

Processes Governing Phosphorus Availability in Temperate Soils

E. Frossard,* L. M. Condron, A. Oberson, S. Sinaj, and J. C. Fardeau

ABSTRACT

Phosphorus losses from agricultural soil to water bodies are mainly related to the excessive accumulation of available P in soil as a result of long-term inputs of fertilizer P. Since P is a nonrenewable resource, there is a need to develop agricultural systems based on maximum P use efficiency with minimal adverse environmental impacts. This requires detailed understanding of the processes that govern the availability of P in soil, and this paper reviews recent advances in this field. The first part of the review is dedicated to the understanding of processes governing inorganic P release from the solid phase to the soil solution and its measurement using two dynamic approaches: isotope exchange kinetics and desorption of inorganic P with an infinite sink. The second part deals with biologically driven processes. Improved understanding of the abiotic and biotic processes involved in P cycling and availability will be useful in the development of effective strategies to reduce P losses from agricultural soils, which will include matching crop needs with soil P release and the development of appropriate remediation techniques to reduce P availability in high P status soils.

SOILS contain between 100 and 3000 mg P kg⁻¹ soil, most of which is present as orthophosphate compounds. The proportion of total soil P present in organic forms ranges from 30 to 65% (Harrison, 1987). The soil solution in agricultural soils, which is the main source of P for plant roots, contains between 0.01 and 3.0 mg P L⁻¹. The quantity of P present in the soil solution represents only a small fraction of plant needs, and the remainder must be obtained from the solid phase by a combination of abiotic and biotic processes. The pro-

cesses involved in soil P transformation are precipitation–dissolution and adsorption–desorption which control the abiotic transfer of P between the solid phase and soil solution, and biological immobilization–mineralization processes that control the transformations of P between inorganic and organic forms (Fig. 1).

Phosphorus losses from soils occur by leaching at very low rates in undisturbed ecosystems (Walker and Syers, 1976; St. Arnaud et al., 1988; Frossard et al., 1989; Letkeman et al., 1996). The implementation of intensive agricultural production has markedly increased P losses from soils through increased runoff, erosion and leaching, which in turn can have adverse effects on water quality. These losses are further increased by the excessive accumulation of bioavailable P in the upper soil horizons, due either to application of inorganic and/or organic P fertilizers in excess of plant needs and/or to inappropriate fertilizer applications (Braun et al., 1994; Beaton et al., 1995; Sharpley et al., 1995; Sharpley and Rekolainen, 1997; Sibbesen and Runge-Metzger, 1995; Sibbesen and Sharpley, 1997; Daniel et al., 1998; van der Molen et al., 1998).

It is the hypothesis of this paper that an efficient way of reducing P losses to the environment while maintaining an optimum plant production is to combine limited but sufficient P inputs with maximum acquisition and utilization of soil and fertilizer P by plants. Whereas traditional P fertilization strategies were designed to increase and maintain plant-available P in soil at levels required for maximum production (i.e., non-P limiting), it is now necessary to use soil and fertilizer P resources more efficiently. Accordingly, there is a need to quantify not only the level of rapidly plant-available soil P but also the amount of P, which can be released by desorp-

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Abbreviations: DMT HFO, Pi extraction with a dialysis membrane tube filled with anhydrous ferric oxide; E1min, Pi isotopically exchangeable within 1 minute; E1min–12wk, Pi isotopically exchangeable between 1 minute and 12 wk; E > 12 wk, Pi that cannot be exchanged within 12 wk; NMR, nuclear magnetic resonance.

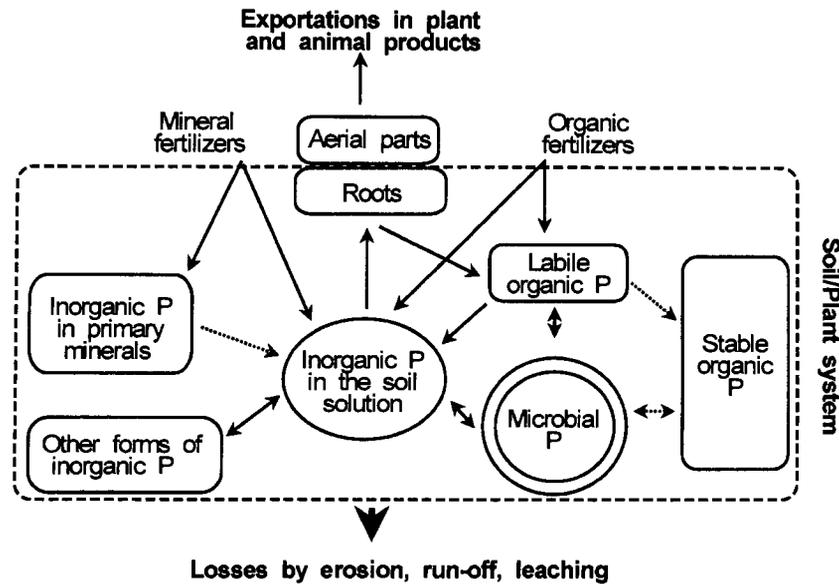


Fig. 1. The soil P cycle (adapted from Stewart and McKercher, 1982).

tion over the season. In addition, it is also necessary to assess the influence of biological components (including microbial biomass, mycorrhiza, earthworms) on the acquisition, storage, and release of P to plants. Finally, these various processes must be integrated to assist in the development of effective strategies to reduce risks associated with P loss from high fertility soils.

Several review papers have been written on various aspects of soil P cycling (e.g., Stewart and Tiessen, 1987; Syers and Curtin, 1989; Frossard et al., 1995; Magid et al., 1996; Gressel and McColl, 1997). The aim of this paper is to review the more recent advances that have been made in understanding the abiotic and biotic processes that determine P availability in temperate soils. The extent to which these processes can be integrated at different levels of the soil-plant system to improve the efficiency of P utilization will also be considered, together with topics for future research.

Abiotic Processes Controlling Inorganic Phosphorus Release to the Soil Solution

The mechanisms underlying the short-term and long-term sorption of inorganic P (Pi) on pure minerals (carbonates and iron/aluminum oxides) and on soils are well documented (Parfitt, 1978; Barrow, 1983; Freese et al., 1995b; Scheinost and Schwertmann, 1995; Torrent, 1997). Much less attention has been paid to assessing the amount of Pi that can be potentially released from soils and soil minerals, although this is highly relevant, both in terms of plant nutrition and P losses. Recent studies have demonstrated that the amount of soil-available Pi and percentage of Pi saturation are related to concentrations of available Pi measured in runoff (Sharpley, 1995; Pote et al., 1996; Sibbesen and Sharpley, 1997) and leachates (Brookes et al., 1997).

Different methods can be used to assess the release of Pi to the soil solution. These include extraction with water (van der Pauw, 1971), dilute solutions of chemicals (Olsen et al., 1954; Cottenie et al., 1982; Ruiz et al., 1997), anion exchange resin (Sibbesen, 1978), isotopic

exchange (Fardeau, 1996), and using an infinite sink such as the Fe oxide-impregnated paper (van der Zee et al., 1987).

Chemical extractants are of great interest because they are rapid and can be used for routine analysis. They must be used with caution because they dissolve variable proportions and forms of available and unavailable soil P (Fardeau et al., 1988; Kato et al., 1995; Tunney et al., 1997). The amount of P extracted by anionic resins is well correlated to the P uptake by plants (Sibbesen, 1978; Tran et al., 1992). However, this method has also some drawbacks: (i) resins do not sorb-desorb specifically phosphate ions, (ii) in highly P fertile soils they are not able to act as an infinite sink for Pi, and (iii) depending on the counter-ion used (Cl^- , F^- , OH^- , CO_3^{2-}), resins might significantly alter the pH of the soil solution (Barrow and Shaw, 1977; Freese et al., 1995a). Therefore, the following sections will focus on two approaches:

1. Isotopic exchange, which measures the rate of Pi transfer between the solid phase and soil solution under steady state conditions

2. The infinite sink method, which measures the amount of Pi located on the solid phase which can migrate to the solution by maintaining a very low concentration of Pi in the soil solution

Isotope Exchange Kinetics

Since the use of isotope techniques to investigate P cycling in soils has been recently reviewed (Fardeau, 1996; Di et al., 1997; Frossard and Sinaj, 1997), this section only focuses on the basic principles of the isotope exchange kinetic approach and highlights some recent results.

Numerous studies have shown that the main source of P for most agricultural plants, in the presence or absence of mycorrhiza, is isotopically exchangeable Pi (Larsen, 1967; Fardeau and Jappé, 1976; Bolan et al., 1984; Morel and Plenchette, 1994; Frossard et al., 1994).

Fardeau et al. (1985) showed that studying the kinetics of isotopic exchange during a short experiment (100 min) can be used to derive parameters for a model to predict the amount of Pi that will be exchangeable, and therefore available, over at least 3 mo (Fardeau et al., 1985; Frossard et al., 1994). This approach suggests that it is not possible to consider soil Pi as existing in discrete pools containing *available* and *unavailable* forms of Pi. In contrast, the kinetic approach suggests that there is a continuum between forms of Pi that are immediately available, such as those present in the soil solution, and forms that are present on the solid phase and that are very slowly exchangeable, including adsorbed and precipitated forms of Pi. This conclusion is consistent with previous studies on the kinetics of Pi sorption and desorption (Barrow, 1983; Lookman et al., 1995).

The isotope exchange kinetic approach takes into account the three parameters, which, according to Beckett and White (1964), are necessary to characterize soil Pi availability (i.e., intensity, quantity, and capacity factors) (Salcedo et al., 1991; Morel et al., 1994; Fardeau, 1996; Frossard and Sinaj, 1997). The intensity factor is provided by the concentration of Pi in the soil solution (Sinaj et al., 1998), the quantity factor is provided by the amount of isotopically exchangeable Pi [$E(t)$], and information on the capacity factor is provided by the rate of disappearance of radioactive Pi from the soil solution. Results from long-term field experiments show that isotopic exchange kinetics can accurately describe the response of crops to P fertilization in a wide range of soils, including Chernozems, Luvisols, Ultisols, and Oxisols (Oberson et al., 1993; Morel et al., 1994; Frossard and Sinaj, 1997; Oberson et al., 1999).

The isotope exchange technique has been used to assess Pi availability in large-scale studies in northeast France on sedimentary soils (Frossard et al., 1992) and in Albania on a wide range of soils developed mainly on sedimentary and metamorphic materials (Sinaj et al., 1992). Results from both studies demonstrated that the Pi buffering capacity of the soils, as determined by isotope exchange kinetics, was strongly related to the Fe and Al oxide content, and therefore could be predicted from soil parent material (Frossard et al., 1992, 1993; Sinaj et al., 1992). On the other hand, the quantity of isotopically exchangeable Pi measured during 1 min (E_{1min}), which is considered to be totally and immediately plant available, is influenced by a combination of soil parameters and management.

Recently, Sinaj et al. (1997) showed that a large proportion of slowly exchangeable Pi becomes more rapidly exchangeable upon total soil dispersion (Table 1). Their results suggest that up to 30% of total soil Pi may be protected within water stable aggregates. This illustrates the importance of aggregation in determining the rate of release of Pi from the solid phase of soil particles, which can be organized in aggregates of various sizes, to the solution where roots take up Pi (Linguist et al., 1997).

Infinite Sink Method

The principle of this approach is to displace the Pi located on the solid phase of the soil by lowering the

Table 1. Comparison of isotopically exchangeable Pi measured on unfractionated soil samples sieved at 2 mm (UF) and calculated for the same samples after dispersion with a Na-resin (D) (adapted from Sinaj et al., 1997).

Soil	E1min†		E1min–12wk‡		E > 12wk§	
	UF	D	UF	D	UF	D
	mg kg ⁻¹					
Uncultivated Calciustoll	9.7	25.7	84.8	180.2	298	183
Cultivated Calciustoll	3.6	5.1	8.9	22.4	227	196
Uncultivated Xeropsamment	5.9	3.6	5.9	25.6	277	240
Cultivated Xeropsamment	7.1	5.6	7.3	12.6	135	111
Uncultivated Haplosalid	2.8	3.5	77.7	40.9	225	247
Cultivated Haplosalid	6.8	3.6	9.6	34.0	297	261
Uncultivated Haplustert	7.2	5.9	10.9	33.2	122	88

† E1min: Pi isotopically exchangeable within 1 min.

‡ E1min–12wk: Pi isotopically exchangeable between 1 min and 12 wk.

§ E > 12wk: Pi that can not be exchanged within 12 wk.

concentration in the solution to a very low level, with minimal alteration of the system. This is done by shaking a soil water suspension with an Fe oxide-impregnated paper, which acts as a sink for Pi (van der Zee et al., 1987). The amount of P extracted by this approach is closely related to the amount of soil Pi available to plants and the amount of sediment Pi available to algae (Chardon et al., 1996; Menon et al., 1997; Myers et al., 1997).

The major source of error encountered using the Fe oxide impregnated paper is the adhesion of small soil particles onto the paper, which can lead to a large overestimation of P availability (Perrott and Wise, 1993; Freese et al., 1995a; Myers et al., 1997). Furthermore, until now this method has been mostly used to measure the amount of Pi, which could be extracted from the soil after a given time (usually 16 h), and therefore it gives only an indication of the quantity factor. The few kinetics studies carried out have been mostly used to establish an appropriate extraction time, and results obtained in these studies were not evaluated in agronomic and environmental terms (van der Zee et al., 1987; Chardon et al., 1996; Myers et al., 1997).

Freese et al. (1995a) recently improved the paper method by using a dialysis membrane tube filled with a hydrous ferric oxide (DMT-HFO) as an infinite sink instead of Fe oxide-saturated paper. The use of the DMT-HFO technique slows down desorption of Pi from the soil, but prevents the adhesion of soil particles. In a further study, Lookman et al. (1995) used this technique to study the desorption kinetics of Pi from acidic sandy soils for 9.5 wk. Their results showed that Pi desorption could be described by a sum of exponential terms, and that even after 9.5 wk of desorption, significant amounts of Pi were still being desorbed (Fig. 2). These results suggest that most of the soil Pi could be desorbed over a sufficiently long period. Using the DMT-HFO technique, Lookman et al. (1995) also showed that the amount of rapidly desorbable Pi was closely related to the quantity of amorphous Fe and Al oxides and to their degree of saturation with Pi.

Lookman et al. (1997) used solid state ³¹P NMR to examine excessively fertilized acid sandy soils and observed the disappearance of a group of protonated ³¹P nuclei (probably Ca-P) as a result of extraction with DMT-HFO (Fig. 3). Although they could not definitely

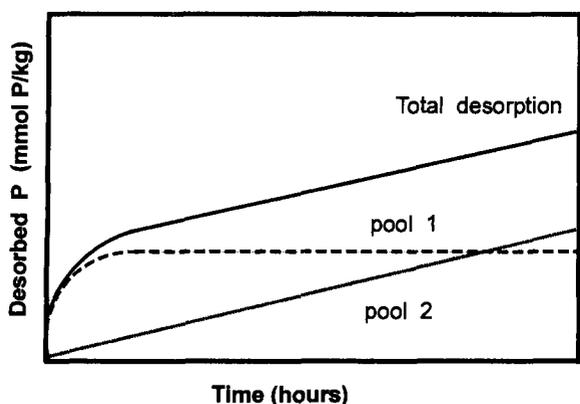


Fig. 2. Fitted curves for desorption of Pi from an excessively fertilized acid sandy soil using the HFO-DMT (hydrous ferric oxide-dialysis membrane tube) technique (adapted from Lookman et al., 1995). The curve *total desorption* shows the total amount of Pi desorbed from the soil. The curve *pool 1* denotes the amount of Pi that can be desorbed from the pool containing rapidly desorbable Pi. The curve *pool 2* shows the amount of Pi that can be desorbed from the pool containing slowly desorbable Pi.

assign their results to desorption or to the displacement of small particles, they showed on another sample that a sixfold water extraction led to a marked decrease in both Ca-P and Al-P, and that no more Pi could be detected in the soil after oxalate extraction. The results obtained with the DMT-HFO technique and solid state NMR physically demonstrated the existence of several pools of Pi in soils: a pool containing Pi that is rapidly desorbed, a pool of slowly exchangeable Pi, and a pool of Pi precipitated in condensed Ca phosphates (e.g., apatite) and/or strongly sorbed onto Al oxides.

Both the isotope exchange kinetics and infinite sink approaches allow precise characterization of the rate of Pi transfer between the solid phase and the soil solution.

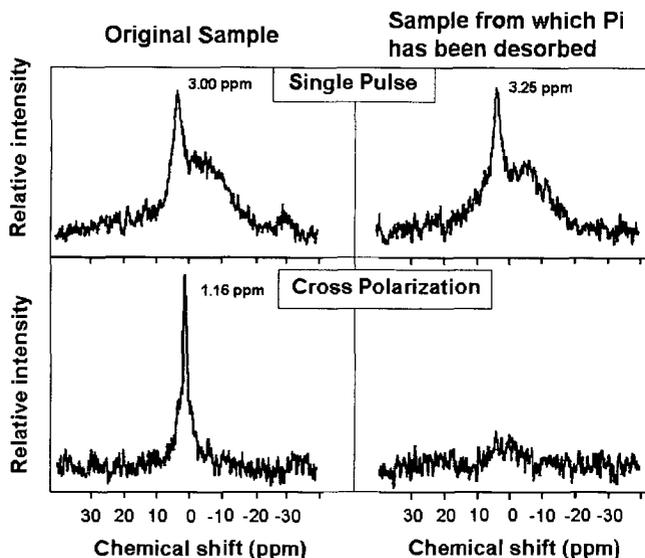


Fig. 3. Solid-state ^{31}P MAS NMR (SP, single pulse; CP, cross polarization) of a particle-size fraction ($<50\ \mu\text{m}$) of an excessively fertilized acid sandy soil before and after desorption of labile Pi with the HFO-DMT (hydrous ferric oxide-dialysis membrane tube) technique (adapted from Lookman et al., 1997). The SP NMR experiment allows to observe all the ^{31}P nuclei in the soil, while the CP NMR experiment shows selectively the ^{31}P nuclei located within a few angstroms from ^1H nuclei.

Both approaches confirm the existence of a continuum of mobility between the Pi found in soil solution and Pi sorbed onto the solid phase, and that it is arbitrary to distinguish *available* and *unavailable* Pi fractions. Furthermore, these approaches allow measurement of the amount of Pi that can be released over the long term (e.g., during the time of intense uptake of nutrient by a crop). A comparison between these two approaches will enable the description of Pi fluxes between the soil's solid phase, soil solution, and plant roots during intense periods of Pi uptake. This information will have to be included in the development of soil fertility management schemes designed to optimize utilization of Pi inputs. However, to achieve this objective, the importance of soil micro- and macrostructure will have to be taken into account, and these Pi fluxes will have to be integrated with parameters that can be easily measured (e.g., soil tests, P inputs, qualitative and quantitative information on soil type).

Biotic Processes Controlling Phosphorus Release to the Soil Solution

The two main biotic processes that influence Pi release into soil solution are P turnover through the microbial biomass and organic P mineralization. Despite extensive investigation, our understanding of organic P dynamics in soil and the precise role of the soil microbial biomass is limited (Stewart and Tiessen, 1987; Magid et al., 1996). Nonetheless, in recent years significant advances have been made in the development and application of appropriate techniques to enable detailed examination of organic P components and biotic P dynamics in the soil environment.

A great deal of work has been focused on the separation and identification of different chemical forms of organic P in soil using detailed chemical fractionation and ^{31}P NMR spectroscopy. The application of both techniques to study soil organic P forms and dynamics has been recently reviewed (Sanyal and DeDatta, 1992; Magid et al., 1996; Preston, 1996; Condon et al., 1997).

Recent studies have confirmed that chemical fractionation of soil organic P based mainly on solubility in different alkali reagents (principally sodium bicarbonate [NaHCO_3] and sodium hydroxide [NaOH]) is most useful in the examination of net long-term changes in organic P forms that occur in response to alterations in land use and/or soil management (e.g., Rubæk and Sibbesen, 1995; Condon et al., 1996; Oberson et al., 1996; Schmidt et al., 1996). Thus, in soils under 17 yr of continuous cropping in the USA, Schmidt et al. (1996) showed that while NaOH-extractable organic P increased in response to inputs of P fertilizer, organic P was released from this pool when P fertilizer addition was discontinued. Condon et al. (1996) found that concentrations of total organic P in topsoil (0–10 cm) decreased markedly (15–38%) in the 10 to 31 yr following the establishment of conifers in grassland soils in New Zealand. Most of the observed decreases in soil organic P under conifers occurred in the main NaOH-extractable pool, with smaller, less consistent decreases in the NaHCO_3 organic P pool. Oberson et al. (1996) found that levels of labile NaHCO_3 and moderately labile

NaOH-extractable organic P were similar in adjacent soils under long-term *organic* (nutrients applied in organic manures) and *conventional* (nutrient applied in mineral and organic fertilizers) farming systems while the soils under *conventional* management consistently showed lower levels of biological activity (ATP content, enzyme activity). These findings reveal the restricted information obtained from extraction-based fractionation. Differences in soils over time show net changes only, and provide little information on the mechanisms responsible for soil organic P dynamics.

In short-term and seasonal studies of P dynamics, there has been some limited success in establishing relationships between soil organic P fractions and P availability. For example, Fabre et al. (1996) demonstrated that NaHCO₃-extractable organic P increased during winter and decreased during summer. This finding is consistent with some earlier studies that indicated NaHCO₃-extractable organic P represented a labile pool of organic P in temperate soils (Magid et al., 1996). It should also be noted that the nature of relationships between soil organic P fractions and exocellular phosphatase enzyme activities in soil remains unclear, and there is a need for continued detailed investigation of the precise role and functions of phosphatase enzymes in soil organic P dynamics (Magid et al., 1996; Gressel and McColl, 1997).

The validity of assigning relative availability to different chemical forms of soil organic P as defined by ease of extraction with reagents such as NaHCO₃, especially over the short term, is uncertain and the need to define appropriate *biologically meaningful* pools of organic P in soil has been highlighted (Magid et al., 1996; Gressel and McColl, 1997). To this end, several recent studies have been concerned with the development of extraction techniques designed to better identify the labile pool of organic P in soil. For example, in a study conducted on an established field trial in Denmark, Rubæk and Sibbesen (1995) showed that organic P extracted from soil by a macroporous anion exchange resin (Lewatit MP500a) decreased in spring and was highest in summer, whereas NaHCO₃-extractable organic P showed no distinct seasonal pattern.

In a related study, Guggenberger et al. (1996) examined the effects of land use (cropping, forestry, pasture) and P fertilizer inputs on Lewatit MP500a resin-extractable organic P in soils from Germany and Denmark. They demonstrated that there was a close relationship between the resin-extractable organic P and labile diester forms of organic P determined in NaOH extracts using ³¹P NMR (Fig. 4). This, together with the consistent C/organic P ratio of organic matter extracted from soil by the resin, indicated that the resin-extractable organic P represented a *potentially labile* pool, which they found increased under pasture and declined under arable cropping.

The findings presented by Guggenberger et al. (1996) suggest close links between organic C and organic P dynamics in soil. These links were demonstrated by Gressel et al. (1996) in litter-soil under a mixed conifer stand in California using a combination of solid-state ¹³C NMR and liquid-state ³¹P NMR. In a recent review,

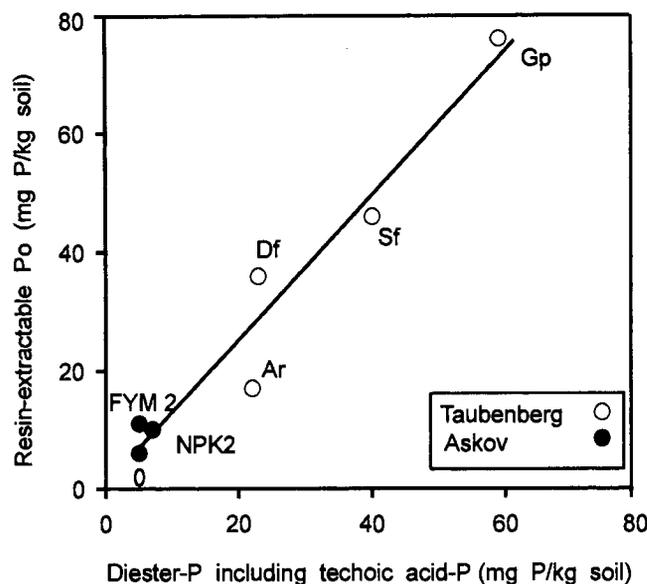


Fig. 4. Relationship between organic P extracted by macroporous resin and diester organic P (including teichoic acid P) determined by ³¹P NMR in alkali extracts from selected German and Danish soils (adapted from Guggenberger et al., 1996).

Gressel and McColl (1997) concluded that continued advances in our understanding of organic P mineralization, particularly in forest soils and litter, will require an integrated approach involving consideration of C and organic P biochemistry and cycling.

The nature and dynamics of water-soluble organic P and labile forms of soil organic P in the rhizosphere are undoubtedly linked to the activities of microflora and exocellular phosphatase enzymes (Joner et al., 1995; Gahoonia and Nielsen, 1995; Firsching and Claassen, 1996). Shand et al. (1994) showed that organic P exceeded Pi by 5 to 20 times in the soil solution from P deficient Cambisols. Pant et al. (1994) obtained soil solutions from various cropped soils and found that organic P accounted for up to 50% of the total P. They also showed that 5 to 70% of the solution organic P was hydrolyzed when incubated with phosphomonoesterase enzymes (phytase, acid, or alkaline phosphatase). Other studies have shown that plants grown in nutrient solutions can readily utilize water-soluble forms of organic P after hydrolysis (e.g., Macklon et al., 1997). The continued development and refinement of appropriate techniques that enable detailed examination of rhizosphere processes (e.g., Zoysa et al., 1997; Gaume et al., 1998) will greatly assist in improving our understanding of organic P cycling and availability in soil.

The turnover or flux of P through the soil microbial biomass is widely acknowledged to be an important factor that determines the availability and utilization of organic P (and Pi) in the soil-plant system (Stewart and Tiessen, 1987; Magid et al., 1996). Several recent studies have highlighted the importance of the microbial biomass in P cycling in temperate soils. Oberson et al. (1996) examined microbial P transformations in soil under contrasting farming systems in Switzerland. They concluded that both levels of microbial P (Fig. 5) and microbial P fluxes were greater under organic farming systems (based on inputs of organic manure) compared

with a conventional farming system (based on inputs of mineral and organic fertilizers), and that they were important in relation to P uptake by plants. He et al. (1997) examined season changes in microbial biomass P in pasture soils in a long-term trial in the UK. They found that microbial P levels and P concentration in the biomass as measured from its C/P ratio were related to P inputs (mineral fertilizer or organic manure) and soil moisture. They calculated average microbial P contents (0–15 cm) of 17 to 290 kg P ha⁻¹, and estimated P fluxes through the soil microbial biomass of 11 to 190 kg P ha⁻¹ yr⁻¹, compared with P removed in herbage of only 2 to 11 kg P ha⁻¹ yr⁻¹. In both studies, the P flux through the microbial biomass was deduced from temporal fluctuations in the microbial P content.

Brookes et al. (1984) calculated an annual P flux through the microbial biomass by dividing microbial P determined at one date by a turnover time of 2.5 yr. This turnover time had been calculated for biomass C by fitting a turnover model to data from field measurements made at Rothamsted (Jenkinson and Ladd, 1981). Both approaches represent estimates of the microbial P flux only, and direct measurement of flux has not been possible thus far. McLaughlin et al. (1988) successfully followed the incorporation of ³³P-labeled plant tissue into soil microorganisms. Of the ³³P applied, 22 to 28% was recovered in the microbial biomass. This incorporation occurred within a short time after the addition of the labeled residues. However, most of the P taken up by the microbial biomass was derived from the soil.

Recent improvements in methodology to determine microbial P in soil (Morel et al., 1996) mean that there is scope for improved understanding of the role of the microbial biomass in organic P cycling and availability, especially when integrated with studies of Pi dynamics and P uptake by plants. However, methods used so far to determine the amount of microbial P are based on chloroform fumigation and assess only a fraction of the

total soil microbial P. The latter is calculated from chloroform released P using a correction factor, k_p (Brookes et al., 1982; Hedley and Stewart, 1982; McLaughlin et al., 1986). This factor can be determined by adding microorganisms to soil, either with known amounts of P (Brookes et al., 1982) or additionally ³³P labeled (Hedley and Stewart, 1982). The k_p factor should be determined for each soil, as microflora differ from soil to soil (McLaughlin et al., 1986). As added cultured populations of microorganisms are unlikely to represent the microbial population of a given soil, and added microorganisms are not physically protected by the soil, it might never be possible to measure the actual size of microbially bound P, and it may be preferable to refrain from using the k_p factor at all.

It is recognized that the microbial biomass may take up P as a result of the addition of organic matter, or in response to changes in physiology caused by moisture fluctuations or other disturbances (McLaughlin et al., 1988; Tiessen et al., 1994b). The process of basal N mineralization using ¹⁵N (Mary and Recous, 1994) has not yet been applied to P. Such basal mineralization processes may occur at constant microbial P compartment sizes. However, since the approaches used to assess microbial P turnover mentioned above do not include this process, which could be closely linked to mineralization, this aspect should be explored.

As described previously, radioactive P isotopes can be successfully used to describe and determine the availability of Pi in soil. The principle of isotopic dilution was believed to be of limited value in determining organic P mineralization due to problems associated with continued Pi sorption reactions (Di et al., 1997). However, recent studies have demonstrated that isotope exchange kinetics can be successfully used to determine gross organic P mineralization in soil. In theory, organic P mineralization can be calculated from the difference between isotopically exchangeable Pi measured after a given period of incubation and isotopically exchangeable Pi predicted for the same period from a short-term (100 min) isotope exchange kinetics experiment conducted on the original (nonincubated) soil (Frossard and Sinaj, 1997; López-Hernández et al., 1998). Correct application of this methodology requires that the soil organic P is not labeled and the specific activity of P in solution is decreased by a combination of abiotic isotopic exchange and P released by mineralization (Frossard et al., 1996; López-Hernández et al., 1998). López-Hernández et al. (1998) used this technique to determine organic P mineralization in a number of low P fixing Mollisols from the USA and obtained mineralization rates of 220 to 900 μg P kg⁻¹ d⁻¹. These findings indicate that there is considerable potential for the continued application and refinement of this technique to a wide range of soils.

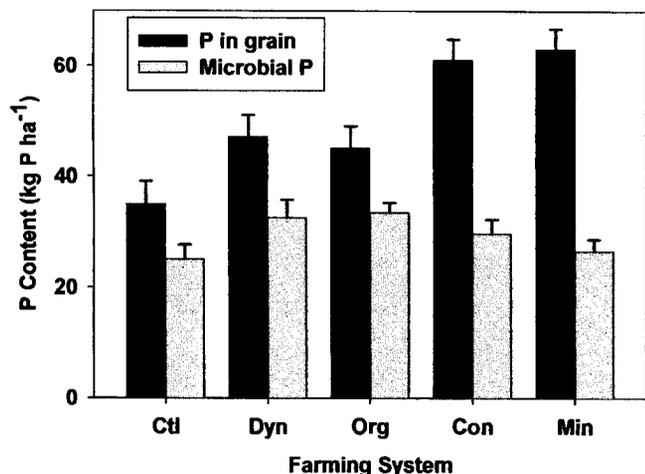


Fig. 5. Phosphorus content in grain and estimated P content of the soil microbial biomass at the time of wheat (*Triticum aestivum* L.) harvest in soils after 14 yr under different fertilization and cultivation regimes (Ctl, unfertilized; Dyn, bio-dynamic [exclusively organic fertilizers]; Org, bio-organic [organic fertilizers]; Con, conventional [mineral and organic fertilizers]; Min, conventional [exclusively mineral fertilizers]) (data from Oberson et al., 1993, 1996).

Integration of Abiotic and Biotic Processes to Minimize Phosphorus Loss from Soil to Water

Besides decreasing the risk of P transport from soil to water bodies by controlling erosion, runoff, and leaching as discussed in other papers in this series, P losses

from agricultural soils may be minimized by a combination of the following.

1. Reducing P availability in excessively fertilized soils by lowering P inputs and either increasing P removal in produce or increasing the P sorption capacity of soil.

Data from long-term field experiments can be used to derive precise relationships between soil-available P (e.g., water-soluble Pi), P inputs, and P removal. These relationships have been shown to be closely related to the P buffering capacity of the soil, which in turn is related to soil mineralogy (clay, carbonates, Fe/Al oxides). Accordingly, it should be possible to use these soil properties in combination with the plant requirements to predict the P input required to achieve optimum levels of water-soluble Pi.

During the last decade, several attempts have been made to decrease soil P availability by applying amendments such as Fe oxides, alum, and coal combustion by-products designed to increase the P sorption capacity of the soil (van der Molen et al., 1998, Shreve et al., 1996; Stout et al., 1998). Improved understanding of the P sorption-desorption properties of different soil amendments would be helpful in predicting decrease in Pi availability. Provided the P sorbing amendment is not an environmental risk and the complexed Pi remains very slowly available, this may be an appropriate strategy to reduce available P in high-fertility soils, especially in combination with efforts to increase P removal in produce.

2. Modify cropping systems to achieve optimum utilization of soil and fertilizer P. Phosphorus is a vital non-renewable resource. In addition to establishing upper limits of soil-available P to minimize P losses in runoff and drainage (Sibbesen and Sharpley, 1997), it is necessary to develop cropping systems that use soil P efficiently. The development of P efficient cropping systems in temperate ecosystems will require the following:

- The overall balance of P in the soil-plant-animal system must be accurately quantified (e.g., Nguyen and Goh, 1992; Yanai, 1992; Zaccheo et al., 1997);
- The long-term release of soil Pi must be understood and monitored;
- Soil biotic processes (micro- and macrofauna, including endomycorrhiza) must be optimized to maintain P, either in the living biomass or in easily degradable forms of soil organic matter (e.g., particulate organic matter);
- Plant breeding and selection to obtain species that are most efficient at extracting soil P, even at low soil P levels;
- Rotational cropping systems that include P-efficient crops should be employed where possible;
- Appropriate measures to protect and maintain soil and water resources should be included (e.g., use of riparian buffer zones between cultivated land and water bodies).

Research Priorities

Our ability to develop and implement effective integrated strategies for the efficient use of soil and fertilizer P is hampered by deficiencies in our understanding of

key processes that control P cycling and availability in soil. To this end, future research in the area of soil P cycling should focus on the following:

- Measurement of rates of abiotic Pi release from solid phase to soil solution, especially over the long term;
- Continued development and refinement of techniques to measure soil organic P mineralization and P turnover through the soil microbial biomass and their temporal relationship to plant needs;
- Establish and quantify relationships between land use and biological soil P transformations (including the role of endomycorrhizae);
- Identify crop plant species with high P acquisition efficiency including at low soil P levels;
- Continued development and evaluation of mechanistic models of soil P cycling that integrate abiotic and biotic processes (e.g., Tiessen et al., 1994a; Grant and Heaney, 1997), including consideration of spatial variability and soil characteristics at field level.

Finally, it should be stressed that effective results can only be obtained if the above research is conducted in an integrated manner since it requires input from a number of specialists.

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