Phosphorus activators contribute to legacy phosphorus availability in agricultural soils: A review

Jing Zhu*, Min Li*, Mick Whelan\textsuperscript{b}

\textsuperscript{a}College of Environmental Science and Engineering, Beijing Forestry University, Beijing, China
\textsuperscript{b}School of Geography, Geology and the Environment, University of Leicester, Leicester, UK

*Min Li
Ph.D., Professor
College of Environmental Science and Engineering
Beijing Forestry University, 60#
35# Qinghua East Rd., Haidian District,
Beijing 100083, P.R.China
Tel: 86 10 62336615
Email: minli@bjfu.edu.cn; liminbjfu@126.com
Abstract

Phosphorus (P) is one of the most limiting macronutrients for crop productivity and P deficiency is a common phenomenon in agricultural soils worldwide. Despite long-term application of phosphate fertilizers to increase crop yields, P availability is often low, due to the high affinity of phosphate for the soil solid phase. It has been suggested that the accumulated (surplus) P in agricultural soils is sufficient to sustain crop yields worldwide for about 100 years. In this paper, we try to clear up the potential for making use of legacy P in soils for crop growth potentially alleviating the global P resource shortage. Specifically, we try to clear up the potential of soil “P activators” for releasing fixed P. P activators accelerate and strengthen process which transform P into bio-available forms via a range of chemical reactions and biological interactions. They include phosphate solubilizing microorganisms, phosphatase enzymes and enzyme activators, low molecular weight organic acids, humic acids, lignin, crop residues, biochar and zeolites. Although reported performance is variable, there is growing evidence that P activators can promote the release of phosphate from soil and, hence, have potential for mitigating the impending global P crisis. Further basic and applied research is required to better understand the mechanisms of interaction of P activators with natural soils and to maximize activator efficacy.

Key words: Agricultural soils; Potential phosphorus pool; Legacy Phosphorus; Phosphorus activation; Phosphorus bioavailability; Activator

1. Introduction

Phosphorus (P) is a major limiting nutrient for crop growth in many agroecosystems and is indispensable in several physiological and biochemical processes (Simpson et al., 2011). Consequently, the availability of P controls the growth and development of all crops (Wyngaard et al., 2016). Plants assimilate P (predominantly as negatively charged primary and secondary orthophosphate ions) from the soil solution. Although most soils contain considerable total P stocks, only a small fraction (<1%) of the total inorganic P (Pi) and organic P (Po) is dissolved at any given time (Bünemann, 2015). The concentration of P in soil solution remains really low, about 0.05-0.30 ug P mL⁻¹ (Bolan, 1991). It is estimated that crop productivity is limited by P deficiency on more than 40% of the world’s arable land (Balemi and Negisho, 2012). P limitations are also a major productivity constraint in many natural (Augustine et al., 2003) and managed (Bünemann et al., 2011) grassland systems. Although other constraints may also limit productivity, soil P often needs to be replenished repeatedly to satisfy plant demand, in part because it is quickly removed from the dissolved (plant available) form by sorption, precipitation and microbial immobilization (Roberts and Johnston, 2015) or is lost through surface runoff (in particulate and dissolved forms), subsurface flow (leaching and throughflow via the soil matrix and macropores), drain flow and even groundwater which may accelerate the eutrophication of P sensitive waters (Heathwaite and Dils, 2000; Hively et al., 2006; Holman et al., 2008; Sims et al., 1998).

Historically, chemical fertilizers, such as manufactured water-soluble phosphate have played a significant role in replenishing this P pool. However, these fertilizers are derived from mined rock-phosphate which has a finite (and rapidly diminishing) supply (Gilbert, 2009). Some studies claim that, at current rates of extraction, global commercial phosphate reserves will be depleted in 50-100 years.
(Abelson, 1999; Sattari et al., 2012). Opinions vary about the reliability of these data because they are principally based on second and third hand information and the figures change all the time (Gilbert, 2009). However, it is likely that remaining reserves will have increasingly lower quality and will be increasingly more costly to extract, which means that the supply of high quality phosphate fertilizer will also become progressively more restricted (Cordell et al., 2009). The rising demand for agricultural commodities in developing countries has put increasing pressure on land resources for higher yields, with associated growth in the demand for phosphate fertilizers (Weber et al., 2014). This realization (sometimes described as a “potential phosphate crisis”) is pushing global fertilizer prices up (Chowdhury et al., 2017).

The high P fixing capacity of most soils and the low P use efficiency (around 10-15%) of most crops means that surplus P input from fertilizers tends to accumulate in soils (Withers et al., 2001). This accumulation is known as “legacy P” which can be calculated as the difference between inputs (mineral P fertilizer, atmospheric deposition and weathering) and outputs (lost through surface runoff, subsurface flow, leaching, plus P in crop uptake etc.) (Havens and James, 2005; Sattari et al., 2012). In Oceania and Western Europe, for example, cumulative inputs of P fertilizer to arable land (560 and 1115 kg P ha⁻¹, respectively, for the period 1965-2007) were much greater than the cumulative uptake (100 and 350 kg P ha⁻¹, respectively) (Sattari et al., 2012). In turn, this results in changes to the concentration of P in the soil solution and its association with soil minerals and organic matter which can lead to enhanced phosphate loss (Guppy et al., 2005; Heckrath et al., 1995) and subsequent degradation of freshwater and marine resources where eutrophication can be triggered by additional P inputs (Tilman et al., 2001).

It is clear that legacy P stocks in soils have the potential to play a vital role in maintaining agricultural productivity with lower P requirements for inputs and reduced P transfers from land to water, if crops can efficiently access this P (Condron et al., 2013). It has even been suggested that the accumulated P in agricultural soils would be sufficient to sustain maximum crop yields worldwide for about 100 years if it were available (Khan et al., 2007).

Unfortunately, most legacy P is not available for plants to absorb easily. However, it may be possible to manipulate soils to increase the availability of this P for crops. The aims of this review are to clarify our current understanding of the cycling and transformation of legacy P in agricultural soils and to present findings from the published literature about so-called “P activators”. Furthermore, we evaluate the contribution of P activators to soil legacy P availability, assess the advantages and disadvantages of different P activator classes and summarize the key mechanisms involved.

2. The forms and mobility of soil P

More than 80% of P applied as fertilizer can become unavailable for plant uptake shortly after application via sorption, precipitation (typically by reaction with Al³⁺ and Fe³⁺ in acidic soils and Ca²⁺ in calcareous soils: Table 1) or microbial immobilization (Gustafsson et al., 2012; Roberts and Johnston, 2015; Yadav and Verma, 2012). To take full advantage of legacy P, it is necessary to understand the long-term
dynamics of this P in soil (Liu et al., 2014a).

Soil P occurs in a number of different forms (organic and inorganic) which vary greatly in their bioavailability (Fig. 1). Those include constituents of rock minerals from the parent material, which are completely inaccessible, various forms of organic matter (including the constituents of humus and the cells of living organisms: Khan et al., 2014) and bioavailable phosphate in solution (Dollard and Billard, 2003). The environmental behavior of P is also a function of its speciation, which is directly linked to P solubility, reactivity, and bioavailability (Liu et al., 2014b). For example, P associated with Fe (hydr) oxides (Fe-P) is sensitive to reducing conditions (Beauchemin et al., 2003). P sorbed to Al (hydr)oxides (Al-P) or calcium phosphate precipitates is likely to be more sensitive to pH changes (Yan et al., 2014), and some species of organic P tend to be more bioavailable (Li and Brett, 2013).

The total P in top soils (0-15 cm) typically ranges from 50 to 3,000 mg P kg⁻¹, depending on parent material, soil texture, vegetation cover and soil management history (Sanyal and De Datta, 1991). According to Stutter et al. (2015), based on 32 soils from the United Kingdom, arable soils tend to be dominated by ortho-P, with orthophosphate monoester species associated with strongly sorbing Al and Fe soil surfaces. Intensive grasslands tend to be dominated by orthophosphate monoesters and as grazing becomes more extensive labile orthophosphate diesters, associated with labile soil organic matter and microbial turnover, dominate. The diversity of P species in less intensively managed soils can be viewed as a positive indicator of ecosystem function and diversity.

P₁ generally accounts for 60-80% of total P in agricultural soils (predominantly in minerals). The total P content is usually at least one order of magnitude larger than the amount of P that rapidly cycles through the soil-plant system and two to three orders of magnitude larger than the amount of P present as plant-available P₁ in the soil solution (HPO₄²⁻ or H₂PO₄⁻) (Frossard et al., 1995). The dominant form of orthophosphate ion present in the soil is pH dependent. At pH 4-5, orthophosphate usually exists as H₂PO₄⁻ ions but as pH increases, first HPO₄²⁻ ions and then PO₄³⁻ ions become more dominant (Yadav and Verma, 2012).

Many abiotic and biotic reactions occur in the soil P cycle (Fig. 1), some occurring within a few seconds, while others occurring slowly over several years (Bünemann and Condron, 2007; Fardeau, 1995; Frossard et al., 2011). The initial breakdown can often be the rate-limiting step for Pₐ mineralization (Das et al., 2014). P availability is also greatly affected by a series of pH-dependent abiotic reactions that influence the ratio of soluble-to-insoluble P pools in the soil (DeLuca et al., 2009). These include desorption and solubilization of rapidly exchangeable P; uptake of P by soil microorganisms and plants (either through their roots or through mycorrhizal hyphae); the release of P₁ from the soil solid phase or from fertilizers, induced by the exudation of phosphatases and organic acids from roots or microorganisms; the release of P₁ from plant residues or organic fertilizers to the soil solution; and the release of P₁ from organic matter mineralization by microorganisms (Frossard et al., 2011).

The concentration of P in the soil solution is primarily controlled by adsorption/desorption equilibria
between labile \( P_i \) in the solid phase (associated with positively charged minerals such as Fe and Al oxides) defined by equilibrium concentration ratios (Haynes and Mokolobate, 2001; Hinsinger, 2001). Specific adsorption (ligand exchange) occurs when \( P \) ions replace the hydroxyl groups on the surface of Al and Fe oxides and hydrous oxides (Haynes and Mokolobate, 2001). \( P \) ions can also interact chemically with minerals in combination with metals such as calcium phosphate, aluminum phosphate, iron phosphate and the reductant-soluble phosphate extractable after removal of the first three forms (Chang and Jackson, 1957). For example, oxygen (\( O_2 \)) can immobilize iron through Fe(II) oxidation which precipitates as Fe(III) (hydr) oxides, resulting in the generation of \( H^+ \) and a decrease in pH (Eq. (1)) (Begg et al., 1994). This process may have some effects on the trapping of phosphate as FePO\(_4\) (Eq. (2)) (Silva and Sampaio, 1998).

\[
\begin{align*}
4 Fe^{2+} + O_2 + 10 H_2 O & \rightarrow 4 Fe(OH)_3 + 8 H^+ \quad (1) \\
Fe(OH)_3 + H_2 PO_4^- & \rightarrow FePO_4 + OH^- + 2 H_2 O \quad (2)
\end{align*}
\]

\( P_o \) may constitute 30-50% in most soils, although it can range from as low as 5% to as high as 95% (Sharpley, 1985). \( P_o \) is typically present as orthophosphate monoesters, including inositol phosphates, and as orthophosphate diesters, organic polyphosphates and phosphonates (Bünemann, 2015). \( P_o \) can be classified into four main fractions: labile, moderately labile, moderately resistant (fulvic-acid \( P \)), and highly resistant (humic-acid \( P \)) (Bowman and Cole, 1978; Randriamanantsoa et al., 2015). All fractions can be transformed into phosphate that is available to plants via microbially mediated mineralization – but at different rates. Labile \( P_o \) is relatively easily mineralized whereas inositol phosphate (up to 50% of \( P_o \)) is considered to be stable (Dalai, 1977). The microbial biomass acts as both a source and a sink of soil \( P \) and is critical for \( P \) cycling (i.e. mineralizing \( P_o \) via enzyme-catalyzed transformations and converting \( P_i \) into their living cells during \( P \) immobilization).

Different soil \( P \) fractions can also be separated into fractions characterized operationally by the used extractants (Hedley et al., 1982): e.g. resin-\( P \), NaHCO\(_3\)-\( P \), microbial-\( P \), NaOH-\( P \), Sonicated/NaOH-\( P \), HCl-\( P \) and residual-\( P \). In this kind of \( P \) fractionation, only resin-\( P \) and NaHCO\(_3\)-\( P \) are relatively available for crops but these account for a small proportion of total \( P \).

Both \( P_i \) and \( P_o \) are strongly bound (or fixed) to the solid phase under most soil conditions via a series of different reactions. Together with \( P \) sorption and net immobilization by the microbial biomass, this can result in low equilibrium \( P \) concentrations in the soil solution which can be manifested as \( P \) deficiency (e.g. reduced crop growth), even in moderately fertilized soils (Gerke, 2015). This can be reversed via dissolution/precipitation (mineral equilibria), sorption/desorption (interactions between \( P \) and mineral surfaces) and mineralization/immobilization (biological transformations between \( P_o \) and \( P_i \)).

### 3. Soil \( P \) activators and the application effect

\( P \) activators refer to a range of different methods intended to accelerate and strengthen soil \( P \)
transformations to plant-available forms in the soil solution. They can be classified into three types: (1) bio-inoculants and bio-fertilizers, including phosphate solubilizing microorganisms (PSMs) and phosphatase enzymes; (2) organic matter, including low molecular weight organic acids, humic acids, lignin, crop residue, manure and biochar and (3) zeolites and other materials, which have been relatively less well studied and applied.

Although activation processes are complex, several key factors have consistently been demonstrated to govern the availability of fixed P. These main factors can be broadly grouped as sorption reactions and changing the chemical and physical properties of the soil (e.g. pH, extractable Al and the net negative charge on surfaces in soil) with the net effect of increased solubility and reduced sorption of P; in soil. It is complex to elucidate the effect of P activators based on changing of these factors. For a review of this nature, it would be useful to define a common and clearly explained set of responses to P activation. Here, we employ the P activation response based on an increase in plant-available P or on saved phosphate fertilizer inputs – focusing on experimental data (field and glasshouse) that excluded the impacts of management practices (e.g. grazing, intercropping).

3.1. Bio-inoculant and bio-fertilizer

3.1.1. Phosphate solubilizing microorganisms

Various organisms are involved in P cycling but microorganisms probably play the most important role. Much of the global cycling of insoluble organic and inorganic soil P can be attributed to bacteria and fungi. Phosphate solubilizing microorganisms are those which are capable of transforming insoluble P into soluble (plant accessible) forms. They are probably the most eco-friendly and inexpensive option for enhancing P availability for plants (Owen et al., 2015). Phosphate solubilizing bacteria (PSB: *Achromobacter, Acreobacter, Agrobacterium, Alcaligenes, Arthrobacter, Aspergillus, Azotobacter, Bacillus, Bradyrhizobium, Burkholderia, Chromobacterium, Enterbacter, Erwinia, Escherichia, Flavobacterium, Klebsiella, Micrococcus, Pantoea agglomerans, Pseudomonas, Rhizobium, Salmonella, Serratia, Thiobacillus*) and phosphate solubilizing fungi (PSF: *Alternaria, Arbuscular mycorrhiza, Aspergillus, Fusarium, Helminthosporium, Penicillus, Rhizopus, Sclerotium*) make up 1-50% and 0.1-0.5% respectively of the total PSMs in soil with an additional minor role played by phosphate solubilizing actinomycetes (PSA: *Streptomycyes, Nocardia*) (Khan et al., 2007). It has long been known that there is significant variation in the ability of bacteria to solubilize P in soil. Bacteria which are known to enhance P availability include species of *Pseudomonas, Azotobacter, Burkholderia, Bacillus* and *Rhizobium* (Jones and Oburger, 2011). Generally, PSF produce more acids than bacteria and consequently exhibit greater P-solubilizing activity. Filamentous fungi known to be able to solubilize phosphate include the genera *Aspergillus* and *Penicillus* (Sharma et al., 2013). In recent years, the P-solubilizing ability of actinomycetes has attracted interest because this group of soil organisms are not only capable of surviving in extreme environments (e.g. drought, fire etc.) but also possess other potential features (e.g. production of
antibiotics and phytohormone-like compounds etc.) that could simultaneously benefit plant growth (Hamdali et al., 2008).

The P solubilizing ability of different PSMs varies with species, strain, soil properties (inherent and seasonal) and the plant species (Table 2). A maximum increase in available P of 208% has been reported (Abdul Wahid and Mehana, 2000). However, de Freitas (1997) found that application of PSB (Bacillus sphaericus, B. polymyxa and B. thuringiensis) significantly increased plant growth, but not P-uptake. Other studies suggest that mixtures of different PSMs may be more effective than using single organisms (Yu et al., 2011). Kim et al. (1997) reported that when vesicular-arbuscular mycorrhizae (VAM; Enterobacter agglomerans) and PSB (Glomus etunicatum) were applied together, the soil microbial colonization and phosphatase activity response was much higher than when they were used separately, suggesting a synergistic interaction, although the exact mechanisms were not fully understood. It is likely that the additional increase in P resulting from dual inoculation was due to enhanced P solubilization by PSB through the production of organic acids, ion chelation and increasing root cell permeability, as well as supporting the establishment and function of VAM (mycorrhizal colonization alters roots physiologically as well as altering microbial populations: Suri et al., 2011).

Various “bio-fertilizers” have been developed which include bacteria (e.g. Pseudomonas spp, Bacillus spp, Rhizobium spp and Klebsiella spp), fungi (e.g. Penicillium spp, Aspergillus spp and Rhizopus spp) and actinomycetes (e.g. Streptomyces spp). These organisms are intended to solubilize solid phosphate in soils (Mahajan and Gupta, 2009). They are cost effective, have no known environmental issues and can be used to supplement chemical fertilizers (Chesti et al., 2013). It has been estimated that inoculation with arbuscular mycorrhizal fungi (AMF) might result in a reduction of approximately 80% of the recommended phosphate fertilizer rate under certain conditions, with similar crop yields (Liu et al., 2016). The effectiveness of mycorrhizal associations on P uptake from slow-release P compounds has been elucidated in the review of Bolan et al. (1991). They noted that for plants the increase in P uptake associated with mycorrhizal infection was found to vary with source of P. Greatest benefit from mycorrhizal inoculation was obtained with the least soluble source of iron phosphate. Compared to normal phosphate fertilizers, dissolved P microbial mixed fertilizers appear to have a significant effect on the utilization of P – enabling a reduction in phosphate fertilizers by 25-467% (Mukherjee and Sen, 2015; Sundara et al., 2002). The effect of PSMs varies at different phases of plant growth. Sundara et al. (2002) shows that the effect of PSB fertilizer treatment on available P status varied with sugarcane growth stage. PSB fertilizers were more effective in the tilling and growth phases than during the ripening phase.

Although it is widely known that PSMs can make P available, the mechanisms underlying this phenomenon are not fully understood. The potential mechanisms are illustrated in Fig. 2 and include the microbial synthesis of a range of exudates (e.g. organic acids and H+, metabolites and enzymes including phosphatase) (Behera et al., 2014) The insoluble forms of P, such as tricalcium phosphate (Ca₃(PO₄)₂), aluminium phosphate (AlPO₄), iron phosphate (FePO₄), may be converted to soluble P by low molecular
weight organic acids as well as facilitated by microbially released H⁺ (Khan et al., 2014). Li et al. (2015) studied mechanisms for solubilizing different types of P by an efficient Aspergillus niger strain An2. They found An2 mainly secreted oxalic acid to solubilize Ca-P, Mg-P, Al-P and Fe-P and secreted tartaric acid to solubilize rock phosphate.

PSMs can secrete enzymes (phosphatase, phosphohydrolase, phytase, phosphonatase and C-P lyase) which catalyze P₀ mineralization. The most commonly secreted enzymes are phosphatase (an enzyme which eliminates P from its substrate by hydrolyzing phosphoric acid monoesters into a P ion and a molecule with a free hydroxyl group.) and phytase because of the predominant presence of their substrates in soil (Othman and Panhwar, 2014). Phosphatases or phosphohydrolases describe a broad group of enzymes that catalyze the hydrolysis of both esters and anhydrides of H₃PO₄ (Jones and Oburger, 2011). Phytase (myoinositol hexophosphate phosphohydrolase) can hydrolyze sodium phytate, resulting in the production of P₀ (Sharma et al., 2013). Phosphonatase and C-P lyase cleave the C-P of organophosphonate. Wanner (1996) found that many enzymes can function as organic phosphoester scavengers, releasing inorganic phosphates from nucleotides and sugar phosphates. PSMs can also solubilize phosphate via the production of NH₄⁺, CO₂ and H₂S, as well as releasing P during substrate degradation (Khan et al., 2010). The production of H₂S can act with ferric phosphate to yield ferrous sulphate with concomitant release of phosphate. The presence of NH₄⁺ can result in pH depression via the formation of H⁺ during nitrification which contributes to P solubilizing (Khan et al., 2014). Proton release also occurs from the dissociation carbonic acid (H₂CO₃) which is formed from the dissolution of CO₂ generated by biological respiration. These protons can readily solubilize Ca-apatite (Kim et al., 1997):

\[
\text{Ca}_5(\text{PO}_4)_3\text{OH} + 7\text{H}^+ \rightarrow 5\text{Ca}^{2+} + 3\text{H}_2\text{PO}_4^- + \text{H}_2\text{O} \quad (3)
\]

3.1.2. Phosphatase enzyme and enzyme activators

Phosphatase enzymes are widely distributed in natural environments and play a major role in P₀ regulation (Fig. 3) by hydrolyzing ester-phosphate bonds in P₀, leading to the release of phosphate (Burns and Dick, 2002).

Phosphodiesterase and phosphomonoesterase may act sequentially in the mineralization of P₀. phosphomonoesterase can dissociate the phosphate group from phosphate monoester compounds and phosphodiesterase can hydrolyze the phosphate diester bond in nucleic acids. Lipids and nucleic acids typically make up about 7% of soil P₀ (Kalsi et al., 2016). The influence of phytase on P₀ mineralization is less well understood but is thought to be important in the mineralization of inositol P (which makes up approximately 50% of soil P₀ in the form of phytin and its derivatives). Phytase can release lower forms of myo-inositol phosphates and inorganic phosphate by catalyzing the hydrolysis of phosphomonoester bonds in phytate. Xiang et al. (2005) observed that soil phosphatase activity and available P content increased significantly after applying exogenous phytase in potted Malus hupehensis experiments.
D-α-glycerophosphatase is a rare phosphatase in soil which has been purified and characterized from *Bacillus licheniformis* (Skraly and Cameron, 1998). The products of the reaction catalyzed by D-α-glycerophosphatase were identified as glycerol and inorganic phosphate. In addition, a kind of inorganic phosphatase (pyrophosphate phosphohydrolase) has been identified which can hydrolyze pyrophosphate (used as a fertilizer) to Pi (Dick and Tabatabai, 1978).

It has been shown that the activities of phosphatase (like those of many hydrolases) depend on several factors including soil properties, soil organism interactions, plant cover, leachate inputs and the presence of other inhibitors and activators (Bünemann et al., 2011).

Phosphatase activity varies with pH (different phosphatases are classified as either alkaline (pH>7) or acid (pH<6) phosphatase). Generally, enzyme activities of soils increase with increasing soil pH with the exception of acid phosphatase which is predominant in acid soils. It is worth underlining that when the effects of soil pH on enzyme activities are studied, the concentration of organic matter should remain constant (Gianfreda and Ruggiero, 2006). For example, the significantly greater activities of alkaline phosphomonoesterase, inorganic pyrophosphatase and phosphodiesterase reported in manure-treated soils by Acosta-Martínez and Tabatabai (2000) are believed to be due to a combination of greater soil pH values and enhanced microbial activity and diversity resulting from manure input over the years. Martens et al. (1992) studied the activities of ten soil enzymes after the application of different organic amendments (poultry manure, sewage sludge, barley straw and green alfalfa) over a 31-month period. They found that the addition of amendments increased enzyme activity by an average of 2- to 4-fold during the first year of the experiment. Similarly, phosphatase activities (acid phosphate, alkaline phosphatase, phosphodiesterase and inorganic pyrophosphatase) were significantly correlated with the organic carbon content of the 40 soil samples tested by Deng and Tabatabai (1997), confirming that organic matter plays an important role in protecting and maintaining soil enzymes in their active forms.

It is well established that phosphatase activities generally decrease with soil depth (Gianfreda and Ruggiero, 2006). Ji et al. (2014) describe a two-year field experiment on the effects of deep tillage on soil microorganisms and phosphatase activity in clay and loam soils. The data showed that tillage and soil texture had a significant influence over soil phosphatase activities and that deep tillage increased soil phosphatase activities, which was affected by soil depth. This may be because deep tillage loosens the soil and mixes organic matter into lower horizons, thereby increasing the abundance of soil microorganisms. Activities of phosphatase in clay soil were higher than those of loam by 10.9%. This may have been due to the fact that the finer particles in clay soils offer more surface area and more small pores which can trap organic matter and offer habitat for soil microorganisms (which are the source of phosphatases). Das et al. (2014) also found that soil phosphatase activity decreases with increasing depth. Soil temperature also has a significant impact on soil phosphatase activity. Higher temperatures can encourage microbial metabolism, increasing phosphatase activity and nutrient availability (Dalai, 1977). However, previous studies have also shown a significant correlation between phosphodiesterase activity...
and organic carbon content which complicates the explanation for this effect (i.e. the degradation of organic
carbon is also correlated with temperature) (Koch et al., 2007; Štursová and Baldrian, 2011).

Some stressors such as salinity and high heavy metal concentrations may have adverse effects on the
phosphatase activities. Alkaline phosphatase was observed to decline exponentially with increasing salinity
and linearly with increasing sodicity by Rietz and Haynes (2003). This was ascribed to the denaturation of
enzyme proteins at high salt concentrations and a decrease in microbial biomass and activity. Heavy metals
affect enzyme activity by modifying protein conformation, due to interaction with the protein active groups
and by inhibiting enzyme synthesis (Karaca et al., 2010). Kandeler et al. (2000) found that phosphatase
activities decreased after 10 years of contamination with different levels of Zn, Cu, Ni, V and Cd.

Although, in principle, some P activators can strengthen (accelerate) the activity of phosphatase and,
hence, activate $P_o$ (Fig. 3b), relatively few studies have been published on this topic. An exception is the
work of Fang et al. (2008) which showed that the addition of glutathione enhanced the affinity between
enzymes and substrates. Ascorbic acid had a similar effect to glutathione on enzyme activity. They take part
in redox reactions and can combine with peroxide and free radicals to protect sulfhydryl from oxidation. As
a result, proteins and enzymes containing sulfhydryl in membranes will not deactivate (Wisnewski et al.,
2005). Li et al. (2014) found that soil available P content could be increased by 10-91% under the influence
of glutathione in paddy soils. They also demonstrated that ascorbic acid promoted the activity of
phosphomonoesterase and accelerated the mineralization of $P_o$. The available P content consequently
increased by 2-203%.

3.2. Organic matter

Organic matter applied to soil as an amendment is also reported to increase the availability of existing
soil P. A comprehensive review of the competitive sorption reactions between phosphorus and organic
matter in soil is given by Guppy et al. (2005). Details of how organic matter contributes to available P in
soil is reviewed further in this Section.

3.2.1. Low molecular weight organic acids

Low molecular weight organic acids are organic compounds that contain at least one carboxyl group
and a molar mass less than 10,000 g mol$^{-1}$. They are thought to activate P since plants secrete them under
conditions of P deficiency and they are also derived from the decomposition of plant and animal residues
(Yuan et al., 2016). The most common low molecular weight organic acids identified in soils are oxalic,
malic, citric, fumaric, tartaric and succinic acids (Table 3). The most frequent agents used for mineral
phosphate solubilization seem to be gluconic acid and 2-ketogluconic acid (Earl et al., 1979; Moradi et al.,
2012). The concentrations of low molecular weight organic acids tend to be low in soils, ranging from
$10^{-3}$-$10^{-5}$ mol L$^{-1}$.

Different concentrations of low molecular weight organic acids have been added to soil in an attempt
to increase P availability. Good P activation responses have been obtained using citric and oxalic acids (Table 4). Low molecular weight organic acids have also been added to soils with phosphatase enzymes in an attempt to activate P. For example, Guan et al. (2013) showed that the application of oxalic acid and phytase together significantly increased available P compared to controls. Amounts of $\text{H}_2\text{O-P}$, $\text{NaHCO}_3\text{-P}_\text{i}$, NaOH-P$_\text{i}$ and NaOH-P$_\text{o}$ increased by 37.8%, 21.1%, 15.5% and 24.4%, respectively, possibly as a consequence of the transformation of stable P fractions to labile fractions by phytase under the influence of exogenous oxalic acid. However, the detailed mechanisms remain speculative.

Low molecular weight organic acids may activate P$_\text{i}$ via: (1) changing soil pH which can promote the dissolution of sparingly soluble minerals containing P$_\text{i}$ (Fox et al., 1990), including calcium phosphate, e.g., octacalcium phosphate or apatite (Andersson et al., 2015); (2) forming complexes with Fe, Al and Ca and releasing P combined with these ions (Ström et al., 2005). The release of phosphate occurs via ligand exchange between the carboxylate anion and HPO$_4^{2-}$ or H$_2$PO$_4$; (3) organic acid ions compete with P$_\text{i}$ for the same sorption sites in soils (Lan et al., 1995). When P and organic acid anions are present in the soil solution, competition is expressed either through direct physical competition for sorption sites or through electrostatic competition.

Low molecular weight organic acids may also promote the release of P$_\text{o}$. Giles et al. (2014) found that the addition of organic anions from bacterial sources can improve Ca myo-inositol hexakisphosphate solubilization and, thus, P availability. Organic anion-driven solubilization of Ca myo-inositol hexakisphosphate appears complementary to hydrolysis by plant and bacterial phytases. In vitro experiments suggest that Ca myo-inositol hexakisphosphate hydrolysis is improved in the presence of organic anions (e.g., gluconate, citrate, oxalate and acetate), due either to Ca$^{2+}$-mediated phytase activation or solubilization via divalent metal chelation (Patel et al., 2010). However, the exact mechanism about possible dissolution of P$_\text{o}$ by organic acids remains unclear. It is possible that low molecular weight organic acids could influence the activity of PSMs and, hence, affect the activity of phosphatases secreted by these organisms (Turner, 2008).

### 3.2.2. Humic acids and lignin

Humic acids and lignin are natural high molecular weight organic compounds derived from the decomposition of plant litter. Humic acids are composed of a range of aliphatic and aromatic structures with a variety of different functional (mainly oxygen-containing) groups. They can improve soil fertility and increase the availability of nutrient elements including P (Çimrin et al., 2010).

Humic acids can activate P via the following mechanisms: (1) Shifting soil pH; H$^+$ is produced during the decomposition of humic acids in soil (Hue, 1991) which can increase soluble P concentrations in calcareous soil by decreasing precipitation rates of Ca-P minerals. H$^+$ from humic acids can also inhibit the precipitation of hydroxyapatite, and favor the formation of dicalcium phosphate dihydrate over other, more thermodynamically stable and less soluble phosphates. The effects appear to be more effective than low
molecular weight acids (Grossl and Inskeep, 1991); (2) Formation of simple organic ligands; Humic acids
contain some active functional groups such as carboxyl and phenolic hydroxyl groups that can complex
metal ions, especially Fe and Al (which are often associated with phosphate fixing). P availability increases
when these metals are removed (Antelo et al., 2007; Gerke and Hermann, 1992). (3) Colloid interactions;
Humic acids are colloidal substances and can adsorb metal and metal oxides in soil and strengthen the
competitive ability of phosphate (Regelink et al., 2015). Competition between humic acids and P for
sorption onto soil colloids may result in higher P dissolved concentrations.

It is worth noting that humic acids may also act as P-sorbing surfaces in some circumstances. Othieno
(1973) added humic acids and P to highly weathered soils and repeatedly observed wheat yield decreases
and lower tissue P concentrations, suggesting that humic acids can have a detrimental effect on P
availability for plants. Further elucidation of the relationships between humic acids and P in competitive
sorption processes is necessary, because both the increase and decrease of P bioavailability may occur in
the presence of humic acids.

Furthermore, the addition of humic acids can also increase the distance of P movement and the
concentration of extractable P in soil surrounding phosphate fertilizer placements (Du et al., 2013). A
similar conclusion was reached by Jones et al. (2007). They found that soluble P concentrations were nearly
identical between soils prior to fertilization (ranging from 0.1 to 0.2 mg L\(^{-1}\)). Soluble P concentrations 3.8
cm below the mono-ammonium phosphate band in the calcareous soil were significantly higher when
humic acids were applied than in the control after 16 and 32 days, suggesting that humic acids may have
increased P solubility and mobility. Fig. 4 demonstrates that P can move relatively far from a phosphate
fertilizer band in a neutral pH, non-calcareous soil after humic acids addition (Jones et al., 2007). Soluble P
concentrations were higher 3.8 cm below the band (Fig. 4c) than 3.8 cm to the side of the band (Fig. 4b),
indicating that there was some downward advection of P.

Lignin is an important precursor in the formation of humic substances and can form humic acids as a
result of microbial activity in soil (Bååth et al., 1995). The major chemical functional groups in lignin
include hydroxyl, methoxyl, carbonyl and carboxyl groups in various amounts and proportions. Lignin can
facilitate the activation of P via similar mechanisms to those described above for humic acids (Sun et al.,
2011), as they share similar functional and structural properties (Tahir et al., 2011). Lignin has been
reported to increase available P in calcareous, red ferrallitic (rich in iron-aluminum oxides), yellow
ferrallitic (rich in ferric oxide hydrate), sandy and humid-thermo ferrallitic soils. Increases in available P of
35% in calcareous soil and 33% in red clay loam have been reported - probably from Al-P complexes
(Chen et al., 2003).

Lignin can be extracted, recycled and modified from black liquor (a waste product from paper pulp
manufacture), bagasse by-products (fibrous sugar cane or sorghum residues) and crop straw. The lignin
content in black liquor solids (which are now regularly applied to agricultural soils) can be up to 35% (Tian
et al., 2015). Dotaniya et al. (2014) reported that organic residues such as press mud and bagasse
by-products could release organic acids and enhance P availability in soil. About 600 million tonnes of crop straw are produced every year in China (Zeng et al., 2007) which is often re-incorporated into soil – increasing both the lignin and organic matter contents. As a result, the content of available P, total P and the activity of phosphatase have been observed to increase (Ding et al., 2012).

3.2.3. Crop residues, manure and biochar

Like chemical phosphate fertilizer, crop residues, manure and biochar (charcoal produced from crop residues) can also act as P sources. Land application is the predominant method for disposing of agricultural residues, thereby recycling their nutrients Fig. 5) (Dai et al., 2016). The magnitude and direction of available P change after the application of these materials to soil depends on the properties of the P source and the receiving soil environment.

All of these materials contain highly soluble P\(_i\) (e.g. orthophosphate) which can act as a primary source of nutrients. They also contain labile P\(_o\) which can be mineralized to release soluble P in the short-term. This soluble orthophosphate can also be assimilated by microorganisms and subsequently released back into soil when these organisms die and are subject to decomposition. Results from a dual-labelling experiment suggest that, on average, 16% of the P contained in surface-applied residues could be classed as labile P (Noack et al., 2014).

A review by Damon et al. (2014) concluded that several key factors govern the mineralization and availability of crop residue-derived P including the quality of the crop residues, the activity of the soil microbial biomass, and the subsequent sorption reactions of mineralized P in soil. The addition of organic residues can also alter the chemical and physical properties of the soil and, hence, affect P availability indirectly. For example, increased organic matter usually provides microbes with a more favorable habitat, enhancing P cycling via mineralization (Lone et al., 2015). However, a general finding is that the transfer of P from break crop residues incorporated into the soil is relatively low (2-20%) and is quite variable (Espinosa et al., 2017). This could be due to variable P concentrations in the crop residues, which depend on the soil P status, the physiological maturity of the residues, and on the extent of the translocation of shoot-P to the developing grain in the break crops under considerations (Thibaud et al., 1988).

Manure P is a valuable resource, comparable to inorganic fertilizer P for crop production. Compared to crop residues, manure can contain even more labile P. A large fraction of the phosphorus in manure is considered to be plant-available immediately after application. Estimated values of phosphorus availability ranged between 12% and 100% (Bahl and Toor, 2002). The total P content in manure varies depending on the animal species, age, diet and how the manure has been stored. Jia et al. (2015) predicted that if manure P inputs from 2012 to 2030 were assumed to continue in soil, Olsen-P could increase to values close to or exceeding the environmentally critical value of 60 mg kg\(^{-1}\), the tipping point above which P leaching has been shown to be significant (Heckrath et al., 1995). In addition to supplying labile P, manure can also help in mobilizing native P. This mobilization occurs by conversion of insoluble Ca, Al and Fe forms of P to
soluble forms through the action of organic acids and chelates that are produced during manure decomposition of manures (El-Baruni and Olsen, 1979). Hountin et al. (2000) found that labile (resin-\(\text{P} \)) + \(\text{NaHCO}_3\)-\(\text{P} \) + NaOH-\(\text{P} \)) in animal manure-amended soil increased from 61% to 79% and the amounts of \(\text{P} \) generally increased in the following order: moderately labile NaOH-\(\text{P} \) and NaOH-\(\text{P} \) > labile \(\text{P} \) extracted by resin and NaHCO\(_3\) > stable \(\text{P} \). However, the potential environmental \(\text{P} \) hazard of swine manure should be considered, especially in agricultural systems in which soils receive high rates of manure application. Although \(\text{P} \) is considered immobile in agricultural soils, a decrease in the \(\text{P} \) adsorption capacity of soil following manure addition at rates greater than the nutritional requirements of plants for \(\text{P} \) may increase the potential for leaching of soluble \(\text{P} \) (Sutton et al., 1982).

The effects of biochar on \(\text{P} \) availability are variable: in some soils, biochar application has been observed to increase \(\text{P} \) availability, while in others, \(\text{P} \) availability was not affected or was observed to decrease, mainly due to increased \(\text{P} \) sorption (Chathurika et al., 2016). Biochar can induce changes in the soil ion exchange capacity. Fresh biochar in the acid pH range has a high anion exchange capacity, which can initially be in excess of the total cation exchange capacity of the biochar (DeLuca et al., 2009). It is possible that these positively charged exchange sites compete with Al and Fe oxides (e.g., gibbsite and goethite) for sorption of soluble \(\text{P} \), similar to that observed for humic acids and fulvic acids (Hunt et al., 2007). \(\text{P} \) associated with biochar can be more labile than that associated with Al or Fe oxides. To date, however, there is noted lack of studies evaluating the effect of short-term anion exchange capacity on \(\text{P} \) cycling and availability. Biochar may also alter \(\text{P} \) availability through sorption of chelating organic molecules like phenolic acids, amino acids and complex proteins or carbohydrates (Joseph et al., 2010). Sorption of organic molecules on biochar surfaces can directly adsorb cations such as Al\(^{3+}\), Fe\(^{3+}\) and Ca\(^{2+}\), resulting in delayed \(\text{P} \) adsorption or precipitation in soil (Xu et al., 2014). A microcosm incubation study by Jin et al. (2016) suggested that soil \(\text{P} \) availability was enhanced after the addition of manure biochar (biochar from manure wastes) due to the fact that orthophosphate and pyrophosphate are the major \(\text{P} \) species in manure biochar and due to enhanced decomposition of some \(\text{P} \) (e.g. monoesters) by enhanced alkaline phosphomonoesterase activities. Xu et al. (2016) reported that biochar application can decrease \(\text{P} \) availability in saline sodic soil due to enhanced \(\text{P} \) sorption and precipitation.

### 3.3. Zeolites and other materials

Zeolites are minerals of hydrated sodium, potassium, calcium and barium aluminosilicates which are often used as cation exchangers (the net negative structural charge of zeolites results in the favorable ion-exchange selectivity for many cations) (Pabalan and Bertetti, 2001). They can increase soil available \(\text{P} \) concentrations and plant uptake of \(\text{P} \). Wei at al. (2001) found that zeolite application could increase soil available \(\text{P} \) by up to 183%. Some modified zeolites can increase the available \(\text{P} \) even more (Yang et al., 2015). Fig. 6 gives a summary of several methods for the modification of natural zeolites (Dai et al., 2011) to enhance cation exchange capacity (due to the favorable ion-exchange selectivity of natural zeolites for...
certain cations, such as Cs⁺, Sr²⁺, and NH₄⁺) and to the pH change induced by zeolites. Acid modified zeolites can decrease the pH of soil and alkali modified zeolites can enhance cation exchange through alkali metal ions. As a result, the release of P can be enhanced (Fadaeerayeni et al., 2015). Modification at high temperatures can remove organic matter in zeolite crevices and increase adsorptive capacity, thereby enhancing the capacity of zeolite to activate P (Quin et al., 1998). The combination of zeolite and NH₄⁺ can also increase the cation exchange effects of zeolite. The exchange-induced dissolution of rock phosphate proposed can be summarized for an ammonium saturated zeolite as Eq. (4):

\[ \text{RP} + \text{NH₄⁺-zeolite} \rightarrow \text{Ca²⁺-zeolite} + \text{NH₄⁺+PO₄³⁻} \]  (4)

where RP is rock phosphate (Allen et al., 1993). The zeolite/RP combination was intended to act as an exchange-fertilizer, with Ca²⁺ exchanging onto the zeolite. It is proposed that the plant uptake of NH₄⁺ or K⁺ frees exchange sites which can then be occupied by Ca²⁺, lowering the soil solution Ca²⁺ concentration and inducing further dissolution of rock phosphate (Pickering et al., 2002).

Fly ash, a coal combustion product (Parab et al., 2015) and wood ash can also be used to activate P in soil (Demeyer et al., 2001). One possible reason for the significant improvement of P availability by fly ash in acid soil was that ash addition had a liming effect which can enhance bioavailable P concentrations (Parab et al., 2015). However, fly ash may also be associated with high heavy metal loads. Compared to coal ash, reported concentrations of Mn, Zn and Cd in wood ash are higher while As, Se and Cr contents are lower (Someshwar, 1996). In addition to P, wood ash is a direct source of other major elements, notably Ca, Mg and especially K in soils. The dissolution of wood ash in soil, and the rate at which P becomes plant-available is complicated. The change in soil nutrient availability is a combination of three factors: (1) P addition from the ash; (2) shifts in pH-dependent soil chemical equilibria; (3) changes (mostly increases) in microbial activity (Demeyer et al., 2001). Furfural is an important organic compound produced from agro-industrial wastes and residues. When applied to soil, it can decrease pH and release Ca²⁺-bound phosphate (García-Domínguez et al., 2015).

Other P activators include water-dissolved organic polymeric compounds, bentonite, mugineic acid and ABT rooting powder (ABT has the ability to promote rooting and to increase the survival rate of plants under abiotic stress). Polyvinyl alcohol, polyacrylamide and polyethylene glycol have been shown to increase P sorption and pH in red ferrallitic soils (rich in ferric oxide hydrate). Polyvinyl alcohol is a polymer containing hydroxyl groups, which has low permeability and high water adsorption capability (Chai et al., 2012). Polyacrylamide is a polymer of acrylamide which is soluble in polar solvents and nontoxic in nature. Polyethylene glycol has special solubility and chain-folding properties. Phosphate is released as a consequence of the protonation of hydroxyl and amide groups in these water-dissolved organic polymers and their interaction with Al³⁺ and Fe³⁺phosphate (Hua et al., 2005). Bentonite can also increase P in soil (principally Ca₂⁻P followed by Ca₈⁻P, Al-P and Fe-P) (Liu and Sun, 2004). Mugineic acid
has a high affinity for Fe and can activate Fe-P (Zhang et al., 1989). ABT rooting powder is an efficient plant growth promoter developed by the Chinese Academy of Forestry, which can promote root P absorption and plant growth. The active components of ABT rooting powder include indole-3-butytric acid and indole-3-acetic acid, which are regulators of plant growth (Zhang et al., 2010). Liu et al. (2002) found that the phosphate fertilizer utilization ratio of wheat increased by 12% when ABT rooting powder was used.

4. Discussion

4.1. Practical applications

A variety of different soil P activators have been reported to activate legacy P over the last few decades. These include PSMs, phosphatase enzymes, enzyme activators, low molecular weight organic acids, humic acids, lignin, crop residues, manure, biochar, zeolites, fly and wood ash, water-dissolved organic polymers, bentonite, mugineic acid and ABT rooting powder. Each P activator can act in one or more ways to promote the solubilization of insoluble P\textsubscript{i} and or the mineralization of P\textsubscript{o}. The range of mechanisms is illustrated in Fig. 7. The most common activation pathways are: (1) Dissolution. P activators change the chemical structure of the sorbing components – principally via metal complexation and removal; (2) Competitive inhibition of P sorption. P activators compete with phosphate for sorption sites or compete with metal oxides for exchange sites in soil; (3) Organic ligands. Functional groups such as carboxyl and phenolic hydroxyl can complex metal ions and release P bound to these cations; (4) Enzymolysis. P activators catalyze the hydrolysis of ester-phosphate bonds, leading to the release of phosphate; (5) P release. Bio-resource P activators contain both P\textsubscript{i} and P\textsubscript{o} which can be released into the soil.

Each type of P activator has its own specific advantages and disadvantages (Table 5), so activator selection should be tailored to the specific context in which the activator is to be used. Key factors to be considered include: (1) Specific optimum conditions, including soil physico-chemical properties. This is particularly the case for PSMs; (2) the nature of the legacy P pool targeted (P\textsubscript{i} or P\textsubscript{o} or both); (3) the cost and access to the P activators under consideration; (4) the influence of the P activator on crops, the prevailing microbial community and soil physical and chemical characteristics.

All types of bio-resource P activator, such as crop residues, manure and biochar, have both primary and secondary P supply functions. They can be usefully separated according to their form and principal mode of action: i.e. direct inorganic nutrient supply for direct uptake and the input of organic nutrients which are subsequently mineralized. They may also contain microbial populations (e.g. rhizobacteria) which can affect secondary nutrient (P) supply and other plant growth promotion functions. P-solubilization in the field tends to be much more difficult to demonstrate than solubilization of P in the laboratory. Crops respond differently to P activator amendments and are dependent on several factors such as soil temperature, moisture content, pH, salinity, organic matter content, microorganisms, the rhizosphere...
effect and soil fauna. The effects of these factors on the role of P activators need to be studied more extensively at the field scale before P activators can be used reliably.

4.2. Environmental considerations

Making full use of legacy P in soil is an environmentally desirable and economically feasible strategy for improving crop production, particularly in P-deficient soils. Agricultural P loss is a global concern due to the effects of P in receiving water bodies (eutrophication). By reducing the need for P fertilizer addition and enabling crops to efficiently access legacy P, the transfer of P from land to water can be reduced. There is now increasing evidence that P availability can be controlled by P activators. However, increasing P availability using P activators in soil can also increase the risk of P losses to surface and ground waters via dissolved and colloid-facilitated transport (Hens and Merckx, 2001).

Some P activators can contain potential toxicants, such as heavy metals (As, Cr, Zn, Cd). For example, ashes originating from co-firing of bark or wood with tyres can contain high levels of Zn (1%) (Someshwar, 1996). Such sources should be avoided (or at least rates should be controlled) in order to avoid contamination of both soil and receiving water bodies after P activator addition.

Although generally regarded as environmentally-friendly, P activators such as PSMs may also have some negative impacts such as triggering plant diseases or problems for native organisms (i.e. they can potentially act as biological invasions in soil: Amsellem et al., 2017). That said, they may also act to curb invasions of other invasive species (i.e. they can act as a biological control) (Santini et al., 2013).

4.3. Future research directions

Despite the fact that P activators have been shown to increase available P, precise mechanisms remain uncertain in most cases and additional insight is required. For example, further elucidation of the relationships between organic P activators and P in competitive sorption processes is necessary. Similarly, the available literature regarding the effects of biochar on P availability is inconsistent and mechanistic understanding of biochar-P interactions remains poor. Moreover, the response of plants to soil P activators in the field is often difficult to predict. Future research is, therefore, required to identify underlying biochemical and physical processes, which alter the bioavailability of legacy soil P.

Application of single activators may be insufficient to activate the broad range of P forms typically present in soil and the use of several activators together may produce better results. For instance, some studies have reported that the addition of citric acid and phosphatase together can be beneficial, although the exact mechanism by which additional P is released in this case is uncertain. Further research is required to uncover the mechanisms underpinning the compound effects of different P activators used in concert. Specifically, long-term field experiments should be conducted to address the interactions of P activators with natural soil and their impacts on crop yield.

Even though, various amendments have been reported for managing legacy P in different soils, some
of these can be costly and or practically difficult to implement. Although P-solubilizing microorganisms have yet to fulfill their promise as commercial bio-inoculants, scientists need to address certain issues, like how to improve the efficacy of bio-fertilizers, what the optimal delivery system should be, how to stabilize these microbes in soil systems, and how nutritional and root exudation aspects can be controlled in order to get maximum benefits from PSM application. Finally, research is also required to discover and enable efficient, feasible, cheap and environmentally-friendly solutions for improving access to legacy P stocks in soil.

5. Conclusion

P is a major macronutrient needed for plant growth and development. It has been historically oversupplied as phosphate fertilizers in many agricultural systems to increase crop yields. However, there are a number of adverse environmental impacts associated with the use of inorganic P fertilizers. These include increasing risk of P loss from soils with elevated P concentrations (so-called legacy P), which may lead to eutrophication in water bodies receiving phosphate in agricultural runoff (diffuse-source pollution), and the depletion of finite resources of high grade phosphate rocks with associated price increases in the context of ever-increasing global demands for food and bioenergy. A more sustainable approach to managing P availability in agriculture is, therefore, required.

Legacy P that has accumulated in soils represents a significant potential secondary P source that could substitute (at least in part) manufactured fertilizers, if ways can be found for unlocking it (P activation). A number of P activators have been proposed which have been reviewed here. However, soil P cycling is extremely complex and varies with soil morphogenesis, mineralogy, organic matter content, pH, water relations, etc. Taking full advantage of legacy P, therefore, remains a significant challenge. Complicating influences include tillage, residue management and fertilization practices, as well as the effects of interactions between soil microbes, soil structure and organic matter on operational soil P fractions.

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No color is needed for any figures in print.
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Table 1
Common phosphorus minerals found in acid, neutral and calcareous soils and their equilibrium dissolution reactions. Modified from Gustafsson et al. (2012) and Yadav and Verma (2012).

<table>
<thead>
<tr>
<th>Soil type</th>
<th>Minerals</th>
<th>Chemical formula</th>
<th>Equilibrium dissolution reaction</th>
</tr>
</thead>
<tbody>
<tr>
<td>Acid soils</td>
<td>Strengite</td>
<td>FePO$_4$·2 H$_2$O</td>
<td>FePO$_4$·2 H$_2$O(s) ↔ Fe$^{3+}$ + PO$_4$$^{3-}$ + 2H$_2$O</td>
</tr>
<tr>
<td></td>
<td>Vivianite</td>
<td>Fe$_3$(PO$_4$)$_2$·8H$_2$O</td>
<td>Fe$_3$(PO$_4$)$_2$·8 H$_2$O(s) ↔ 3Fe$^{2+}$ + 2PO$_4$$^{3-}$ + 8H$_2$O</td>
</tr>
<tr>
<td></td>
<td>Variscite</td>
<td>AlPO$_4$·2 H$_2$O</td>
<td>AlPO$_4$·2 H$_2$O(s) ↔ Al$^{3+}$ + PO$_4$$^{3-}$ + 2H$_2$O</td>
</tr>
<tr>
<td>Neutral and calcareous soils</td>
<td>B-tricalcium phosphate</td>
<td>Ca$_3$(PO$_4$)$_2$</td>
<td>Ca$_3$(PO$_4$)$_2$(s) ↔ 3Ca$^{2+}$ + 2PO$_4$$^{3-}$</td>
</tr>
<tr>
<td></td>
<td>Dicalcium phosphate</td>
<td>CaHPO$_4$</td>
<td>CaHPO$_4$(s) ↔ Ca$^{2+}$ + PO$_4$$^{3-}$ + H$^+$</td>
</tr>
<tr>
<td></td>
<td>Dicalcium phosphate dihydrate</td>
<td>CaHPO$_4$·2 H$_2$O</td>
<td>CaHPO$_4$·2 H$_2$O(s) ↔ Ca$^{2+}$ + PO$_4$$^{3-}$ + H$^+$ + 2H$_2$O</td>
</tr>
<tr>
<td></td>
<td>Fluorapatite</td>
<td>Ca$_5$(PO$_4$)$_3$ F</td>
<td>Ca$_5$(PO$_4$)$_3$ F(s) ↔ 5Ca$^{2+}$ + 3PO$_4$$^{3-}$ + F$^-$</td>
</tr>
<tr>
<td></td>
<td>Hydroxyapatite</td>
<td>Ca$_5$(PO$_4$)$_3$ OH</td>
<td>Ca$_5$(PO$_4$)$_3$ OH ↔ 5Ca$^{2+}$ + 3PO$_4$$^{3-}$ + OH$^-$</td>
</tr>
<tr>
<td></td>
<td>Octacalcium phosphate</td>
<td>Ca$_8$H$_2$(PO$_4$)$_6$·5 H$_2$O</td>
<td>Ca$_8$H$_2$(PO$_4$)$_6$·5 H$_2$O ↔ 8Ca$^{2+}$ + 6PO$_4$$^{3-}$ + 6H$_2$O</td>
</tr>
</tbody>
</table>
Fig. 1. The physicochemical and biological transformations of P in soil-plant and soil-solution systems (modified after Frossard et al., 2011). Po: orgainc P; Pi: inorganic P. The dotted arrow indicates external phosphorus input and the solid arrow shows the internal phosphorus transformations.
Table 2

Examples of common phosphate solubilizing microorganisms in agricultural soils (Behera et al., 2014; Bolan, 1991; Thingstrup et al., 2000).

<table>
<thead>
<tr>
<th>Microorganism</th>
<th>Biological taxa</th>
</tr>
</thead>
<tbody>
<tr>
<td>PSB</td>
<td>Achromobacter, Aerobacter, Agrobacterium, Alcaligenes, Arthrobacter, Aspergillus, Azotobacter, Bacillus, Bradyrhizobium, Burkholderia, Chromobacterium, Enterbacter, Erwinia, Escherichia, Flavobacterium, Klebsiella, Micrococcus, Pantoea agglomerans, Pseudomonas, Rhizobium, Salmonella, Serratia, Thiobacillus,</td>
</tr>
<tr>
<td>PSF</td>
<td>Alternaria, Arbuscular mycorrhiza, Aspergillus, Fusarium, Helminthosporium, Penicillium, Rhizopus, Sclerotium</td>
</tr>
<tr>
<td>PSA</td>
<td>Streptomyces, Nocardia</td>
</tr>
</tbody>
</table>

PSF: phosphate solubilizing fungi; PSB: phosphate solubilizing bacteria; PSA: phosphate solubilizing actinomycetes.

Fig. 2 Potential mechanisms for the solubilization of insoluble P by phosphate solubilizing microorganisms (PSMs). PSF: phosphate solubilizing fungi; PSB: phosphate solubilizing bacteria; PSA: phosphate solubilizing actinomycete; VAM: vesicular-arbuscular mycorrhizae (modified after Khan et al., 2010).
Table 3

PSM isolated from different soils and their effects on soil available phosphorus (AP) and the equivalent application rate of phosphate fertilizers (PF) when PSM and PSM fertilizers are applied.

PSMF: phosphate solubilizing microbial mixed fertilizers; RP: rock phosphate; PBF: phosphate solubilizing bacteria fertilizers; VAM: vesicular arbuscular mycorrhizae; PSB: phosphate solubilizing bacteria; PSF: phosphate solubilizing fungi; AMF: arbuscular mycorrhizal fungi; CGMCC: China General Microbiological Culture Collection Center of China Administration Committee for Culture Collection of Microorganisms.

<table>
<thead>
<tr>
<th>Microorganism</th>
<th>Strains</th>
<th>Isolated area</th>
<th>Experiment soil</th>
<th>Crop</th>
<th>P activation response on soil</th>
<th>References</th>
</tr>
</thead>
<tbody>
<tr>
<td>PSB</td>
<td>Enterobacter agglomerans</td>
<td>Purchased</td>
<td>Silt loam (pH=5.6), Mexico</td>
<td>Tomato</td>
<td>P concentration increased by 10.5-31.6%; mixed higher</td>
<td>(Kim et al., 1997)</td>
</tr>
<tr>
<td></td>
<td><em>Glomus etunicatum</em></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td><em>Fusarium oxysporum</em></td>
<td>Obtained from the culture collection of University of Mysore, India</td>
<td></td>
<td>Sandy loam soil (pH=6.9), India</td>
<td>Tomato</td>
<td>AP increased by 26.7-183.3%</td>
<td>(Hariprasad and Niranjana, 2009)</td>
</tr>
<tr>
<td><em>Alternaria solani</em></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td><em>Pseudomonas chlororaphis</em></td>
<td>Walnut production areas from Sichuan, China</td>
<td></td>
<td>Sandy loam soil (pH=7.1), China</td>
<td>Walnut</td>
<td>AP increased by 9.7-50.7%; mixed higher; <em>B. cereus</em> had no effect</td>
<td>(Yu et al., 2011)</td>
</tr>
<tr>
<td><em>Bacillus cereus</em></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
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<tr>
<td><em>Pseudomonas fluorescens</em></td>
<td></td>
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<td></td>
<td></td>
</tr>
<tr>
<td><em>Pseudomonas synxantha</em></td>
<td>Rhizosphere of <em>Aloe barbadensis</em> from Indian</td>
<td></td>
<td>Unsterile loamy soil (pH=7.8), India</td>
<td><em>Aloe barbadensis</em></td>
<td>AP increased by 230-403%</td>
<td>(Gupta et al., 2012)</td>
</tr>
<tr>
<td><em>Burkholderia gladioli</em></td>
<td></td>
<td></td>
<td></td>
<td></td>
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<tr>
<td><em>Enterobacter hormaechei</em></td>
<td></td>
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<tr>
<td><em>Serratia marcescens</em></td>
<td></td>
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</tr>
<tr>
<td><em>Ocimum basilicum</em></td>
<td>Sodic soil sites from Sultanpur, India</td>
<td></td>
<td>Sodic soils (pH=9.3), India</td>
<td>Sweet basil (<em>O. basilicum</em>)</td>
<td>Residual fertility P conversion rate increased by 17.2-32.1%</td>
<td>(Sahay and Patra, 2014)</td>
</tr>
<tr>
<td>PSF</td>
<td><em>Penicillium oxalicum</em></td>
<td>Mine landfills of RP from Udaipur, India</td>
<td>Sandy loam soil (pH=8.1), India</td>
<td>Wheat and maize</td>
<td>AP increased by 35.1-85.3% for wheat and 106.3-127.7% for maize</td>
<td>(Singh and Reddy, 2011)</td>
</tr>
<tr>
<td><em>Candida tropicalis</em></td>
<td>Crops rhizospheric soil in Birbhum, West Bengal, India.</td>
<td></td>
<td>Silty clay soil (pH=5.2), India</td>
<td>Maize</td>
<td>PF application rate reduced by 34.9-467.4%</td>
<td>(Mukherjee and Sen, 2015)</td>
</tr>
<tr>
<td>PSB &amp; VAM</td>
<td>Aspergillus niger</td>
<td>Plant rhizosphere in soils from Ismailia and South Sinai Governorates</td>
<td>Sandy soil (pH=8.4), Egypt</td>
<td>Wheat and faba bean</td>
<td>AP increased by 12.1-194.2% for wheat and 15.9-208.3% for faba bean</td>
<td>(Abdul Wahid and Mehana, 2000)</td>
</tr>
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<tr>
<td>PSB &amp; PSF</td>
<td>Pantoea cypripedii</td>
<td>Rhizospheric soil of Stevia rebaudiana from Pojewa, India</td>
<td>Sandy loam (pH=8.3), India</td>
<td>Maize-wheat Cropping</td>
<td>P absorption increased by 41.2-117.6% for maize and 41.05-41.8% for wheat</td>
<td>(Kaur and Reddy, 2015)</td>
</tr>
<tr>
<td>PSB &amp; PSF</td>
<td>Bacillus subtilis QST713</td>
<td>Purchased Siliceous and calcareous growing media (pH=7.8)</td>
<td>Siliceous and calcareous growing media (pH=7.8)</td>
<td>Cucumber</td>
<td>AP increased by 16.8% (Fungi T34) and 39.85 (Bacteria QST713) in siliceous soil; 6.6% (T34) and 60.7% (QST713) in calcareous soil</td>
<td>(García-López et al., 2016)</td>
</tr>
<tr>
<td>PSB &amp; AMF</td>
<td>Mortierella sp. Claroideoglomus claroideum</td>
<td>Andisol from Hawai‘i, America</td>
<td>Sandy loam (pH=5.4), Clay loam (pH=4.9), America</td>
<td>Leucaena leucocephala</td>
<td>Varied from soil type; Mixed higher</td>
<td>(Osorio and Habte, 2015)</td>
</tr>
<tr>
<td>PBF</td>
<td>Penicillium sp. Aspergillus foetidus</td>
<td>Rhizospheric soil of tomato, eggplant and cucumber, Iraq</td>
<td>Silt clay soil (pH=8.2), Iraq</td>
<td>Typical torrifluvent</td>
<td>AP increased by 24.2-138.6%</td>
<td>(Salih et al., 1989)</td>
</tr>
<tr>
<td>AB-233</td>
<td>Purchased Loamy soil, (pH=7.4), China</td>
<td>Soybean</td>
<td>AP increased by 11.8-57.1%</td>
<td>(Zhou et al., 2005)</td>
<td></td>
<td></td>
</tr>
<tr>
<td>PSMF</td>
<td>Pseudomonas Rahnella Aspergillus niger Penicillium</td>
<td>Purchased Clay soil (pH=8.1), China</td>
<td>Rape</td>
<td>AP increased by 1.84-15.44%; Mix PSB &gt; mixed PSF, both higher than mixed PSB with PSF</td>
<td>(Li et al., 2014)</td>
<td></td>
</tr>
<tr>
<td>PSB &amp; PSMF</td>
<td>Bacillus subtilis Bacillus licheniformis</td>
<td>Depository in CGMCC</td>
<td>Silt clay soil (pH was not)</td>
<td>Corn Rice</td>
<td>P absorption increased by 25.93-50.56% for corn and</td>
<td>(Ho et al., 2013)</td>
</tr>
<tr>
<td>Strep. celluloseae</td>
<td>Asp. versicolor</td>
<td>India</td>
<td>18.58-44.09% for rice</td>
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<tr>
<td>PSF &amp; PSMF</td>
<td>B. megatherium var</td>
<td>Purchased</td>
<td>Clay soil (pH was not mentioned), India</td>
<td></td>
<td></td>
<td></td>
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<tr>
<td></td>
<td>P. Phosphaticum</td>
<td>Sugarcane</td>
<td>PF application rate reduced by 25%</td>
<td></td>
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<td></td>
</tr>
</tbody>
</table>

(Sundara et al., 2002)
Fig. 3. (a) Phosphate regeneration through enzymatic decomposition of various P compounds in soil (modified after Burns & Dick, 2002); (b) The role of phosphatase enzymes in mineralizing P to phosphate and enzyme activators activating phosphatase enzymes.
The main low molecular weight organic acids identified in agricultural soils (Gerke, 2015; Jones, 1998; Song and Cui, 2003; Strobel, 2001; van Hees et al., 1999).

<table>
<thead>
<tr>
<th>Classification</th>
<th>Carboxylate</th>
<th>Organic acid</th>
<th>Molecular formula</th>
<th>Molecular mass</th>
<th>Organic acid</th>
<th>Molecular formula</th>
<th>Molecular mass</th>
</tr>
</thead>
<tbody>
<tr>
<td>Aliphatic</td>
<td>Monobasic</td>
<td>Formic</td>
<td>HCOOH</td>
<td>46.03</td>
<td>Lauric</td>
<td>C₁₀H₂₀COOH</td>
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<tr>
<td></td>
<td></td>
<td>Acetic</td>
<td>CH₃COOH</td>
<td>60.05</td>
<td>Palmitic</td>
<td>C₁₅H₃₁COOH</td>
<td>256.42</td>
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<td></td>
<td></td>
<td>Propionic</td>
<td>C₂H₅COOH</td>
<td>74.08</td>
<td>Linolenic</td>
<td>C₁₇H₃₃COOH</td>
<td>278.44</td>
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<tr>
<td></td>
<td></td>
<td>Butyric</td>
<td>C₃H₇COOH</td>
<td>88.11</td>
<td>Linoleic</td>
<td>C₁₇H₃₃COOH</td>
<td>282.47</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Valeric</td>
<td>C₄H₉COOH</td>
<td>102.14</td>
<td>Oleic</td>
<td>C₁₇H₃₃COOH</td>
<td>282.47</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Pyruvate</td>
<td>CH₂COCOOH</td>
<td>88.06</td>
<td>Arachidic</td>
<td>C₁₉H₃₉COOH</td>
<td>312.53</td>
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<tr>
<td></td>
<td></td>
<td>Glycolic</td>
<td>CH₂OHCOOH</td>
<td>76.05</td>
<td>Carnaubic</td>
<td>C₂₀H₄₀COOH</td>
<td>368.62</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Lactic</td>
<td>CH₃CH(OH)COOH</td>
<td>90.08</td>
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<tr>
<td>Dicarboxylic</td>
<td></td>
<td>Oxalic</td>
<td>HOOC-COOH</td>
<td>90.04</td>
<td>Maleic</td>
<td>C₄H₄O₄</td>
<td>116.08</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Malonic</td>
<td>HOOCCH₂-COOH</td>
<td>104.06</td>
<td>Fumaric</td>
<td>C₄H₄O₄</td>
<td>116.08</td>
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<tr>
<td></td>
<td></td>
<td>Succinic</td>
<td>HOOC(CH₂)-COOH</td>
<td>118.09</td>
<td>Ketoglutaric</td>
<td>C₅H₈O₄</td>
<td>146.11</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Tartaric</td>
<td>HOOC(CHOH)₂-COOH</td>
<td>150.09</td>
<td>Aconitic</td>
<td>C₆H₆O₆</td>
<td>174.11</td>
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<tr>
<td></td>
<td></td>
<td>Malic</td>
<td>HOOCCH(OH)CH₂-COOH</td>
<td>134.09</td>
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<tr>
<td>Tribasic</td>
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<td>Oxalacetic</td>
<td>C₆H₄O₅</td>
<td>132.07</td>
<td>Citric</td>
<td>C₆H₈O₇</td>
<td>192.43</td>
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<tr>
<td>Aromatic</td>
<td>Monobasic</td>
<td>Benzoic</td>
<td>C₆H₅-COOH</td>
<td>122.12</td>
<td>Sinapic</td>
<td>C₁₁H₁₂O₅</td>
<td>224.22</td>
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<tr>
<td></td>
<td></td>
<td>Phenylacetic</td>
<td>C₆H₅-CH₃-COOH</td>
<td>136.15</td>
<td>Phthalic</td>
<td>C₄H₄O₄</td>
<td>166.13</td>
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<tr>
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<td></td>
<td>Cinnamic</td>
<td>C₆H₅CH=CH-COOH</td>
<td>148.16</td>
<td>P-coumaric</td>
<td>C₆H₄O₃</td>
<td>164.16</td>
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<td>Ferulic</td>
<td>C₁₀H₁₀O₄</td>
<td>194.19</td>
<td>Salicylic</td>
<td>C₇H₆O₃</td>
<td>138.12</td>
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<tr>
<td></td>
<td></td>
<td>2-Naphthoic</td>
<td>C₁₁H₈O₂</td>
<td>172.18</td>
<td>Gallic</td>
<td>C₇H₆O₃</td>
<td>170.10</td>
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<td>Indole-3-acetic</td>
<td>C₁₀H₈O₂</td>
<td>175.18</td>
<td>Vanillic</td>
<td>C₉H₈O₄</td>
<td>168.15</td>
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<tr>
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<td></td>
<td>P-phenylcinnamic</td>
<td>C₁₁H₁₂O₂</td>
<td>224.25</td>
<td>Syringic</td>
<td>C₇H₁₀O₅</td>
<td>198.18</td>
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<tr>
<td></td>
<td></td>
<td>P-hydroxybenzoic</td>
<td>C₇H₈O₃</td>
<td>138.12</td>
<td>Shikimic</td>
<td>C₇H₁₀O₅</td>
<td>174.16</td>
</tr>
<tr>
<td></td>
<td></td>
<td>3-Phenylpropanoic</td>
<td>C₉H₁₀O₂</td>
<td>150.17</td>
<td></td>
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</tr>
</tbody>
</table>

Amino acid | NH₂-CH(R)-COOH*  

*R indicates R-groups in amino acids. The R-groups give the amino acids their physical-chemical characteristics such as polarity (hydrophobic or hydrophilic) and charge (acidic or positive versus basic or negative).
Table 5
The type and concentration of low molecular weight organic acids added to different types of agricultural soil and their effects on P transformation.

<table>
<thead>
<tr>
<th>LMWOAs</th>
<th>Concentration</th>
<th>Soil type</th>
<th>Soil area</th>
<th>Effects</th>
<th>References</th>
</tr>
</thead>
<tbody>
<tr>
<td>Citrate and Tartrate acid</td>
<td>0.1-0.5 mmol L⁻¹</td>
<td>Black loam and gravelly clay soil</td>
<td>Egmont, Okaihau in New Zealand</td>
<td>Fe and Al sorb P; citrate (50% reduction), tartrate (20% reduction).</td>
<td>(Earl et al., 1979)</td>
</tr>
<tr>
<td>Acetic, formic, lactic, citric, malic, oxalic, tartaric acid</td>
<td>10 mmol kg⁻¹</td>
<td>Silt loam derived from volcanic ash and clay loam from loess</td>
<td>Patua, Italy; Tokomaru, New Zealand</td>
<td>Decrease the adsorption of P by soils: tricarboxylic acid &gt; dicarboxylic acid &gt; monocarboxylic acid.</td>
<td>(Bolan et al., 1994)</td>
</tr>
<tr>
<td>Acetic acid</td>
<td>0.2, 2 mmol kg⁻¹</td>
<td>Arid soil</td>
<td>Rajasthan, India</td>
<td>No effect of acetic acid application alone on the Olsen-P. Having an effect when applied along with fertilizers P.</td>
<td>(Tripathi, 2005)</td>
</tr>
<tr>
<td>Oxalic, citric acid</td>
<td>10 mmol kg⁻¹</td>
<td>Plantation and natural forest soils</td>
<td>Subtropical and tropical areas in Australia</td>
<td>Solubilizing of P; citric acid, 34.7%, malic and oxalic, no significant differences.</td>
<td>(Wei et al., 2010)</td>
</tr>
<tr>
<td>Malic acid</td>
<td></td>
<td>Vegetable, garlic, and potato land use calcareous soils</td>
<td>Hamedan, Iran</td>
<td>Malic and citric acids had no effect on Fe-Al oxide fraction, but oxalic acid significantly reduced this fraction.</td>
<td>(Taghipour and Jalali, 2013)</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Calcareous clay soil, neutral silty clay loam, and acidic silty clay soil</td>
<td>Luancheng (wheat), Shenyang (corn), and Taoyuan (rice), China</td>
<td>Pₗ release induced by low molecular weight organic acids is ascribed to their ability to mobilize NaHCO₃-Pₗ rather than to chelate cations (i.e., Fe³⁺, Al³⁺) bound to Pₗ in soil.</td>
<td>(Wang et al., 2015)</td>
</tr>
<tr>
<td>α-naphthaleneacetic acid</td>
<td>10⁻⁴, 10⁻⁵, 10⁻⁶ mmol L⁻¹</td>
<td>Nutrient solution</td>
<td>Spain</td>
<td>The secretion of common organic acid increased by proteoid roots of white lupin.</td>
<td>(Gomez and Carpena, 2014)</td>
</tr>
</tbody>
</table>
Fig. 4. (a) Water-soluble P concentrations 1.9 cm to the side of a mono-ammonium phosphate (MAP) fertilizer band, uncoated (control) or coated with a humic acid (HA) product (HA 1 or HA 2). The pre-fertilization sample (pre) was collected less than 1 h prior to fertilization; (b) Water-soluble P concentrations 3.8 cm to the side of a MAP band; (c) Water-soluble P concentrations 3.8 cm below a MAP band. *Significant difference from control p<0.05) (Jones et al., 2007, original images with permission from Taylor & Francis Group).
**Fig. 5.** Schematic illustration of the recycling of phosphorus between agricultural residues and soil, including the application of biochar. Pᵢ is inorganic phosphorus, Pₒ is organic phosphorus.

**Fig. 6** Schematic summary of different approaches to modify zeolite.
Fig. 7. Illustration of different mechanisms involved in the solubilization and mineralization of insoluble P by P activators in soils. (A) Enlarged drawing of root; (B) Metal ions and metal oxides which easily form P-O-M compounds in soil. (a) Effect of soil pH on sparingly soluble P (inorganic phosphorus) minerals; (b) Formation of complexes with (B) to release phosphate; (c) Competition for sorption sites with phosphate in soil; (d) Adsorption of metal and metal oxides to release phosphate; (e) Sheltering sorption sites in soil; (f) Hydrolysis of Po (organic phosphorus) and Pi; (g) Strengthening of phosphatase enzyme activity; (h) Promotion of root growth; (i) Biochar increasing mycorrhizal-plant associations; (j) Exchange-induced dissolution; (k) Production of organic acids by microbial processes; (m) Po & Pi retention and release. ABT rooting powder: have the ability to promote rooting and to increase the survival rate of plants under abiotic stress.
Table 6
Summary of the advantages and disadvantages of the P activators described in this review.

<table>
<thead>
<tr>
<th>P activators</th>
<th>Advantages</th>
<th>Disadvantages</th>
</tr>
</thead>
<tbody>
<tr>
<td>Phosphate solubilizing microorganisms</td>
<td>Economically feasible; Activate both P&lt;sub&gt;0&lt;/sub&gt; and P&lt;sub&gt;i&lt;/sub&gt;; High activation effect</td>
<td>Complex screening and domesticating process; Easily affected by environment; Time consuming; Potential invasions</td>
</tr>
<tr>
<td>Phosphatase enzyme and enzyme activators</td>
<td>Mineralize complex compounds P&lt;sub&gt;0&lt;/sub&gt;</td>
<td>High specificity; Easy to be inactive</td>
</tr>
<tr>
<td>Low molecular weight organic acid</td>
<td>Fast responding speed; High activation effect; Amend soil salinization</td>
<td>Cause soil acidification</td>
</tr>
<tr>
<td>Humic acid and lignin</td>
<td>Increase soil organic matter; Improve soil conditions</td>
<td>May sorb phosphate itself</td>
</tr>
<tr>
<td>Crop residue and manure</td>
<td>Biodegradable, abundant resources found in nature; Low processing costs; Release both P&lt;sub&gt;0&lt;/sub&gt; and P&lt;sub&gt;i&lt;/sub&gt;; Secondary nutrient supply</td>
<td>Bring soil pests; Contain microbial populations; Easily lost in runoff</td>
</tr>
<tr>
<td>Biochar</td>
<td>Improve soil conditions; Release P&lt;sub&gt;0&lt;/sub&gt; and P&lt;sub&gt;i&lt;/sub&gt;; Good for microorganisms</td>
<td>Preparation process is complex; Mixed reports on activation effect</td>
</tr>
<tr>
<td>Zeolites and others</td>
<td>Reuse and recycle waste resources</td>
<td>Seldom used in soil; Non-indigenous in soil; May bring heavy metal to soil</td>
</tr>
</tbody>
</table>