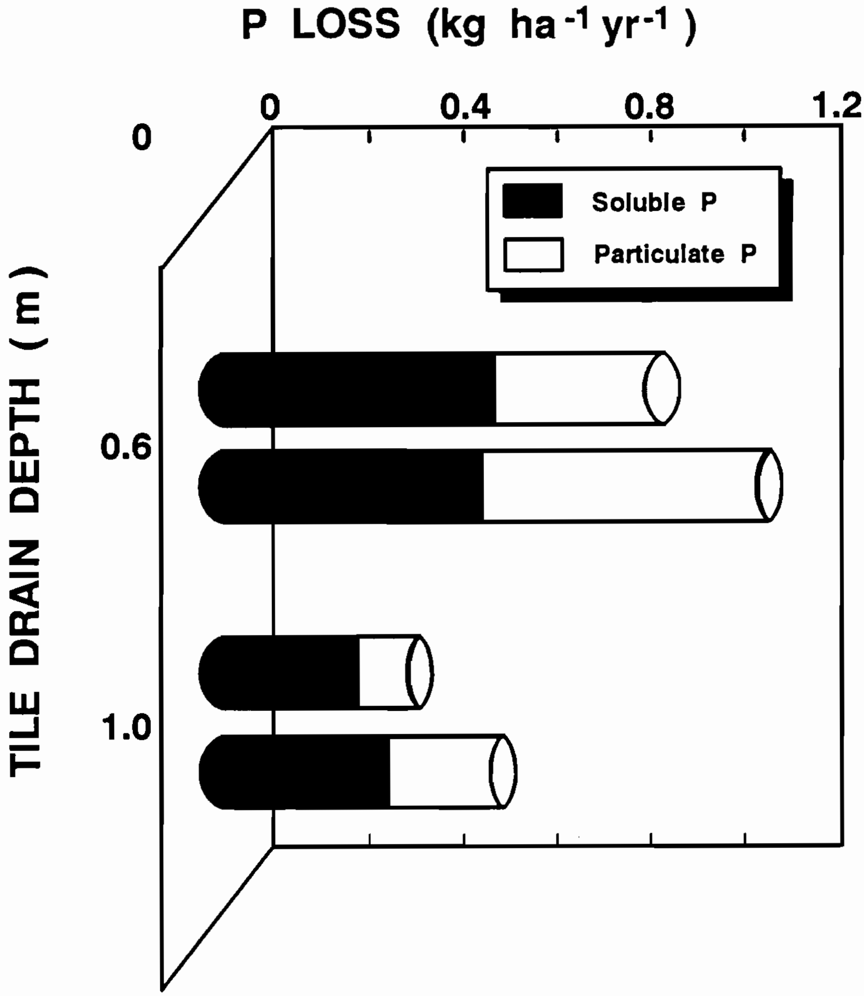

Figure 17. Phosphorus loss in tile drainage from a Brookston clay loam under alfalfa in Ontario, Canada as a function of tile drain depth. (Data adapted from Culley et al., 1983.)

in P loss must be weighed in terms of a potential increase in NO_3-N loss in tile drained fields (Bengtson et al., 1988).

VI. Predicting Phosphorus Availability and Transport

Accurate predictions of soil P availability and its transport in runoff are required to evaluate the relative effects of current and proposed agricultural management

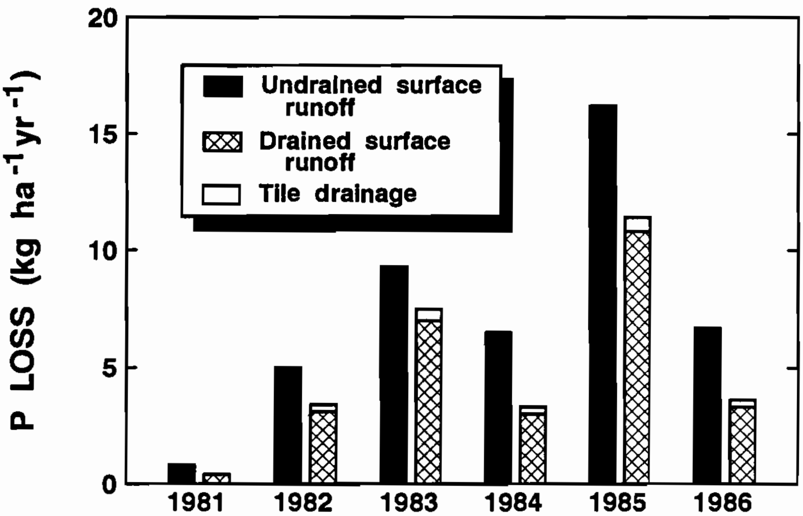

Figure 18. Phosphorus loss in surface runoff and tile drainage from a drained and undrained Commerce clay loam. (Data adapted from Bengston et al., 1988.)

practices on the biological response of a water body. Numerous comprehensive models have been developed to simulate the fate of agricultural chemicals in soil and their transport to surface waters, with the purpose of aiding selection of management practices capable of maximizing soil productivity and minimizing associated water quality problems.

A. Soil Availability

Claassen and Barber (1974) developed a conceptual model to simulate soil nutrient flux to plant roots, which has been used to predict P uptake by corn (Schenk and Barber, 1979a) and soybean (Silberbush and Barber, 1983). Although the model does not simulate P cycling in inorganic and organic pools, it has provided a quantitative assessment of the importance of several factors, such as root morphology (Schenk and Barber, 1979a, b), distribution (Kuchenbuch and Barber, 1987), soil temperature and water content (Barber et al., 1988; Mackay and Barber, 1984), liming (Alder and Barber, 1991), and fertilizer placement (Anghioni and Barber, 1980; Kovar and Barber, 1987) on the availability of P to several crops.

A detailed model of P cycling in grassland soils was developed by Cole et al. (1977). Plant uptake of soil solution P is assumed to be limited by diffusion. The solution P pool is replenished from a labile P pool, which in turn is supplied by slightly soluble P minerals, adsorbed P, and organic P mineralization. Many of the concepts used in this and soil organic matter models (Parton et al., 1983) were used in developing the Century model (Parton et al., 1987; 1988), which simulates the dynamics of C, N, P, and S cycling in the soil plant system using a monthly time step. These complex, long-time frame models, incorporate the effects of moisture, temperature, soil properties, plant phenology, and organic matter decomposition on nutrient flows and simulate many soil and plant processes. However, their use is limited by the availability of detailed soil and plant data. Even so, they have revealed gaps in our knowledge of processes such as the contribution of organic matter cycling to long-term soil productivity as a function of management (Parton et al., 1988). Thus, these models provide valuable direction for future research.

A user-oriented model, "Decide," was developed by Bennett and Ozanne (1972) and Helyar and Godden (1976) to simulate residual fertilizer-P recovery and used to give advice on fertilizer use to farmers. The model constructs a fertilizer response curve for each farming situation by combining research information with the farmers knowledge of past soil fertility and future yield goals. The model was developed using highly weathered and leached soils of western Australia and is presently being modified for use on slightly weathered soils of eastern Australia. Thus, the model only estimates fertilizer P recommendations for a limited range of soil types.

A more comprehensive model simulating soil P availability and fertilizer requirements (Jones et al., 1984b), is included in the EPIC model, which is composed of physically based components for simulating erosion, plant growth and related processes, and economic components to assess the cost of erosion and determining optimum management strategies (Erosion-Productivity Impact Calculator, Sharpley and Williams, 1990; Williams et al., 1984). The P model simulates uptake and transformations between several inorganic and organic pools in up to 10 soil layers of variable soil thickness. Fertilizer P is added to the labile inorganic P pool which rapidly achieves equilibrium with active inorganic P. Crop uptake from a soil layer is sensitive to crop P demand and the amounts of labile P, soil water, and root in the layer. Stover and root P are added to the fresh organic P pool upon their death and/or incorporation into the soil. Decomposition of fresh and stable organic matter may result in net immobilization of labile P or net mineralization of organic P. Regression equations were developed to estimate labile P from soil test P, organic P from total N or organic C, and fertilizer availability index from soil chemical and taxonomic characteristics (Sharpley et al., 1984a). Thus, except for soil test P, the minimum data set required to run the P model can be obtained from soil survey information.

Simulation of soil P transformations and availability under various long-term management strategies have been evaluated (Jones et al., 1984a). Changes in surface soil labile (i.e., Olsen P) and organic P contents of several Great Plains soils observed by Haas et al. (1961), were in most cases, accurately simulated by the EPIC model (Table 6). Measured and simulated values after the period of cultivation were not significantly different (at the 0.10 probability level). The

Table 6. Measured changes in surface soil organic and labile P within the Great Plains and changes simulated by the EPIC model

Location	Duration of study	Rotation	Organic P			Labile P		
				Cultivated			Cultivated	
			Virgin	Meas.	Sim.	Virgin	Meas.	Sim.
	years		mg kg^{-1}					
Havre, MT	31	SWF[a]	157	102	108	11	13	15
Moccasin, MT	39	"	308	183	169	14	14	20
Dickinson, ND	41	"	292	148	174	10	12	14
Mandan, ND	31	"	139	132	97	9	12	7
Sheridan, WY	30	"	120	93	86	12	14	9
Laramie, WY	34	"	142	91	96	13	24	9
Akron, CO	39	WWF[a]	115	82	81	26	45	19
Colby, KS	31	"	158	61	92	34	30	27
Hays, KS	30	W. Wheat	174	97	108	11	40	8
Lawton, OK	28	"	128	71	73	8	9	8
Dalhart, TX	29	Maize	84	39	53	17	13	9
Big Spring, TX	41	W. Wheat	55	30	29	12	12	6
Mean	34		156	94	97	15	20	13

[a] SWF and WWF represent spring wheat-fallow and winter wheat-fallow, respectively.
Data adapted from Haas et al. (1961).

slight overestimation of P forms may be due in part to the fact that soil erosion by wind and water, was kept minimal during the simulation. The decline in residual soil P availability (Olsen P) in 12 years following a single broadcast P application to a Williams loam under spring wheat in the Great Plains (Black, 1982), was also reliably simulated by the EPIC model (Figure 19). These examples illustrate the potential use of the model in the effect of long-term soil management of processes influencing soil P availability.

B. Transport in Runoff

Numerous comprehensive mathematical models have been developed to simulate the transport of agricultural chemicals in runoff, with the purpose of evaluating the relative effectiveness of management practices to minimize transport. These include, but are not limited to, ANSWERS (Areal Nonpoint Source Watershed Environment Response Simulation, Beasley et al., 1985), AGNPS (Agricultural Nonpoint Pollution Model, Young et al., 1989), GAMES (Guelph Model for Evaluating the Effects of Agricultural Management Systems on Erosion and

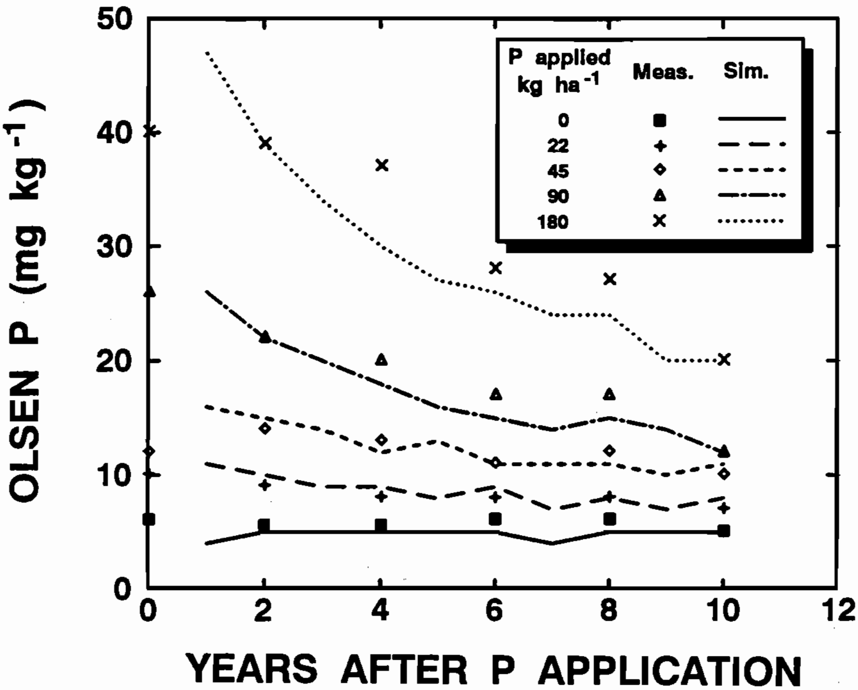

Figure 19. Effect of a single broadcast fertilizer P application on Olsen P content of a Williams loam under spring wheat at Culbertson, MT. (Data adapted from Black, 1982.)

Sedimentation, Cook et al., 1985), ARM (Agricultural Runoff Model, Donigian et al., 1977), and EPIC (Erosion-Productivity Impact Calculator, Sharpley and Williams, 1990).

Although physically-based descriptions of the various transport processes are used, the lack of data to drive the models and limited field data for testing has resulted in an over-simplified representation of P transport processes. In particular, equilibrium extraction coefficients have generally been used to predict SP, BPP has been assumed to be a constant proportion of total P, and no attempts have been made to predict BPP. Conceptually-based equations have been developed to describe the chemical and physical processes involved in the release from soil and transport of SP and BPP in runoff; they are described below. Soluble P, PP, BPP, and BAP concentrations were predicted using these equations and compared with values measured in runoff from 28 watersheds in Oklahoma and Texas, over a 5-yr period (Sharpley et al., 1992). These comparisons are presented (Figure 20) on a mean annual loss basis, which were calculated from concentrations and volumes of each runoff event.

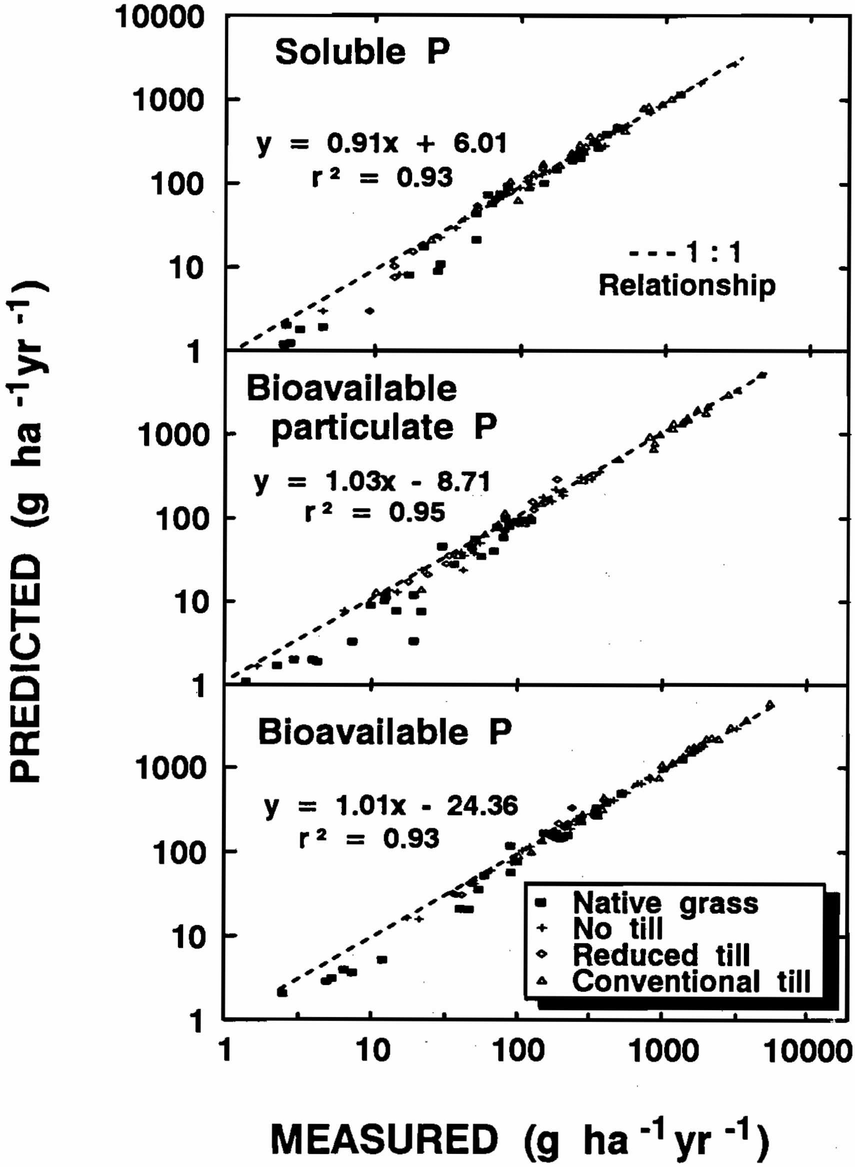

Figure 20. Relationship between measured and predicted P loss in runoff from the Oklahoma and Texas watersheds.

1. Soluble Phosphorus

The SP concentration of runoff is predicted by the following equation describing the kinetics of soil P desorption (Sharpley and Smith, 1989b):

$$P_r = \frac{K\, P_a\, B\, D\, t^{\alpha}\, W^{\beta}}{V} \qquad [1]$$

where P_r is average SP concentration of an individual runoff event (mg L^{-1}), P_a soil test (Bray-I) P (mg kg^{-1}) of surface soil (0-50 mm depth) before each runoff event, D effective depth of interaction between surface soil and runoff (mm), B bulk density of soil (Mg m^{-3}), t runoff event duration (min), W runoff water/soil (suspended sediment) ratio, V total runoff during the event (mm), and K, α, and β constants for a given soil. Values of D and Eq. [1] constants were estimated from soil loss (Sharpley, 1985b) and surface soil clay/organic C content (Sharpley, 1983), respectively. Accurate SP predictions were obtained for each management practice, covering a wide range in measured losses (1 to 2980 g ha^{-1} yr^{-1}, Figure 20). A general underestimation of SP loss in runoff from native grass and no till compared to the other watersheds was observed (Figure 20). This may result from an inadequate representation of the contribution of P release from vegetative material to P loss in runoff.

The extent of surface-soil vegetative cover will influence the degree of interaction between surface soil and runoff (D, Eqs. [1]), initiating P extraction and transport and the differential release of P from vegetation. It is well established that the release of P from vegetation can be an important source of P in runoff, which is influenced by several soil and crop factors such as soil nutrient status, soil water content, crop type, and growth stage (Burwell et al., 1975; Klausner et al., 1974; Sharpley, 1981; Timmons et al., 1970; Wendt and Corey, 1980). However, there has been limited success in simulating these processes, particularly for growing plants (Schrieber, 1990).

2. Particulate Phosphorus

The selective transport of clay-sized particles in runoff has led to the concept of enrichment ratios (ER) for P, defined as the ratio of the P content of runoff sediment to that of surface soil, used to predict particulate P transport (Menzel, 1980; Sharpley et al., 1985). Particulate P concentrations of runoff are calculated from total and bioavailable P contents of surface soil, respectively, using ER:

$$\text{Particulate P} = (\text{Soil total P}) * (\text{Sediment concentration}) * (\text{PER}) \quad [2]$$

$$\text{Bioavailable particulate P} = (\text{Soil bioavailable P}) * (\text{Sediment concentration}) * (\text{BIOER}) \quad [3]$$

where the units of soil total and bioavailable P are mg kg^{-1} and those for sediment concentrations in runoff are g L^{-1}. The enrichment ratios were predicted from soil loss (kg ha^{-1}) using the following equation developed by Sharpley (1985c):

$$\text{Ln (ER)} = 1.21 - 0.16 \text{ Ln (Soil loss)} \quad [4]$$

Accurate predictions of PP and BPP loss from each management practice were obtained, for a wide range in measured values (Figure 20). The enrichment ratio approach has also been successfully used for other U.S. regions and soils in AGNPS, CREAMS, GAMES, and ARM models to predict PP transport in runoff (Beasley et al., 1985; Cook et al., 1985; Donigian et al., 1977; Knisel et al., 1988). As for SP, BPP predictions were generally underestimated for the native grass and no till compared to reduced and conventionally tilled watersheds, particularly at low concentrations (Figure 20). This discrepancy may result in part from a lower soil loss from the grassed and no till watersheds (Figure 16).

As the predicted relationship between ER and soil loss (Eq. [3]) is logarithmic, predicted values of ER will be affected more than a unit quantity of soil loss at low values of loss (< 50 kg ha^{-1} yr^{-1}) than higher values (> 500 kg ha^{-1} yr^{-1}). Additionally, preliminary testing of Eq. [3], showed that this relationship varied between watersheds (Sharpley et al., 1985). It may, thus, be inappropriate to use constant slope and intercept values for different management practices, where the runoff-surface soil interaction may differ. For example, in grassed and no till practices, organic matter may contribute a greater proportion of particulate material transported. Consequently, making slope and intercept of Eq. [4] a function of factors affecting soil loss or runoff, such as rainfall intensity, vegetative cover, and management practice, should improve predictions. This may involve use of specific surface area and density of eroded material and enrichment of particle size fractions, rather than total sediment loss.

3. Bioavailable Phosphorus

Bioavailable P loss was calculated as the sum of predicted SP and BPP losses. Accurate BAP predictions were obtained (Figure 20). As both SP and BPP losses in runoff from native grass and no-till watersheds were slightly underestimated, predicted BAP losses were lower than measured values for these watersheds (Figure 20). However, above BAP losses of about 100 g ha^{-1} yr^{-1}, predictions closely followed a 1:1 relationship.

C. Model Limitations

There are many models available which simulate soil nutrient cycling and transport in runoff, facilitating identification of alternative management strategies and basic research needs. However, a major limitation is often the lack of input data to run a model, as many models require detailed information on soil physical, chemical, and biological properties as well as crop and tillage operations. As model output will only be as reliable as the data input, use of these models to provide quantitative estimates of P availability and transport in runoff under specific environmental conditions is limited. Thus, their use is recommended for comparison of relative effects of P management on soil and water productivity.

Some limitations to model use and application may be overcome by development and generation of adaptable data bases, which may fill information gaps. Weather generators and soil data bases are being developed to provide needed inputs, so that all a user needs to know is the soil name, location, and management practice to be imposed. Linkage of soil productivity and water quality models may be necessary to evaluate the effect of agricultural management strategies on water quality. Such an effort is underway to combine the soil productivity and water quality models of EPIC and GLEAMS. In addition, it is still difficult to relate P loss in runoff as a function of watershed management to the biological productivity of a receiving water body. Models simulating watershed (AGNPS) and lake (FARMPND) processes were linked by Summer et al. (1990). However, a lack of adequate field data limits rigorous testing of their ability to simulate a lake's response to changes in agricultural management and weather conditions.

D. Indexing Phosphorus Availability

Because of the above limitations, indices which identify soil and management practices that may enrich the P content of surface waters should be developed. The basic principle of water quality indices is to synthesize data such as analytical results and experimental information, by means of a simple quality vector or algorithm. Hence, the index will be a simplified expression of a complex combination of several factors or parameters, that make information more easily and rapidly interpretable than a list of numerical values. Such an index would be helpful for field personnel with limited resources working with land owners, to identify sensitive areas and suggest management alternatives to reduce the risk of water quality problems associated with P.

For initial development of such an index, critical factors influencing soil P availability, transport in runoff, and susceptibility of receiving water bodies to P inputs, have been identified in Figure 21. These are: erosion potential; runoff

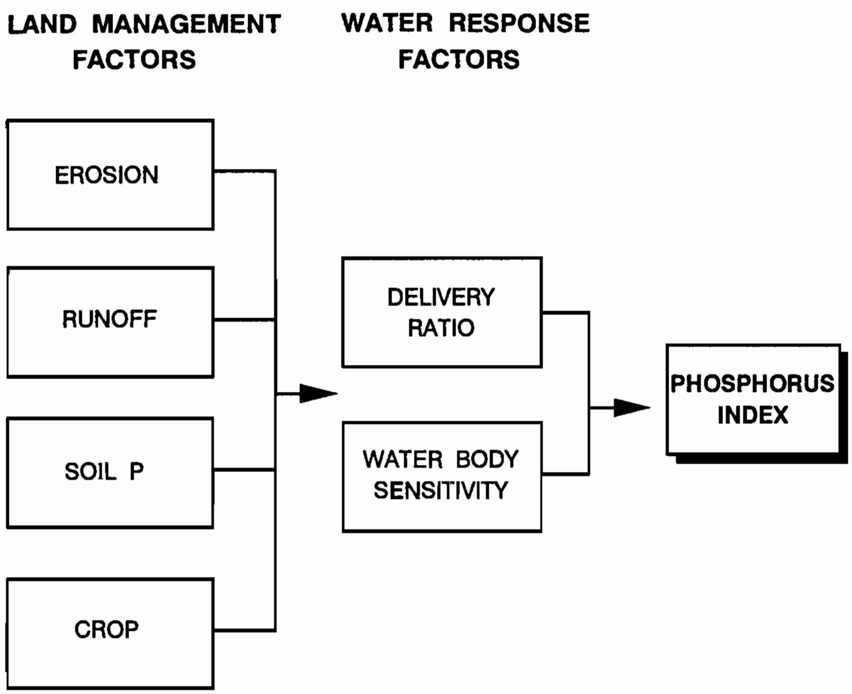

Figure 21. Components of a soil and water P availability index.

potential; soil P; crop type and yield goals; water delivery ratio; and waterbody sensitivity.

As the major portion of P transported in runoff from cropland is associated with eroded soil material, parameters describing ***soil erosion and runoff potential*** will be primary components of the index. An estimate of soil erosion and runoff potential may be obtained from soil survey and site information, such as soil texture, permeability, and cover; slope length and gradient; crop management and tillage; rainfall; and erosion control practices. As a first approximation, this information is used with equations predicting erosion such as the Universal Soil Loss Equation (USLE, Wischmeier and Smith, 1965; 1978), Modified USLE (MUSLE, Williams, 1975), Rangeland USLE (RUSLE, Renard et al., 1991), or Water Erosion Potential Predictor (WEPP, Laflen et al., 1991).

The ***soil P factor*** (Figure 21), includes available soil P content and P sorption capacity. Because of the importance in available soil P in determining P loss in runoff and its dynamic nature in surface soil, an estimate of its content will be required for the index. Soil test P may be used as a first approximation. However, an estimate of the bioavailable P content of surface soil (Fe_2O_3 strip or 0.1 *M* NaOH extractable) may be more appropriate for indexing the potential impact of P transport on accelerated eutrophication. Because of the ease of bioavailable soil P determination and a lack of general correlation with soil test P (Sharpley and Smith, 1992; Wolf et al., 1985), its measurement should not be substituted by soil test P. The P sorption capacity of a soil will determine the disposition of added P into available and unavailable forms, which will ultimately influence the portion of P transferred to runoff in soluble and particulate forms. For example, in a soil of low sorption capacity, more added P will remain in an available form and be more susceptible to loss as SP than an equivalent addition to a soil of high P sorption capacity. Phosphorus sorption may be determined by a single point isotherm (Bache and Williams, 1971) or from regression equations relating it to associated soil properties, such as pH, clay, organic C, and $CaCO_3$ content (Sharpley et al., 1984a).

Crop factors such as type and yield will influence the degree of vegetative soil cover and amount of P removed from the soil system, if the crop is harvested. In addition to influencing soil erosion, type of crop cover can affect the amount (Sharpley, 1981; Burwell et al., 1975; Timmons et al., 1970) and relative bioavailability of P transported (Sharpley et al., 1992; McDowell and McGregor, 1984; Wendt and Corey, 1980). For example, Muir et al., (1973) found a significant correlation between SP concentration in major streams of Nebraska and legume acreage statewide. They suggested that SP concentration in the Platte River, Nebraska, may reflect P leached from alfalfa residues, carried in runoff during the growing season. Soil P utilization also varies from crop to crop, which will influence the amount of available soil P that could be transported in runoff.

The effect of in-stream physical processes and chemical transformations on the amount and form of P transferred from edge-of-field to lake, can be summarized by a ***P delivery ratio***. Physical processes involve deposition and erosion of stream bank and bed material with a change in stream flow velocity. These processes accentuate P transformations by the overall selective transport of fine materials, which have a greater capacity to sorb or desorb P. The relationship between the form of P transported in runoff and P sorption capacity was demonstrated by data from several unfertilized watersheds in the Southern Plains (Sharpley and Smith, 1990). A decrease in SP concentration with an increase in suspended sediment concentration of runoff was found to be a function of P sorption capacity of the dominant soil type at each watershed (Sharpley and Smith, 1990). Thus, SP concentration can be modified to a great extent during transport in runoff by suspended sediment from source soil having a higher sorption capacity. Clearly, in-stream changes in the relative amounts of SP and PP transported, approximated by the P delivery ratio, will influence the short- and long-term impact on lake productivity.

The ***sensitivity of a water body*** to either SP or PP inputs may determine management control strategies on source areas. For example, depth of photic zone, degree of surface mixing, development of reducing conditions at the water-sediment interface, and water residence time of a water body will influence its sensitivity to P inputs. If properties of the water body are such that particulate material and associated P rapidly settle from the water column (i.e.,

Table 7. An indexing to rate the potential P loss in runoff from given site characteristics

Site characteristic (weight)	**Phosphorus Loss Potential (Value)**				
	None (0)	Low (1)	Medium (2)	High (4)	Very High (8)
Transport Factors					
Soil erosion (1.5)[a]	Negligible	<10	10 - 20	20 - 30	>30
Irrigation erosion[b] (1.5)	Negligible	Tailwater recovery for QS <6 for very erodible soils or QS <10 for others	QS >10 for erosion resistant soils	QS >10 for erodible soils	QS >6 for very erodible soils
Runoff class (0.5)	Negligible	Very low or low	Medium	High	Very high
Phosphorus Source Factors					
Soil P test (1.0)	Negligible (0)	Low (1)	Medium (2)	High (4)	Excessive (8)
P fertilizer application rate (0.75)[c]	None applied	1 - 15	16 - 45	46 - 75	>76

P fertilizer application method (0.5)	None applied	Placed with planter deeper than 5 cm	Incorporated immediately before crop	Incorporated >3 months before crop or surface applied <3 months before crop	Surface applied >3 months before crop
Organic P source application rate (0.%)[c]	None applied	1 - 15	16 - 30	30 - 45	>45
Organic P source application method (1.0)	None	Injected deeper than 5 cm	Incorporated immediately before crop	Incorporated >3 months before crop or surface applied <3 months before crop	Surface applied >3 months before crop

[a] Units for soil erosion are Mg ha^{-1}.

[b] Q is irrigation furrow flow rate (gal min^{-1}) and S is furrow slope (%).

[c] Units for P application are kg P ha^{-1}.

a deep, stratified lake of long water residence time), then control of SP inputs may be of greater short-term benefit in reducing biological productivity. The trophic response or sensitivity of a given water body to P input may be approximated by an empirical equation, such as those developed by OECD (1982), Vollenweider and Kerekes (1980), and Jones and Lee (1982).

The main land management and water response factors that should be considered in developing an index to assess the risk of agricultural P loss in runoff to degrade water quality are described above. These factors are incorporated into a P index being developed by a team of scientists (Phosphorus Indexing Core Team - PICT), led by the USDA-SCS, National Water Quality Technology Development Staff. The index is outlined in Tables 7 and 8. Each site characteristic has been arbitrarily assigned a weighting, assuming that certain characteristics have a relatively greater effect on potential P loss than others. Each site characteristic is given a rating value (Table 7), although each user must establish a range of values for different geographic areas.

As discussed earlier, erosion and runoff class can be obtained from soil loss equations (i.e., USLE, RUSLE) and soil survey data (i.e., estimated soil saturated hydraulic conductivity and slope), respectively. Irrigated erosion is calculated as the product of flow rate in the furrow (Q) and furrow slope (S). Soil test P is represented by the recommended procedure for each state. At the present time, the user must categorize soil test P into low, medium, high, and very high based on regional experience. Categories for rate and method of fertilizer and organic P application are self-explanatory (Table 7). In general, increasing from low to very high category values for application method, depict longer surface exposure time between P application, incorporation, and crop utilization.

An assessment of site vulnerability to P loss in runoff is made by selecting the rating value for each site characteristic from the P index (Table 7). Each rating value is multiplied by the appropriate site characteristic weight factor. Weighted values of all site characteristics are summed and site vulnerability obtained from Table 8. A hypothetical site is used as an example, where soil erosion is 25 Mg ha^{-1} (weighting $\times$ value; $1.5 \times 4 = 6$), irrigated erosion is not applicable (0), runoff class is medium ($0.5 \times 2 = 1$), soil test P is medium ($1 \times 2 = 2$), 20 kg P ha^{-1} of fertilizer ($0.75 \times 2 = 1.5$), and 50 kg P ha^{-1} of animal manure ($0.5 \times 8 = 4$) are broadcast in early spring prior to planting ($1 \times 4 = 4$). The sum of these weighted values (6, 0, 1, 2, 4, 1.5, 4, and 4) is 22.5, which has a high site vulnerability (Table 8). In this hypothetical situation, conservation measures to minimize erosion and runoff as well as a P management plan should be implemented to reduce the risk of P movement and probable water quality degradation.

The index is intended for use as a tool for field personnel to easily identify agricultural areas or practices that have the greatest potential to accelerate eutrophication. It is intended that the index will identify management options available to land users that may allow them flexibility in developing control strategies.

Table 8. Site vulnerability to P loss as a function of total weighted rating values from the index matrix

Site vulnerability	Total index rating value
Low	<10
Medium	10 - 18
High	19 - 36
Very High	>36

VII. Management Challenges

A. Minimizing Phosphorus Transport

Efforts to minimize P transport in runoff can be divided into fertilizer management and erosion control measures. Emphasis should be placed on minimizing the potential sources of P for transport.

1. Fertilizer Management

Efforts to reduce P losses to surface runoff by fertilizer management, should include subsurface placement, credit for alternative P sources such as manures, and utilization of accumulated residual P.

a. Subsurface Placement

With the increasing adoption of conservation tillage and more efficient agricultural systems, where P fertilizer and/or manure is broadcast on land which may not be tilled for several years, it will be necessary from a water quality perspective to minimize excessive surface soil accumulations of P. Where possible, this may be achieved by subsurface placement of fertilizer and manure away from the zone of removal by surface runoff. Preliminary data on fertilizer P management under conservation tillage systems in the Midwest region of the U.S. (Rehm, 1991), show that without this redistribution of stratified P, higher soil test P levels may be required for optimum production compared to conventional tillage practices.

b. Alternative Sources

Recent economic pressures have resulted in a diversification of agricultural systems, which in Alabama, Arkansas, Delaware, and Oklahoma for example, have been a rapid increase in poultry and swine operations. Although manure is a valuable source of nutrients, utilization of large amounts of accumulated animal manure is an increasing problem facing the industry. Consequently, manure can be utilized as a supplement or alternative to fertilizer P, preferably by subsurface placement or injection. However, it may be necessary in the future to establish cost sharing programs or subsidies, involving the consumer, producer, and farmer to transport excess manure from areas of high to low intensity production.

Because of the rapid fixation of fertilizer P by soil and conversion to unavailable forms, application rates usually exceed starter requirements, in all soils except those with very high P test values. A lack of agronomically effective slow release P fertilizers has limited the reduction in application rates of traditional soluble P fertilizers. However, the use of new sources of more reactive North Carolina RP material, adoption of more efficient systems using naturally occurring amendments, and inclusion of potentially soil acidifying forage legumes in crop rotations, may enhance the potential agronomic and economic benefit of RP as a slow release fertilizer on acid soils. The use of RP may decrease the potential loss of P in runoff, via banded or subsurface application in combination with smaller amounts of soluble fertilizer P. Research is needed, however, to evaluate the relative agronomic, economic, and environmental benefit of manure and RP as alternative sources of P as a function of different climatic conditions, soil types, and management systems. This should determine under what conditions RP can compete with high analysis fertilizers due to greater transportation costs of the former.

c. Utilization of Residual Soil P

In some cases, continued manure application has resulted in an excessive accumulation in surface soil P. For example, soil test P contents (Bray-I) in the surface 10 cm of soil receiving long-term (8 to 35 years) poultry and swine manure (Sharpley et al., 1991b) and cattle feedlot waste (Sharpley et al., 1984b), were increased up to 38-fold compared to untreated soils (Figure 22). We estimated the P concentration of a 2.5 cm runoff event of 20 kg ha^{-1} soil loss from the treated and untreated soils, using the predictive equations presented earlier (Eqs. [1] to [4]) with measured soil P content. These runoff and soil loss values are means for all events occurring on grassed watersheds in the Southern Plains during the last 15 years. Predicted P concentrations of runoff from treated soils were dramatically higher than from untreated soils, if runoff occurs (Figure 22). Under grass, erosion is minimal and, thus, most of the transported P will be in a bioavailable form (57 to 98%). Although this is a hypothetical runoff situation, it clearly emphasizes the need to carefully manage repeated applications of manure over a long period of time to minimize the potential for runoff to occur from these soils.

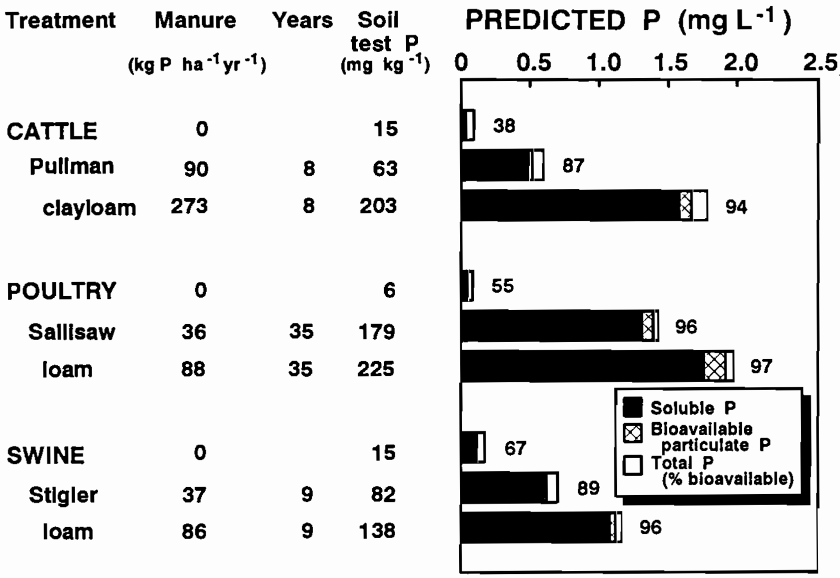

Figure 22. Effect of animal manure application on soil test P (Bray-I, 0-10 cm) and predicted P concentration of runoff from three soils in Oklahoma and Texas.

Brown et al. (1989) investigated the effect of various animal manure control practices on nonpoint source P loading to the eutrophic Cannonsville Reservoir in the West Branch of the Delaware River watershed, New York. Reductions in P loss from dairy operations of 50 to 90% were shown to be achievable using practices that reduced the volume of runoff from these areas (Brown et al., 1989). However, the contribution of P in runoff from dairy operations to P loss from the West Branch watershed, was substantially less than that from corn, on which manure had been spread. Hence, manure spreading schedules that guide the location and timing of spreading had the potential to reduce P loading from the area studied by as much as 35%. However, demands on farmer's daily schedules often limit the practicality of precise timing of manure applications. As a result, application timing is possibly the greatest practical obstacle to better manure management, with many best management practices needing to be done at the busiest times of the year for farmers.

2. Erosion Control

a. Conservation Tillage

Because of the fixation of P by soil, practices that reduce erosion will also reduce total P losses in runoff. Soil erosion and associated P transport may be reduced by increasing vegetative and residue ground cover through reduced tillage practices (see Figure 16). As a trade-off to this benefit, the concentration of SP in runoff from conservation tillage wheat and sorghum was greater than from conventional practices, even though similar or less P fertilizer, respectively, was added (Figure 16). Several other studies have suggested that conservation tillage may increase the bioavailability of P transported in runoff (McDowell and McGregor, 1980; Wendt and Corey, 1980). Consequently, implementation of conservation tillage alone may not reduce the potential for P-associated eutrophication of surface waters, as great as may be expected from inspection of total P loads.

In an attempt to improve the trophic status of Lake Erie, the U.S. Army Corps of Engineers (1982) showed that nonpoint source P load needed to be reduced by about 26% to reduce lake eutrophication significantly. From field experimentation and calculation, Forster et al. (1985) concluded that accelerated implementation of best management systems, which includes conservation tillage on soils suited to those practices on the U.S. side of the Lake Erie basin, can achieve the required P load reduction in 20 years.

b. Buffer Strips/Riparian Zones

In addition to conservation tillage, filter strips or zones can effectively reduce sediment and PP transport in runoff from cropland or animal production facilities. Vegetative filter strips are bands of planted or indigenous vegetation and riparian zones, are normally areas of native forest or woodland.

Dillaha et al. (1989) reported that a vegetative filter strip of orchardgrass (*Dactylis glomerata*) effectively reduced sediment and PP loss in runoff from bare Groseclose silt loam plots (11% slope and 5.5 by 18.3 m) in Virginia (Figure 23). The loss of soil and PP in two 60 min and four 30 min rainfalls of 50 mm hr^{-1} was reduced a respective 82 and 76% for a 4.6 m strip and 97 and 93% for a 9.1 m strip compared to no strip. Although SP loss in runoff was reduced 44% by the 9.1 m strip, SP increased 78% with the 4.6 m strip. Dillaha et al. (1989) attributed the SP increase to lower removal efficiencies for soluble nutrients and the release of previously trapped PP in the filters.

Because of the low installation and maintenance costs and effectiveness, filter strips are approved cost-share practices in many states and were recently incorporated into the USDA's Conservation Reserve Program (Alder, 1988). However, inadequate information on their effectiveness in reducing SP losses and on the dynamics of sediment and nutrient transport in the strips has resulted in strip installation in areas where they are inappropriate because of topographic limitations. Detailed reviews of filter strips are given by Dillaha et al. (1987) and Magette et al. (1987).

Riparian forests adjacent to surface waters have also been shown to reduce P inputs from agricultural runoff (Lowrance et al., 1985; Schlosser and Karr, 1981; Yates and Sheridan, 1983). For example, Peterjohn and Correll (1984)

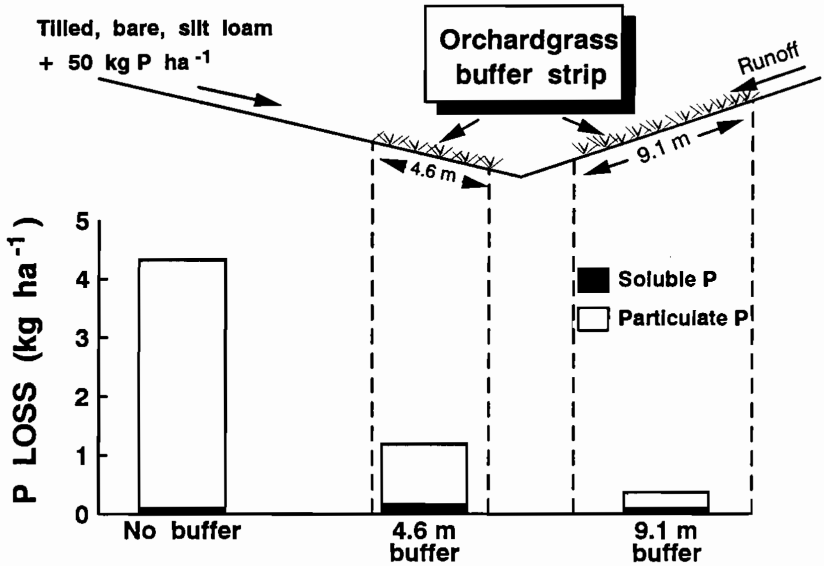

Figure 23. Phosphorus loss in runoff from Groseclose silt loam of 11% slope as a function of length of orchardgrass buffer strip. (Data adapted from Dillaha et al., 1989.)

conducted a detailed study of nutrient dynamics in a 16.3 ha agricultural watershed adjacent to Chesapeake Bay, Maryland. In the watershed, 10.4 ha was planted with corn and 5.9 ha as a riparian forest of broadleaved, deciduous trees. Peterjohn and Correll (1984) found that the total P input from cropland to riparian forest in surface runoff and ground water was 3.4 and 0.091 kg ha^{-1} yr^{-1}, respectively (Figure 24). Phosphorus export in stream flow from the riparian forest was reduced 80%, with similar amounts exported in surface runoff (0.43 kg ha^{-1} yr^{-1}) and ground water (0.30 kg ha^{-1} yr^{-1}). Clearly, riparian forests can reduce the transfer of P in agricultural runoff to receiving surface waters. Thus, coupled natural and agricultural systems within a watershed may reduce the occurrence of P-associated eutrophication. Even so, vegetative filter strips or riparian zones should not be relied upon as the sole or primary means to reduce P losses in runoff from agriculture (Magette et al., 1989).

B. Management Implications

It is apparent that several questions regarding the management of P availability in soil and water systems, must be answered to minimize P transport in runoff and bring about a further improvement in both soil productivity and surface water quality. These questions involve manure management based on P or N;

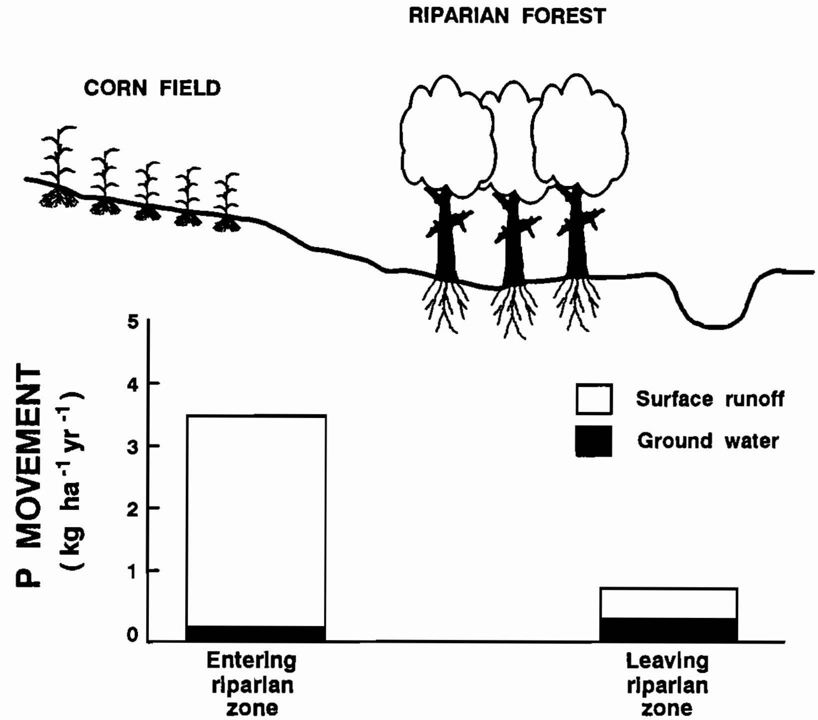

Figure 24. Amounts of total P entering a riparian forest from a corn field and leaving the forest in stream flow for a Maryland watershed. (Data adapted from Peterjohn and Correll, 1984.)

management effects on the dynamics of soil P cycling; tillage, crop, and residue management; soil test methodology and interpretation; and impact assessment.

1. Manure Management

Manure is a valuable source of both N and P for crop uptake in efficient management systems. However, continual long-term applications of manure have increased NO_3-N leaching (Cooper et al., 1984; Ritter and Chirnside, 1984) and the potential for P loss in runoff. For example, Liebhardt et al. (1979) applied poultry manure at rates of 0 to 179 Mg ha^{-1} (wet wt. basis) to sandy soils in Delaware and found a direct relationship between application rate and NO_3-N concentrations in ground water in excess of the recommended 10 mgL^{-1} limit (U.S. EPA, 1976). In addition, the buildup of P at the surface of soils receiving continual long-term manure applications has increased the potential for P enrichment of associated surface waters (Magette, 1988; Sharpley, 1991; Sims, 1992). Consequently, should manure application rates be based on soil N or P?

This question may be partially answered by determining if the soil, to which manure is to be applied, is susceptible to runoff and erosion or leaching as by the P indexing procedure (Tables 6 and 7). If the potential for runoff or erosion from an application site exists, then P should be a priority management consideration.

Currently, most manure application rates are based primarily on the management of N to minimize nitrate-N losses by leaching. In most cases, this has led to an increase in soil P levels in excess of crop requirements due to the generally lower ratio of N:P added in manure than taken up by crops. For example, dairy, beef, swine, sheep, and poultry manure has an average N:P ratio of 4.1 (Gilbertson et al., 1979), while the N:P requirement of major grain and hay crops is 7.3 (Fertilizer Handbook, 1982).

Sims (1992) emphasized that basing manure applications on P rather than N management, would present several problems to many farmers. For many crops and soils, manure application rates would be low if based on soil P, requiring farmers to identify larger areas of land to dispose of generated manure. In addition, farmers presently relying on manure to supply most of their crop N requirements may be forced to buy commercial fertilizer N, instead of using their own manure N. Clearly the development of environmentally sound management systems for the use of P originating from manure is a challenge from both research and environmental standpoints. For example, basing recommendations on soil P levels may resolve the issues of P enrichment of surface water, but place unacceptable economic burdens on farmers.

2. Soil Management

In conservation tillage systems, surface crop residues minimize soil water evaporation and erosion. Similarly, in other sustainable systems where cover crops are being increasingly included, the cover crop is killed before maturity and left in the soil to minimize water and light competition with the subsequent cash crop. What effect will this have on the amount and bioavailability of P in runoff? If the crop residue is left on the surface or occasionally plowed into the soil, is the potential loss of residue P in runoff affected? Crop residue management can affect P cycling and availability as a function of residue amount, type, and degree of incorporation with tillage. Sharpley and Smith (1989a) found that mineralization and leaching of residue P was greater when the residue of several crop types was surface applied compared to incorporated. A greater amount of residue will increase the amount of P being cycled and, particularly if left on the surface of the soil, will reduce evaporation losses and keep surface soil moist for more days during the growing season, thereby enhancing microbial activity and mineralization.

Under conservation tillage practices or continuous heavy fertilizer, manure or sludge applications, P may accumulate in certain soil horizons. Thus, is it possible to select a "scavenger" crop that may have a higher affinity or requirement for P and thereby reduce soil nutrient stratification? Alfalfa for example, has reduced subsoil nitrate accumulations (Mathers et al., 1975). May the same be true for surface soil accumulations of P? It is possible that by utilizing residual soil P, careful crop selection will reduce the amount of nutrients potentially available to be transferred to surface waters. However to be successful, the "scavenger" crop and P must be economically sustainable and leave the farm and move to a P-deficient site.

The benefits of improved technology and information on fertilizer and residue management on the fate of P in soil and water, are to a large extent dependent upon reliable soil test procedures. However, with an increase in soil P stratification and amounts in organic forms under no-till systems, are soil test procedures adequate to determine positional and chemical availability? This may be of particular importance to conservation tillage systems and where a cover crop is returned to the soil, contributing to an increase in organic matter content of the surface soil. In these situations, mineralizable organic P may be an important source of P to crops (Table 2) and subsequently runoff. In tropical soils, more frequent soil wetting and drying cycles and greater amounts of soil organic P, increase potential mineralization of organic P compared to temperate region soils (Table 2). Considering maximum organic P mineralization rate, crop uptake of P and runoff potential frequently occur at about the same time of year (i.e., spring) which emphasizes the need to quantify the contribution of organic P to P transfer within terrestrial ecosystems.

Several studies have shown that soil P supply, represented by crop yields, were more closely estimated by including extractable organic P (Abbott, 1978; Adepetu and Corey, 1976; Bowman and Cole, 1978; Daughtrey et al., 1973). Bowman and Cole (1978) used a modification of the Olsen P test (16 hr instead of 30-min extraction), and measured the total amount of P (inorganic plus organic) extracted by the reagent. Such modified soil P tests may be conducted on soils identified as having a potentially large contribution of organic P mineralization to the plant available P pool. Caution must be exercised, however, in relating amounts of organic P extracted to crop response in the field, as the conditions of P extraction may not duplicate conditions for organic P mineralization in the field and sorption of mineralized organic P could affect its availability. Even so, soil test methods that estimate or give credit for mineralizable organic P may avoid potentially excessive fertilizer P applications.

3. Water Management

It is apparent from the above discussion that conservation tillage can reduce soil and P transfer in surface runoff, although the proportion that is bioavailable both in soluble and particulate forms may increase. Is this increase in bioavailability sufficient to increase the short- and long-term biological productivity of receiving water bodies? In all examples given, SP and TP concentrations of runoff were consistently above the critical values associated with accelerated eutrophication of a water body (0.01 and 0.02 mg L^{-1}, respectively; Sawyer, 1947; Vollenwieder and Kerekes, 1980). This is true even for unfertilized native grass watersheds, where native soil fertility is apparently high enough in some cases to enrich the SP concentration of runoff. As a result, sustainable or conservation tillage practices may not lower P concentrations of runoff from cultivated or grazed land to critical values. Consequently, the critical P level approach should not be used as the sole criterion in quantifying permissible tolerance levels of P in surface runoff as a result of differing management practices.

Furthermore, as conservation tillage systems reduce P loss in surface runoff, but not its bioavailability, should eutrophication-agricultural management decisions be based on total loss or bioavailability? Several studies have indicated little decrease in lake productivity with reduced P inputs and have attributed this to an increased bioavailability of P entering lakes, as well as internal recycling of P (Gray and Kirkland, 1986; Logan, 1982; Young and DePinto, 1982). Consequently, the measurement of P bioavailability, as both SP and BPP, is essential to more accurately estimate the impact of agricultural management practices on the biological productivity of surface waters.

A detailed procedure to estimate the amounts and form of P in water was presented earlier (Figure 13). However, does the complexity of this scheme limit its use, particularly in the estimation of bioavailable P as recommended? In many situations, the answer is often yes, especially in developing countries where limited resources restrict even simple analytical capabilities. Consequently, there is a need for methodology that will facilitate P estimation. One such method may be the use of Fe_2O_3 strips, which have been developed (Menon et al., 1989a, b) and successfully applied to the estimation of plant available P in a wide range of soils and cropping situations (Menon et al., 1989c; Sharpley, 1991). The strips are made by soaking filter paper in a 10% $FeCl_3$ solution, conversion to $Fe(OH)_2$ by ammonia vapor, and cutting the paper into 10 × 2 cm strips. One strip is shaken with soil (1 g soil in 40 mL 0.01 *M* $CaCl_2$) for 16 h and P removed from the strip by 40 mL 0.1 *M* H_2SO_4.

It is proposed that one Fe_2O_3 strip be shaken with 50 mL of unfiltered runoff with the amount of P removed from the strip representing potentially bioavailable P (subsequently referred to as strip BAP). The strip BAP content of runoff sediment from several Southern Plains watersheds, is related to the growth of P-starved *Euglena* and *Selenastum* incubated for 29 days with runoff sediment as the sole source of P (Figure 25). This relationship is similar to that shown earlier for the frequently used NaOH extractable BAP (Figure 12). In fact, the BAP concentration of runoff from the Southern Plains watersheds, determined by Fe_2O_3 strip and NaOH methods were not significantly different (at the 0.1% level, Figure 26).

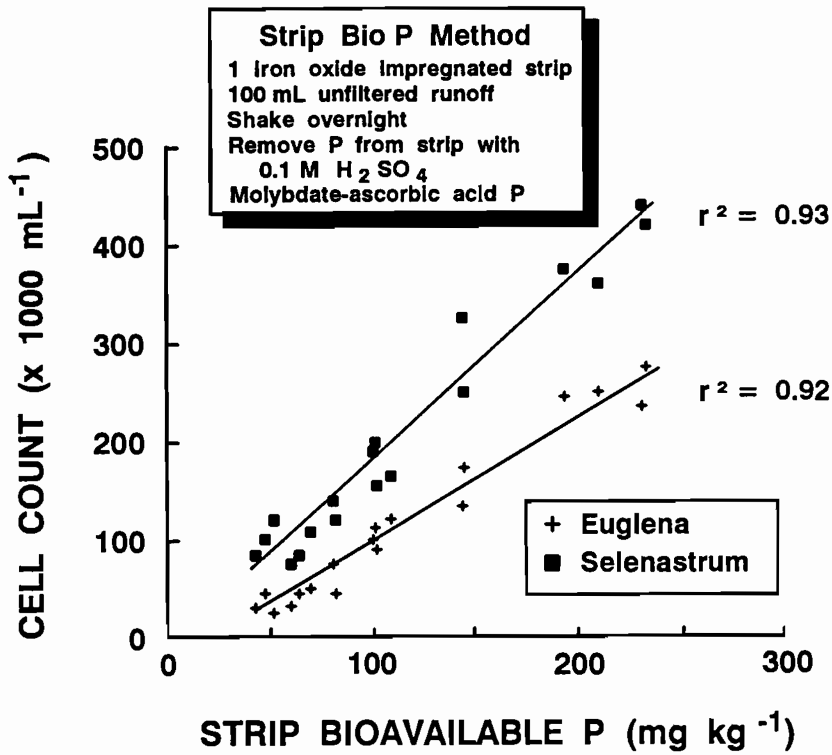

Figure 25. Relationship between the bioavailable P content of runoff sediment, extraction by iron oxide-impregnated paper strip, and growth of P-starved algae during a 29-day incubation.

It is suggested that the Fe_2O_3 strips act as a P-sink and thereby more closely simulate P removal from sediment-water systems by algae. In addition, the validity of relating the form or availability of P extracted by chemicals differing by more than 1 pH unit from runoff or sediment, to *in-situ* bioavailability may be questioned. In as much, the strip method has a stronger theoretical justification for its use over chemical extractants in estimating BAP. In addition, prepared paper strips may be sent to a field location of limited resources and strip BAP measured using only a 100 to 500 mL bottle in which a strip and unfiltered runoff sample is shaken overnight. The strip may then be dried and returned to an analytical laboratory for P removal and measurement. This would also avoid potential problems with P transformations during sample storage and shipping (U.S. EPA, 1979). Future strip development with the use of prepackaged color reagents may allow P determination in the field by comparison with a color chart, similar to measurement of pH with litmus paper.

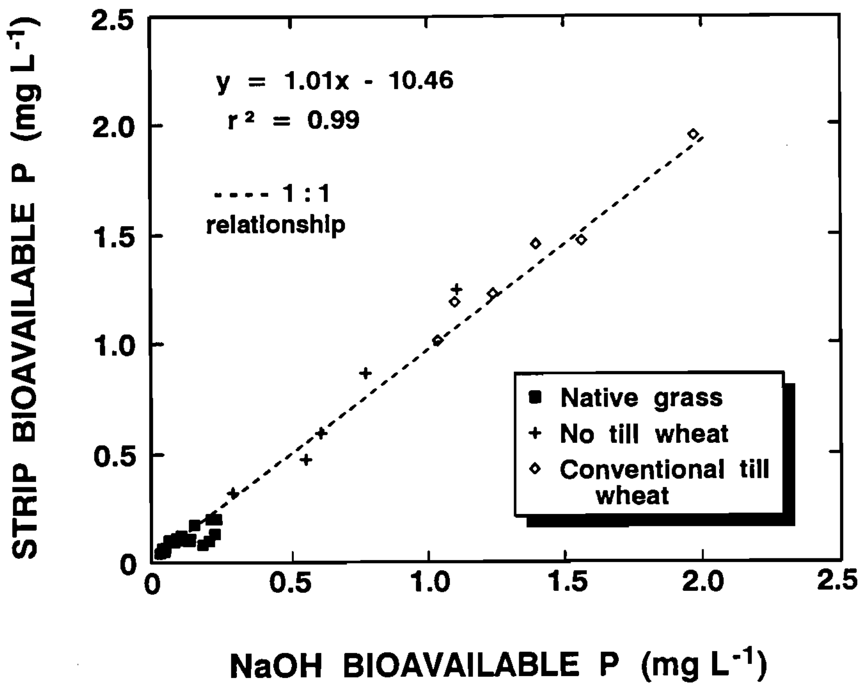

Figure 26. Relationship between the bioavailable P concentration of runoff from the Oklahoma and Texas watersheds, as determined by NaOH and iron oxide-impregnated paper strips.

VIII. Research Needs

The proper management of soil P fertility is essential in maintaining crop yields for an ever increasing world population and to keep unit cost of production low. While its use has been associated with accelerated eutrophication of surface waters, judicious P amendments can reduce P enrichment of agricultural runoff via increased crop uptake and vegetative cover. Yet it is clear that natural soil fertility and P release from plant residues, can result in SP concentrations of runoff from unfertilized pristine land, great enough to stimulate increased biological growth of surface waters. Nevertheless, it is of vital importance that we implement management practices that minimize surface P accumulations, utilize alternative P sources and residual soil P levels, and improve methods estimating P bioavailability in soil and runoff, to affect a decrease in agricultural P inputs to surface waters. Otherwise, the perception by the public that agriculture cannot manage itself for the good of the environment will increase. This may lead to legislation, such as that limiting sewage sludge application to soil, based on soil test P levels to regulate P fertilizer and manure use. Unfortunately, the benefit from implementation of control measures on water quality improvement, will not be immediately visible to a concerned public. Consequently, future research and policy should emphasize the long-term economic and environmental benefits of these measures.

From this discussion, several general areas of future research are suggested to minimize agricultural P loss to surface waters.

Systems Research:

Future research regarding residual fertilizer P in soils needs to consider how a farmer might apply high enough rates of P fertilizer to eliminate P as a deficient nutrient in crop production systems while maintaining an economically sustainable and environmentally sound system in the short-term. Other residual P factors that need to be considered include evaluating one-time, high rate applications of P fertilizer as one method of managing P needs of crops grown with reduced- and no-till systems for several years. On soils that already have excessively high soil P levels, reliable methods are needed to determine how long these soils can supply adequate levels of available P for optimum crop yields before P fertilization beyond low starter rates is required. In addition, we must evaluate the effectiveness/efficiency of residual P availability for crop production versus recently applied (< 2 months) P fertilizer and the need for higher than normal P rates on more soil types, especially when high yield, crop management practices are being used.

Information is also needed on the long-term effects of conservation and low-input systems on the dynamics of P cycling, in terms of the buildup of residual soil levels, fertilizer use, and transfer of bioavailable forms to runoff. This will involve the development of methods to credit residual and organic P effectiveness when considering P fertilization needs, economics, and P fertilizer-use efficiency. In addition, a better understanding of the effect of subsurface placement or injection of P fertilizers and manures under various agricultural systems on soil productivity and surface water quality is warranted. We must also develop more accurate methods of metering and applying manure.

Interdisciplinary Research:

More emphasis should be placed on interdisciplinary research, crossing agricultural and limnological boundaries. Innovative methods must be developed to eliminate P as a deficient nutrient for crop production, which give credit for residual fertilizer and organic P effects, while maintaining short-term economic sustainability and low labor-machinery requirements. Considerable research has been conducted to quantify soil and fertilizer P losses in runoff as a function of management practice. Yet it is still difficult to relate P inputs to lakes to a quantitative description of water quality. While the effect of P concentration on algal growth receives continued attention, little information is available on how lake macrophytes are affected, even though macrophytes present a more serious economic problem than algae in many lakes.

Modeling Research:

Computer simulation of soil productivity and impact of agricultural practices on water quality have improved dramatically over the last decade and are expected to have a major positive impact on farmers and agricultural-eutrophication management decisions. Research is needed on the development of mathematical models simulating changes in residual soil and organic P availability with time. Although first order desorption kinetics have explained soil P release over short-time periods, they have not adequately described long-term changes. As the emphasis on input efficiency increases, accurate predictions of changes in P availability will be required to determine the sustainability and potential transport of bioavailable P in runoff from systems having minimal external P inputs. Considering the need to fertilize the soil rather than the field, models or expert systems should be formulated to variably apply P fertilizer by soil type and need, rather than on a field basis. Credit should be given for previously applied (residual) P fertilizer, such as for example, a budget-type systems approach.

Future research should also be directed towards improving partitioning of soluble, particulate, and especially bioavailable P transported in runoff and their dynamics in lakes. This should focus on the mechanisms of exchange between sediment and solution P and standardization of routine methods to quantify bioavailable P. With the accumulation of fertilizer and residual P at the soil surface under conservation tillage practices, the relative importance of the partitioning processes outlined above may need to be reevaluated. In particular, more accurate simulations of P release from residual soil P under long-term cropping are needed. With the move to more efficient systems, models will enable evaluation of P transport in runoff from soils with high residual P levels with minimal external P inputs.

Although many models are available, it is difficult to select the most appropriate model to obtain the level of detailed information required. Once the appropriate model is chosen, a major limitation is often the lack of input data to drive the model. This most frequently limits model use and the output will only be as reliable as the data input. Consequently, use of these models to provide quantitative estimates of P loss under specific environmental conditions is limited, and the use of many models is recommended to be restricted to the relative comparison of the impact of management options. Because of these limitations more research should be directed to development of a soil index, to identify soil and management practices that may increase the bioavailable P content of surface waters. Agronomic and economic studies have shown that measures to control nonpoint source pollution are much more effective if concentrated on specific source areas rather than on a general basis over large areas (Heatwole et al., 1987; Prato and Wu, 1991). Consequently, index procedures that will specify source areas vulnerable to P loss in runoff, will aid implementation of nonpoint source abatement programs.

It is hoped that answers to these questions will result in agricultural management practices that are both economically and environmentally viable, efficiently utilizing fertilizer and on-farm waste materials, and thereby improving water quality for the next century.

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