Chapter 6 Role of Mycorrhizal Symbioses in Phosphorus Cycling

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6.1 Introduction

6.1.1 Mycorrhizal Symbiosis: Definition, Partners, Diversity

Mycorrhizal symbioses are associations of plant roots or rhizoids with fungi that, at least under some conditions, are beneficial to both partners. Arbuscular mycorrhizal (AM) symbiosis was established at the dawn of terrestrial plant evolution, some 400–500 million years ago, between ancestral vascular plants (*Cooksonia, Rhynia, Aglaophyton*) and fungi belonging to the phylum Glomeromycota (Pirozynski and Dalpé 1989; Redecker et al. 2000; Schüßler et al. 2001). AM symbiosis has been identified in thousands of plant species among all major plant lineages including bryophytes, ferns, gymno- and angiosperms (Brundrett 2009; Wang and Qiu 2006). It is the most widespread type of mycorrhizal symbiosis with respect to the number of plant species it involves (Trappe 1987; Wang and Qiu 2006) and can be found in virtually all ecosystems on Earth. Despite its broad host range, the fungal diversity is limited to a few hundred species (Redecker and Raab 2006), inferring that the

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association is particularly species-unspecific. Quantitative preferences for root colonization have, however, been repeatedly found between some plant and the fungal species (Jansa et al. 2002; Öpik et al. 2009; Sýkorová et al. 2007; Vanden-koornhuyse et al. 2003) and it is possible that a broader specificity between some fungal and plant genotypes does exist. This remains difficult to test experimentally, given the current limitations in the understanding of AM fungal genomic organization and molecular diversity (Croll and Sanders 2009; Martin et al. 2008).

Evolutionarily more recent types of mycorrhizal associations include ectomycorrhizas (ECM), ericoid mycorrhizas (ERM), and orchid mycorrhizas (Cairney 2000). These types are restricted to narrower groups of plant taxa, and involve fungi from the phyla Basidiomycota, Ascomycota, and Zygomycota. Some of these fungi form quite species-specific associations (e.g., Cortinarius and Suillus associate with rather a narrow range of plant species, such as Pseudotsuga, Betula, Larix, and *Pinus*), whereas others (e.g., *Cenococcum* spp.) may colonize roots of a broad range of plant species (Bruns et al. 2002; Smith and Read 2008; Tedersoo et al. 2008). A few plant species (such as eucalyptuses, willows, and alders) have the capacity to interact with both AM and ECM fungi. This can give rise to root systems simultaneously colonized by mycorrhizal fungi belonging to different types (Adams et al. 2006; van der Heijden and Vosátka 1999). The different types of mycorrhizal symbioses have also shown predominance for different plant biomes (Read and Perez-Moreno 2003; Smith and Read 2008 and references therein). Plants in polar regions and at high altitudes are usually not mycorrhizal, but often have roots extensively colonized by dark septate fungal endophytes with unclear function (Mandyam and Jumpponen 2005; Newsham et al. 2009). By definition, endophytes live solely within the plant tissues, and often fulfill their entire lifecycle including reproduction inside the plant body (Faeth and Fagan 2002). Mycorrhizal fungi, in contrast, inhabit and interconnect two kinds of environment, namely the inner volume of the plant roots and the surrounding soil (Jansa and Gryndler 2010). The status of many root-inhabiting fungi is unclear because it is inherently difficult to demonstrate absence of hyphal growth outside the roots. Heathlands (here we refer to heathlands at both high latitudes and altitudes as well as nutrient-limited and wildfire-prone ecosystems in Mediterranean climates, known as fynbos, chaparral, maquis or matorral in different parts of the world) are usually dominated by ERM, and Taiga (coniferous boreal forest) is dominated by the ECM (Read et al. 2004; Thormann et al. 1999). Most trees in deciduous forests establish ECM symbiosis, although the understorey plants are primarily colonized by AM fungi (Helgason et al. 2002). Most plants in grasslands and in tropical forests also establish AM symbiosis (Castillo et al. 2006; Treseder and Cross 2006). Plants inhabiting highly weathered and severely phosphorus (P)-impoverished soils (as in Western Australia) are often devoid of mycorrhizas; these plants have other adaptations, such as cluster roots, that fulfill their P requirements (Lambers et al. 2008). Under field conditions, the root system of a single plant is usually colonized by different mycorrhizal fungal species simultaneously (Burke et al. 2005; Jansa et al. 2003b; Merryweather and Fitter 1998; Miller et al. 1991). This diversity could have important consequences for acquisition of nutrients as well as for maximizing symbiotic benefits for the plants (Baxter and Dighton 2001; Jansa et al. 2008; Koide 2000).

6.1.2 Mycorrhizal Functioning

The unequivocal importance of mycorrhizal symbioses to plants (and soils) is inherently difficult to demonstrate because the mycorrhizal condition is normal, and absence is rare under natural settings (Merryweather and Fitter 1995). Establishing non-mycorrhizal control treatments in pot experiments or in the field is a great challenge and may potentially introduce experimental artefacts (Jones and Smith 2004; Kahiluoto et al. 2000a). Soil sterilization by steaming or autoclaving, for example, changes chemical soil properties (Serrasolses et al. 2008). It also eliminates other soil organisms, introducing further confounding effects. While these artefacts should always be considered in the interpretation of experimental results, there have been numerous independent studies comparing the performance of mycorrhizal and non-mycorrhizal plants under a range of environmental conditions. These experiments have collectively demonstrated that plants benefit from their mycorrhizal associations through improved nutrient acquisition, mainly of elements with low mobility in the soil (e.g., P, zinc, and copper), and through greater resistance to drought and biotic stresses (Clark and Zeto 2000; Jansa et al. 2003a; Marschner 1995; Redon et al. 2009; Sikes et al. 2009).

The benefits of mycorrhizal colonization to the plants result from expansion and/ or complementation of the root function. Mycorrhizal fungi colonize two environments: the inner root volume (sometimes extending to the hyphal sheath on the root surface) and the surrounding soil, thus directly connecting the root system with a greater soil volume. By increasing soil contact, the plants are able to acquire resources from zones lying far beyond the direct reach of the roots and the root hairs. This effect is not trivial because the hyphae of some AM fungi can extend many centimeters away from the root surface, unlike root hairs, which only extend a few millimeters (Jakobsen et al. 1992; Jansa et al. 2003a, 2005). ECM fungal mycelium can bridge even greater distances, particularly if the fungi form thick rhizomorphs (Allen et al. 2003). P transport through the ECM hyphae over distances up to 40 cm has been documented (Finlay and Read 1986; Timonen et al. 1996). In contrast, some mycorrhizal fungi establish very dense mycelial networks in the vicinity of the roots, growing only a few millimeters from their host (Smith et al. 2004). These mycelial networks significantly expand the capacity of the plant to acquire mineral nutrients from the soil. In addition, in the case of ECM fungi, these mycelial networks also play an important role in acquisition of water by the plants from the soil (Allen 2007; George and Marschner 1996). Root colonization by mycorrhizal fungi and their mycelial networks may also be important for interactions between plants and soil-borne pathogens. This is due to (1) changes in plant nutritional status upon mycorrhiza development, (2) direct competition for plant carbohydrates between the mycorrhizal fungi and the pathogens, and/or (3) changes in the activity and composition of the microbial communities in the rhizosphere (Fitter 2005; Graham 2001; Newsham et al. 1995; Toljander et al. 2007).

Many ERM and ECM fungi are also thought to be involved in mineralization of organic nutrients (Bueé et al. 2007; Finlay 2008; Tibbett and Sanders 2002) and bioweathering of recalcitrant inorganic nutrients from carbonates, micas, and apatites (Blum et al. 2002; Wallander 2000). It appears, however, that even in some of the well-documented case studies, other soil microorganisms (particularly the prokaryotes) might have played important roles in the release of nutrients from minerals or organic compounds. Therefore, the contribution of mycorrhizal fungi to these processes remains, in most cases, poorly quantified (Finlay 2008; Koele et al. 2009). Evidence for direct involvement of AM fungi in mineralization of significant amounts of organic P is still inconsistent. In particular, the role of other soil microorganisms associated with the mycorrhizal mycelium, which might be very important under nonsterile soil conditions, is not properly understood (Finlay 2008; Joner and Jakobsen 1995; Joner and Johansen 2000; Koide and Kabir 2000; Tarafdar and Marschner 1994; Toljander et al. 2006). The presence of mycorrhizal fungi also significantly modifies soil conditions in the rooting zone (aggregation, wettability, biological activity), modulates intra- and interspecific competition within the plant community, and affects soil microbial communities (Barea 2000; Facelli et al. 1999; Johansson et al. 2004; Rillig and Mummey 2006).

Mycorrhizal fungi are invariably heterotrophic organisms, and they mostly derive the organic carbon needed for their growth, respiration, and biological maintenance directly from their host plants in the form of recently fixed photosynthates. Some of the fungi (especially those forming ECM and ERM associations) are currently considered capable of limited saprophytic growth (Azcón-Aguilar and Barea 1995; Gibson and Mitchell 2004; Koide et al. 2008; Zeller et al. 2008), though under natural conditions its widespread realization has recently been questioned (Baldrian 2009; Taylor and Alexander 2005). The dependence of the fungi on the host plant for organic carbon can, depending on environmental conditions (e.g., availability of soil P, availability of light and CO₂ to the plants, plant density, and possibly other factors), cause the association to vary from highly beneficial to apparently parasitic (Johnson et al. 1997; Schroeder and Janos 2004; Smith and Smith 1996; West et al. 1993; Whitbeck 2001). For example, under very low soil P availability ($<0.2 \text{ mg P kg}^{-1}$, extractable with 0.5 M NaHCO₃) caused either by low total P levels or high P sorption in highly weathered soils, mycorrhizal symbiosis may be less beneficial in terms of net P acquisition than if small amounts of P were added to the soil (Bolan 1991; Bolan et al. 1983). Therefore, under very low soil P availability, plants specialized in modes of P mobilization and uptake other than mycorrhizal symbiosis (e.g., those forming cluster roots) can dominate the plant communities (Lambers et al. 2008). In contrast, under conditions of very high soil P availability (>50 mg kg⁻¹, extractable with 0.5 M NaHCO₃), such as following application of high rates of water-soluble P-fertilizers, the plant can gain access to enough P with its root system, resulting in little or no net demand for mycorrhizal P uptake (Kahiluoto et al. 2000b; Sorensen et al. 2003). In such situations, the extent of mycorrhizal development in roots is often reduced (Bolan et al. 1984; Jansa et al. 2009; Youpensuk et al. 2008). In addition, depending on the plant species and soil and climatic conditions, the association with AM fungi could have negative effects on plant growth (Jifon et al. 2002; Morgan et al. 2005; Ryan and Graham 2002; Kahiluoto et al. 2000b). Under conditions where P availability limits plant growth (NaHCO₃-extractable P levels typically between 5 and 20 mg kg^{-1}), mineral nutrient acquisition benefits conferred to the plants by the AM fungi will usually far outweigh the costs of carbon supply to the fungus and will result in a net growth benefit to the plant (Jakobsen 1995; Li et al. 2005; Morgan et al. 2005; Ortas et al. 2002). These studies indicate that the benefits to the plants of association with AM fungi across a range of environmental conditions (Fig. 6.1) are best described by a bell-shaped (or unimodal) response curve (Bolan et al. 1984; Gange and Ayres 1999; Picone 2002). Even so, plants may benefit from mycorrhizal symbiosis even though the growth and/or nutritional benefits (such as net P uptake) may not be apparent, and special techniques such as isotopic labeling are necessary to demonstrate mycorrhizal function in these cases (Fitter 2005; Grace et al. 2009; Smith et al. 2004, 2009).

Although negative effects of some ECM fungi on host plant growth have been occasionally reported (Burgess et al. 1993; Corrêa et al. 2008), the great majority of



Fig. 6.1 Conceptual model of whole-plant effects of the interactions between a plant, mycorrhizal fungi, and the environment. This scheme delineates interdependencies between mycorrhizal costs and benefits, resulting in a continuum of outcomes, ranging from highly beneficial to potentially detrimental effects. This scheme is based mainly on the evidence gathered for arbuscular mycorrhizas, but also appears to be generally applicable to ectomycorrhizas. Processes in other mycorrhizal types (especially orchid mycorrhizas and mycorrhizas of achlorophylous plants) may follow different trajectories. M+ mycorrhizal plant, NM non-mycorrhizal plant

studies on both ECM and ERM associations report growth benefits to the host plants upon colonization of the roots by the symbiotic fungi (Choi et al. 2005; Diedhiou et al. 2005; Finlay et al. 1992; Jansa and Vosátka 2000). The extent of root colonization by the different fungi and the magnitude of benefits, however, depend on a broader environmental context and not only on the P availability (Hoeksema et al. 2010). Additional factors that determine the extent of root colonization and mycorrhizal benefits are: soil nitrogen (N) levels, seed size and nutrient reserves contained in the seed, plant age and growth rate, and the identity of both plant and fungal species (Corrêa et al. 2006; Duponnois et al. 2008; Egerton-Warburton and Allen 2001).

6.2 Different Forms of P in the Soil and Their Accessibility to Mycorrhizas

6.2.1 Forms of P

Phosphorus is present in different forms in the soil. Inorganic forms (crystalline apatites; amorphous phosphates of calcium, potassium, iron and aluminum, and other phosphates; inorganic polyphosphate; and orthophosphate) differ greatly in their solubility in water and in their chemical reactivity (Dou et al. 2009; Holford 1997). P is also a component of an array of organic compounds present in the soil, such as nucleic acids, phospholipids, inositol phosphates, and many metabolic intermediates (see also Doolette and Smernik 2011; Bünemann et al. 2011). In contrast to the diversity of P forms present in the soil, the only form taken up in significant amounts across the plasmalemma of both the plant and mycorrhizal fungal cells is orthophosphate (P_i), preferentially as $H_2PO_4^-$ ions (Rausch and Bucher 2002; Smith 2002). Although various organic P forms have been reported as potentially utilizable by some microorganisms, it appears that their enzymatic cleavage to P_i actually occurs before the cross-membrane uptake, in the close vicinity of the microbial cells (Heath 2005). Therefore, P_i in the soil solution close to the plant and/or fungal cells plays a pivotal role in the uptake of P by the plants, both via the direct and the mycorrhizal pathways (Smith 2002; Smith et al. 2004). Direct P uptake pathway refers to acquisition of P_i by the plants from the soil solution through rhizodermis cells or root hairs. Mycorrhizal P uptake pathway refers to acquisition of P from the soil solution by mycorrhizal hyphae, translocation of P throught the extraradical mycelium, release of P from the mycorrhizal hyphae within the roots, and uptake of this released P by the root cells (usually in the cortical layer).

6.2.2 Kinetics of P Acquisition by Hyphae

Uptake of P from the soil solution to the cells (either of plant roots or mycorrhizal fungal hyphae) is mediated by P_i transporters (Rausch and Bucher 2002; Smith 2002; Tatry et al. 2009). These transporters are large proteins with several

transmembrane domains and are responsible for the proton or sodium symport of phosphate molecules across the membrane and against a steep electrochemical gradient (Karandashov and Bucher 2005; Smith 2002). These proteins have been most studied in AM fungi, but recently, two P_i transporters from ECM fungus Hebeloma cylindrosporum have also been characterized (Tatry et al. 2009). The first mycorrhizal P_i transporter was identified from cDNA libraries of Medicago truncatula roots colonized by Glomus versiforme, using hybridization with a probe derived from a yeast P_i transporter (Harrison and van Buuren 1995). Further experiments, including expression of this P_i transporter in a yeast P_i transporter (pho84)-mutant, indicated Michaelis–Menten kinetics with an apparent $K_{\rm m}$ value of 18 µM. This value is one to two orders of magnitude higher than that predicted for high-affinity P transporter systems of AM fungi (Schweiger and Jakobsen 1999; Smith et al. 2001; Thomson et al. 1990). The discrepancy, however, could easily be due to problems with heterologous gene expression, as suggested earlier (Smith et al. 2001). Recently reported $K_{\rm m}$ values for the two P_i transporters of the ECM fungus *Hebeloma cylindrosporum* (4 and 55 μ M) were closer to the values predicted from earlier hydroponic experiments (Tatry et al. 2009; van Tichelen and Colpaert 2000). All other reported details on P_i transporters from different AM fungi only refer to the so-called high-affinity transporter family, important for uptake of P_i from the soil solution, where the P_i concentration does normally not exceed 10 µM (Harrison 1999; Marschner 1995). The other (low-affinity) P_i transporter system, apparently operating in germ tubes of Gigaspora margarita (Thomson et al. 1990), has not yet been characterized at the gene level.

The described P_i transporters of both AM and ECM fungi show a high degree of structural conservation and other similarities to the P_i transporters of other organisms (Karandashov and Bucher 2005; Schachtman et al. 1998). The energetic expenditure of this high-affinity P_i acquisition is not fully resolved, but the estimates are between two and four protons per molecule of phosphate (Jennings 1996; Leggewie et al. 1997; Rausch and Bucher 2002 and references therein), with the proton gradient being generated by H⁺-ATPases (Requena et al. 2003; Smith and Read 2008 and references therein). The other aspect of P_i uptake kinetics is the P_i inflow per unit of hyphal biomass or hyphal surface. Previous studies indicate some interspecies variation in both the AM and ECM fungal groups (Smith and Read 2008; van Tichelen and Colpaert 2000). This might relate to the kinetic parameters of the different P_i transporters and their expression patterns in the fungal mycelium in the soil. The uptake of P_i into the hyphae may further be modulated by possible differences between the different fungal species and genotypes in the intensity of hyphal proliferation and the dynamics of the hyphal networks.

Improved acquisition of P by mycorrhizal plants appears to be derived from several characteristics. The mycorrhizal hyphae are capable of penetrating smaller soil pores (5–30 μ m) than the roots (>50–100 μ m), thus expanding access to the soil. In addition, due to their size, the formation of a prominent P depletion zone around the individual hyphae is minimal. Mycorrhizal hyphae are also more efficient at spatial exploration of the soil volume (up to 50 m hyphae g⁻¹) as compared to roots (up to 0.1 m roots g⁻¹) and have a lower carbon cost per unit

of hyphal surface as compared to the root surface (Gregory 2006; Jansa et al. 2003a, 2005; Li et al. 1991; Schnepf et al. 2008; Tinker 1975). In contrast, a poorly studied aspect is the dynamics of the hyphal networks (de Vries et al. 2009; Fitter et al. 2004). It appears that short-lived hyphal structures, like the branched-absorbing structures described for AM fungi from the genus Glomus, can provide a highly flexible pathway for acquisition of P and other nutrients to the plants from the soil. These structures have rather high turnover rates (7–35 days from initiation to death under axenic conditions) as compared to backbone hyphal strands, which retain cytoplasm over 3 months under axenic culture conditions (Bago et al. 1998, 2004; de Vries et al. 2009). Hyphal turnover of AM fungi from genus Glomus, as estimated by isotopic signature of carbon supplied to the plants and recovered in the AM hyphae, was only about 5-6 days (Staddon et al. 2003). These results are congruent with the time-span of the fine and short-lived hyphal structures, as presented above, as well as with previous estimates (5–7 days) based on microscopy of soil hyphae (Friese and Allen 1991). The rates of hyphal turnover in ECM and ERM networks are not well known, but are probably lower than in the AM fungi. Estimates from a carbon-flux study comparable to the study of Staddon et al. (2003) indicated an average lifespan for ECM mycelium of about 9 days, whereas rhizomorphs of some ECM fungi were observed to live for a number of months (Godbold et al. 2006 and references therein). Great levels of variability with respect to the hyphal growth and/or turnover rates have been recognized within each mycorrhizal type and between different fungal taxa (Downes et al. 1992; Godbold et al. 2006; Wallander 2006). Possibly, part of this variability could be explained by the different P acquisition strategies of the different mycorrhizal fungi. For example, soil P mining (defined here as accessing recalcitrant P sources through solubilizing or hydrolyzing exudates according to Lambers et al. 2008) would assume longerlived mycelium in the same soil patch, whereas P scavenging (defined here as collecting the easily available P beyond the reach of roots) could more efficiently be carried out by fungi with fast hyphal turnover, expanding rapidly into uncolonized soil patches. This hypothesis remains to be tested experimentally within each of the mycorrhizal types as well as between the different types.

6.2.3 Access to Recalcitrant P Forms, Weathering, and Mineralization

Varying amounts of information are known about different mycorrhizal types and their effects on release of P from recalcitrant forms present in soil. Many ECM fungi have been shown to be able to release P_i from poorly soluble P sources such as apatite. This function has been demonstrated in numerous pure culture, pot, and microcosm studies, and was recently reviewed by Rosling (2009). The relevance of pure culture studies has, however, been questioned because large amounts of added organic carbon can induce production of acids at higher rates than under natural conditions. Pot and microcosm studies are also difficult to interpret because other

microorganisms (some of them having the capacity to solubilize sorbed phosphate and/or the capacity to produce exocellular phosphatases) are usually present in the system (Jones and Smith 2004; Vessey 2003). These factors imply that the ECM could potentially take up P_i primarily released by the other microbes and transfer it to plants. In extreme cases, this could lead to measurable elevation of plant P uptake from recalcitrant sources, even when the involved mycorrhizal fungus was incapable of solubilization of the recalcitrant P on its own. However, Smits et al. (2008) recently demonstrated fungal-induced weathering of apatite in sterile microcosms with *Pinus sylvestris* seedlings colonized by the ECM fungus *Paxillus involutus*. Fungal colonization of apatite grains (Fig. 6.2) increased weathering rates threefold, and ¹⁴C simultaneously supplied to the plant was preferentially allocated to apatite patches colonized by the fungus. The proposed mechanism for apatite dissolution is enhanced acidification and chelation of calcium cations from the apatite through fungal exudation of oxalic acid. Complexation of calcium with oxalic acid would then lead to formation of calcium oxalate crystals on the surface of the fungal hyphae in contact with the apatite. These crystals have previously been observed (Allen et al. 1996; Landeweert et al. 2001), as have the elevated levels of oxalic acid in the ectomycorrhizal mats of forest soils (Griffiths et al. 1994).

This relationship is supported by the observation that apatite grains introduced into forest soil usually become heavily colonized by ECM hyphae (Hagerberg et al. 2003; Turpault et al. 2009). Furthermore, Wallander and Thelin (2008) demonstrated that this fungal colonization became more intense when P levels of Norway spruce needles dropped below 1.5 mg P g⁻¹ dry weight. This value is close to P-limiting conditions (1.3 mg P g⁻¹ dry weight) according to Linder (1995), suggesting that carbon allocation to fungal-colonized nutrient patches is regulated by the nutrient status of the tree. In spite of these studies, direct evidence that ECM fungi enhance the rates of apatite weathering is still limited. For example, Turpault et al. (2009) incubated apatite grains in mesh bags for 4 years in a beech forest in



Fig. 6.2 Apatite grains (1 mm diameter) colonized by ectomycorrhizal fungus *Paxillus involutus* in sterile microcosms. Reproduced from Smits et al. (2008), with permission

western France. Half of the bags were placed in trenched plots to which roots and mycorrhizal hyphae had no access and the other half were placed at different depths in soil accessible to roots and mycorrhizal hyphae. Apatite grains in untrenched plots were heavily colonized by fungal hyphae (presumably ECM) and, in contrast to the grains without roots, showed many weathering marks (as assessed by electron microscopy). However, with exception of the treatment at a soil depth of 25 cm (0.2% apatite weight loss over 4 years without mycorrhizal roots versus 0.5% weight loss with the roots), the apatite dissolution measured as a loss of mass was seemingly unaffected by the presence of mycorrhizal roots when compared to the trenched plots established in the same soil. Similarly, in a study by Wallander and Thelin (2008), apatite grains from P-limited forests did not dissolve significantly faster than apatite grains from P-sufficient forests, based on the amount of rare earth elements (La, Nd, Sm, Eu, Tb and Yb) from the apatite that accumulated in mycorrhizal roots surrounding the mesh bags. Both the above studies, however, were carried out under acidic soil conditions (pH \leq 4.3), which can preclude strong mycorrhizal effects on apatite solubilization through mycorrhizosphere acidification.

Although it seems indisputable that ECM fungi have the potential to release P from phosphorus-containing minerals under laboratory conditions, the extent to which this has an influence on field weathering rates of apatite is still the subject of much debate (Hutchens 2009; Rosling et al. 2009; Sverdrup et al. 2002; van Scholl et al. 2008). Uptake of elements from apatite by ECM fungi under field conditions has been previously demonstrated (Blum et al. 2002; Hagerberg et al. 2003), but the quantitative role of this process, operating on very long time scales (for contemporary science), is difficult to estimate.

Different approaches have been used to examine the roles played by different mycorrhizal fungi in mobilizing P (and N) from organic substrates and these have been reviewed by Read and Perez-Moreno (2003), among others. These approaches range from axenic systems in which the fungal hyphae are exposed to identified model compounds, to microcosm and field studies using more natural biological substrates. ERM fungi produce a range of organic polymer-degrading enzymes that can attack molecules such as chitin, lignins, polyphenols, and tannins, which either contain N or protect access to organically bound or spatially inaccessible N and P sources (organic or inorganic). Detected enzymes include lignases, polyphenol oxidases, laccase, and catechol oxidase, and the ability to degrade hydrolyzable polyphenols appears to be more extensively developed in ERM than in ECM fungi (Bending and Read 1996a, b). In addition to degrading structural components of plant litter, the ERM fungi also produce enzymes that hydrolyze P-containing molecules. Experiments performed under axenic conditions using DNA as a sole P source (Leake and Miles 1996; Myers and Leake 1996) have shown that phosphodiesters can be used by the ERM fungi as sole P sources without the intervention of other saprotrophs. These findings were corroborated by studies of the ERM fungi isolated from Woollsia pungens roots (Chen et al. 1999). All four of the studied isolates were able to utilize various organic compounds as a sole carbon and N source, and two of the isolates were able to grow on DNA or inositol sodium hexaphosphate as sole sources of P, with higher biomass production than

Hymenoscyphus (now *Rhizoscyphus*) *ericae*. The relevance of the above results for the rates of acquisition of P by the ERM fungi and associated host plants from complex organic substrates under nonsterile soil conditions needs, in many cases, to be refined by coupling enzymatic assays and isotope labeling, as in the N studies of Wurzburger and Hendrick (2006, 2009).

Similar studies to those on ERM listed above have also been conducted on ECM fungi involving "natural" substrates. In these studies, pollen grains or dead nematodes were added to microcosms containing mycorrhizal (Paxillus involutus) or non-mycorrhizal Betula pendula seedlings (Perez-Moreno and Read 2001a, b). More than 96% of the P in pollen (measured as P concentration in the pollenenriched soil patches) added to mycorrhizal microcosms was removed and, on average, 25% of this was transferred to the mycorrhizal seedlings (calculated from the system P budget, comparing mycorrhizal and non-mycorrhizal plants and assuming no significant contribution of seed P to the plant P content). In contrast, only 25% of the P in pollen added to non-mycorrhizal microcosms was removed and only 7% of this ended up in the non-mycorrhizal plants, suggesting that ECM fungal hyphae play an important role in resource capture from organic substrates. In the study with nematodes, 65% of the P originally present in the nematodes was removed from the site of addition and 73% of this was transferred to the mycorrhizal B. pendula plants. In non-mycorrhizal systems, the plants gained half as much P as the mycorrhizal systems, representing only 22% of the total originally present in the nematodes. In earlier studies, Bending and Read (1995a, b) examined the structure and function of the ECM fungal mycelium in relation to nutrient mobilization from forest litter. In microcosms containing Pinus sylvestris seedlings, colonization of organic material from the fermentation horizon by Suillus bovinus reduced concentrations of P by 22%, but colonization by Thelephora terrestris had no effect. Activities of nutrient-mobilizing enzymes in birch litter colonized by Paxillus involutus were studied and phosphomonoesterase activity increased 28-50 days but decreased again between 50 and 98 days after the initial colonization of the organic patches by the fungal mycelia. The final levels of activity were below those of uncolonized litter, but in these unsterile substrates it was not possible to distinguish between the activities of mycorrhizal fungi and those of saprotrophs. Another study addressing the utilization of inositol hexaphosphate by ECM fungi (Colpaert et al. 1997) demonstrated substantial extracellular acid phosphatase activity associated with mycelia of Thelephora terrestris and Suillus luteus that was correlated with mycelial biomass and increasing P nutrition of the mycorrhizal plants. Phytase activity of the mycelium could not be detected, but activity at the surface of mycorrhizal roots was higher than that at the surface of non-mycorrhizal roots, though the relative contributions of plant roots and fungi to hydrolysis of soluble inositol hexaphosphate were unclear. Tibbett and Sanders (2002) have shown that colonization of willow roots by Hebeloma syrjense resulted in substantial improvement of P capture by the plants from plant litter (8% of the added P transferred to the shoots within 35 days in mycorrhizal plants as compared with only 1% in the non-mycorrhizal plants). This may be due to either the shortcircuiting of the organic P re-cycling between soil and plants via mycorrhizal

hyphae, or secondarily through the effects of the ECM on other components of the system involved in P cycling (such as bacteria, saprophytic fungi, mites, collembolans etc.). In this context, it is interesting to mention an earlier study that documented efficient transfer of P from the mycelium of the saprophytic fungus *Hypholoma fasciculare* to the ECM fungi *Suillus variegatus* or *Paxillus involutus* (Lindahl et al. 1999). In this study, up to 25% of the ³²P contained in the hyphae of *Hypholoma* appeared in the mycorrhizal plants, whereas transfer from the mycorrhizal fungi to the saprophyte was at least one order of magnitude lower.

Studies of the extraradical hyphae of Glomus intraradices (Koide and Kabir 2000) have demonstrated that this AM fungus can hydrolyze organic P and transfer it through the mycelium. The magnitude of these processes under unsterile soil conditions, however, remains poorly quantified, and the importance of organic P mineralization by the AM fungi themselves has been called into question by other experiments (Joner and Jakobsen 1995; Joner and Johansen 2000). In another study, wheat was grown in chambers composed of several compartments. These compartments permitted both root and mycorrhizal hyphae growth, or, blocked root access and allowed for only hyphal growth. The soils in different compartments were then supplemented with large amounts of P (200 mg kg⁻¹ soil) in inorganic or organic forms. Control chambers without mycorrhizal fungi were also established. Elevated phosphatase activity was observed in the root-free soil colonized by Glomus mosseae when compared to AM-free soil, particularly upon organic P addition (Tarafdar and Marschner 1994). Similarly designed experiments using red clover and Glomus versiforme suggest that AM fungal colonization of a root-free soil amended with organic P makes a significant contribution to plant uptake of P from sources such as lecithin, RNA, and sodium phytate (Feng et al. 2003). On the other hand, the study by Antibus et al. (1997) demonstrated that field-collected AM roots of red maple had consistently lower levels of phosphatase activity than ECM roots of the same plant species.

Tarafdar et al. (2001) showed that fungal (Aspergillus spp.) acid phosphatases were more efficient than plant enzymes at mobilizing P from lecithin and phytate. This could be interpreted, on one hand, as proof of the capacity of fungi in general to efficiently hydrolyze organic P in the soil or, on the other hand, as a demonstration of the capacity of certain specific fungal groups (particularly the soil saprophytes) to hydrolyze such substrates. It is now clear from a range of studies that ERM and ECM fungi have the saprotrophic capacity to intervene in microbial mobilization-immobilization cycles and to sequester both N and P from the organic complexes formed during the decomposition of microbial, faunal, and plant remains. In heathland and forest ecosystems, these are the dominant sources of both N and P, and the enzymatic capacity to sequester these nutrients from complex organic substrates is probably most highly developed in the ERM fungi (Smith and Read 2008). The evidence for AM fungi is less clear and is complicated by the need to distinguish between the physiological activity of the AM hyphae and that of other fungi or bacteria that might be associated with them. Experiments by Hodge et al. (2001) demonstrated accelerated decomposition and N uptake from organic material associated with AM hyphae, but the potential contribution of other soil saprotrophs was unclear. Further experiments by Leigh et al. (2009) have shown uptake of P and N associated with patches of organic material, but again, additional uptake of P from bone-meal and Terragreen substrate present within the microcosms cannot be ruled out. The ambiguity here indicates the need for further experiments investigating the potential role of mycorrhiza-associated bacteria. The role of mycorrhizal fungal hyphae as primers of soil microbial activity has been discussed by a number of authors (e.g., Jones et al. 2004; Talbot et al. 2008; Toljander et al. 2007), but so far our understanding of how this regulates nutrient acquisition and transfer within the mycorrhizosphere is limited.

6.2.4 Mycorrhizas as Compound-Specific Filters

In an overwhelming number of studies, the role of mycorrhizal symbiosis in plant acquisition of P has been documented under a wide range of environmental conditions, in different ecosystems and for different host plants (Arihara and Karasawa 1998; Cardoso and Kuyper 2006; Jansa et al. 2009 and references therein; Smith and Read 2008 and references therein). Nevertheless, to regard mycorrhizas only as P pumps would be utterly incorrect, particularly if considering ECM and ERM fungi in comparison to the most abundant form of mycorrhizal symbiosis formed by the AM fungi. In addition to their role in P nutrition, ECM, ERM and, to a lesser extent, AM fungi are involved in acquisition of N by plants, both from inorganic and organic sources (Finlay et al. 1992; Johansen et al. 1992; Mäder et al. 2000; Read et al. 2004). Involvement of ECM in plant water uptake has been shown (Allen 2007; Plamboeck et al. 2007) and the mycorrhizas are also known to alleviate deficiencies in micronutrients such as zinc and copper. In addition, at least some genotypes of mycorrhizal fungi have the capacity to protect their host plants from acquisition of soil pollutants such as radiocaesium (de Boulois et al. 2008; Joner et al. 2004; Ladeyn et al. 2008) and heavy metals (Joner et al. 2000; Martino et al. 2000; Sharples et al. 1999; Sudová et al. 2008), while maintaining the P and/or zinc supply to the plants (Joner et al. 2004; Soares and Siqueira 2008).

6.3 Translocation of P Within the Hyphae and Its Release to the Plants

6.3.1 Transport Within the Hyphae

The P taken up by mycorrhizal fungi from the soil solution is used to meet the physiological demand of the fungus, with the remainder transported to the plants or stored in the hyphae. Transport of P to the plant implies a long-distance transfer through the hyphal network, which, in both ECM and AM fungi, is assumed to involve polyphosphates (Bucking and Heyser 2003; Ezawa et al. 2002). The short-chain polyphosphates, resulting from depolymerization of longer polyphosphate

chains transferred over long distances, appear to be the immediate source of P for the plants (Ohtomo and Saito 2005; Solaiman et al. 1999; Takanishi et al. 2009). Rapid transfer rates of P via mycorrhizal fungal hyphae have been measured either using radioisotope labeling or microscopy (Bago et al. 2002; Cooper and Tinker 1978; Cox et al. 1980; Nielsen et al. 2002; Rhodes and Gerdemann 1978; Timonen et al. 1996). Together with the capacity of some ECM and ERM fungi to release P from recalcitrant sources, this rapid transfer represents what has been referred to as a mycorrhizal short-circuit in soil–plant P cycling, bypassing release from minerals or organic sources by free-living soil microorganisms (Johnson et al. 2005; Pankow et al. 1991).

6.3.2 Release of P to the Plant

The mechanics of how the P is released from the fungi to the plants has not yet been described for any mycorrhizal type. It is also not yet known whether this process is through passive leakage or an active transport system. It is likely that some fungi can, to different extents, retain P in their mycelium during translocation, resulting in partial immobilization and perhaps storage of P on the way from the soil to the plants (Boddington and Dodd 1999; Chilvers and Harley 1980; Harley and McCready 1981; Solaiman et al. 1999) - either as a result of slow transfer within the hyphae or due to limited release to the plant. In either case, P cycling between soil and plant is slowed down and the P could potentially also be released back to soil from the fungal hyphae or transferred to other soil organisms upon hyphal death due to soil disturbance, parasitism, or grazing. On the other hand, transitional storage of P in the fungal mycelium may function as a buffer ensuring continuous supply of P into long-lived plants such as trees or perennial herbs in a changing environment (Genet et al. 2000; Lussenhop and Fogel 1999; Read 1984). Thus, in the long run, the rapid provision of P to the host plants by fungi may not necessarily be the most beneficial system.

The plant side of the transfer, namely the mycorrhiza-inducible P_i transporters expressed in the close vicinity of fungal structures such as hyphal coils or arbuscules, has already been characterized for several plant species establishing AM symbiosis (e.g., Glassop et al. 2005; Javot et al. 2007; Nagy et al. 2005; Paszkowski et al. 2002; Rausch et al. 2001). Similar transporters are also likely to exist in ECM and ERM plants, but have not yet been characterized at the molecular level.

6.3.3 Consequences of Mycorrhizal P Acquisition for the Plants and for Maintenance of Mutualism

The efficiency of P acquisition by the mycorrhizal fungi, the temporary P immobilization in the fungal biomass, and the controlled P release to the plants all have important consequences for plant P nutritional status and growth. Although large improvements of plant P uptake have been reported upon mycorrhizal colonization of the roots by certain AM fungal species, association with other fungi may yield negative growth responses, and thus qualify the relationship as parasitic (Johnson et al. 1997; Smith et al. 2004). The potential for fungal control of P release to the plants points to one possible mechanism for maintaining the mycobiont diversity and mutualistic nature of the symbiosis through diversification of the benefits of carbon trading between the plant and the mycorrhizal fungi (Cowden and Peterson 2009; Helgason and Fitter 2009; Kiers and van der Heijden 2006).

In cases where the roots are colonized simultaneously by several species and/or genotypes of the mycorrhizal fungi, there is some evidence for preferential carbon distribution to more beneficial mycorrhizal symbionts (Bever et al. 2009; Fitter 2006). This preference is probably dependent upon the rates of P transfer within the specific root cells or fragments colonized by the different fungi. This is corroborated by physiological studies that showed that the release of P from intraradical hyphae of Gigaspora increased upon glucose addition (Solaiman and Saito 2001). However, there is also experimental evidence that some AM fungi can gain carbon from plants in spite of the net P gain of the host plant being very limited, such as under P- and N-sufficient conditions (Hoeksema et al. 2010; Pearson and Jakobsen 1993; Smith et al. 2003, 2004, 2009). Additionally, the existence of mycoheterotrophic plants, where the plants apparently receive both mineral nutrients and carbon from the fungus (Bidartondo et al. 2002; Imhof 2009; Taylor et al. 2004), is difficult to reconcile with the hypothesis of preferential carbon allocation to the most beneficial fungal symbiont. Possibly, some of the processes are regulated at the ecosystem level and not at a single plant level. This may mean that the answers are hidden in plant community ecology, source-sink relationships and so called "common mycorrhizal networks" interconnecting different plants (Bever 1999; Bever et al. 2009), although little evidence is so far available about mycorrhizamediated transfer of P and carbon between plants (Newman and Ritz 1986; Philip and Simard 2008; Robinson and Fitter 1999; Selosse et al. 2006; Yao et al. 2003). Furthermore, experimental data have recently been gathered showing a more inconspicuous contribution of mycorrhizas to the P acquisition by the plants without detectable plant growth or net P uptake improvements (Smith et al. 2003, 2004). These effects are, however, only seen in carefully designed radioisotope experiments, using non-mycorrhizal mutant genotypes, or expression analyses of fungal and plant P transporters (Burleigh and Bechmann 2002; Grace et al. 2009; Li et al. 2008; Poulsen et al. 2005; Smith et al. 2009). Results of the above studies showed that the symbiosis may be fully functional, even if traditional measurements of mycorrhizal "benefits" such as growth or net P uptake improvements were not able to detect its contribution (Facelli et al. 2009). They also indicate that the definition of the symbiosis, currently often implicating measurable benefits to both partners, may need to be broadened to cover these cases of association with no obvious "benefits" (Cavagnaro et al. 2004; Jones and Smith 2004).

6.4 Functional Diversity of Mycorrhizas with Respect to P Uptake

In addition to major functional differences between the different mycorrhizal types, variation in P uptake patterns and efficiency have been recognized between species and genotypes of both AM and the ECM fungi (Boddington and Dodd 1999; Cairney 1999; Cavagnaro et al. 2005; Jansa et al. 2005; Munkvold et al. 2004; Smith et al. 2003). In these studies, radioisotope labeling has proven to be a particularly important approach (see also Frossard et al. 2011). The recorded functional diversity among the different mycorrhizal fungi appears to be important for understanding mycorrhizal functioning in the field. This is because roots of most plants are colonized by a mixture of mycorrhizal fungal species, usually, but not always, belonging to the same type (van der Heijden and Vosátka 1999; van der Heijden et al. 1998).

It has been postulated that fungi differing in P acquisition strategies could complement each other when sharing the same root system, resulting in greater symbiotic benefits than those conferred by each of the fungi in isolation (Koide 2000). Although this may well be the case, direct experimental evidence for functional complementarity in P acquisition within mycorrhizal communities is still limited (Jansa et al. 2008; Maherali and Klironomos 2007). Better understanding of the components of the mycorrhizal uptake pathway in different mycorrhizal fungi (e.g., numbers of alternative P transporters, their regulation and expressional dynamics, dynamics of the polyphosphate pool in the fungal hyphae) and how the functional diversity is structured within fungal communities, will be necessary before more general conclusions can be drawn. A combination of approaches spanning molecular biology (Burleigh et al. 2002; Grace et al. 2009; Jansa et al. 2008; Tatry et al. 2009), isotopic labeling (Jakobsen et al. 1992; Jansa et al. 2005), carefully designed pot experiments (Maherali and Klironomos 2007; Smith et al. 2004), and modeling (Antoninka et al. 2009; Deressa and Schenk 2008; Schnepf et al. 2008, 2011) is essential for further progress in this area. Likewise, interactions between mycorrhizal P uptake, carbon costs, and cycling of other nutrients such as N at the whole plant or plant community levels must be considered for an ecologically relevant picture (Grelet et al. 2009; Johnson et al. 2010; Smith et al. 2009).

6.5 Human Impact on the Mycorrhizal Pathway of P Acquisition by Plants

Agricultural activities, pollution, climate change, and other anthropogenic environmental influences affect mycorrhizal symbioses and their role in P acquisition by plants. Substantial information has been accumulated on these influences over the years; however, only a fraction is discussed here. For a more detailed overview, the reader is referred to other sources (Allen et al. 2003; Allison and Treseder 2008; Drigo et al. 2008; Jansa et al. 2006).

In agricultural systems, the application of water-soluble mineral P and N fertilizers usually reduces the dependency of plants on nutrient uptake via the mycorrhizal pathway. This forced selection may eventually promote fungi that are less beneficial to plants or promote an abundance of plants that are less mycorrhizadependent (Covacevich et al. 2007; Jansa et al. 2006; Johnson 1993). Crop breeding efforts for high yields have been shown to inadvertently select for lower mycorrhizal dependencies, probably through selection for greater dependency on mineral fertilizer inputs (Hetrick et al. 1993; Tawaraya 2003; Zhu et al. 2001). These effects are not limited to agricultural plants and soils as nutrients, and pollutants often leach into natural ecosystems from agricultural fields, inappropriate waste management, and/or industrial activities. These effects are especially pronounced in forests and heathlands. The activity of fungi sensitive to elevated nutrient availability and pollutants decreases, and plant species or genotypes are favored that can better tolerate the pollution and/or depend less on nutrient acquisition via mycorrhizal fungi (Dighton 1995; Egerton-Warburton and Allen 2000; Kieliszewska-Rokicka 1999: Reišek 1991: Robertson et al. 2007).

Elevation of CO_2 levels in the atmosphere may transiently increase the biomass and metabolic activities of mycorrhizal fungi, such as P transfer from soil to plants, due to a reduction of carbon limitation and through creating a greater requirement for P and other nutrients by the plants (Alberton and Kuyper 2009; Millard et al. 2007). In a longer perspective, however, this may result in a more rapid exploitation of soil nutrient reserves and an increased sensitivity of ecosystems to disturbance events (Pritchard et al. 2008). Global warming and redistribution of precipitation patterns may also change the activity of soil microorganisms in general and mycorrhizal fungi in particular (Fitter et al. 2004), but the future prospects remain rather blurred – usually the availability of water and not soil nutrients appears to be the primary driver of the expected ecosystem changes (Aerts et al. 2009; Allison and Treseder 2008; Heinemeyer et al. 2007). If water availability is not limiting the microbial activity, decay of roots and plant debris may be accelerated by global warming, which could in turn speed up mycorrhizal P capture from these sources (e.g., Carleton and Read 1991) thus speeding up the P cycling. Nevertheless, genotypic differences in temperature tolerance and acclimation between different mycorrhizal fungi might also contribute to the variability in the observed responses in mycorrhizal community composition and functioning to climatic changes (Malcolm et al. 2008).

A final aspect of human activity worth mentioning here is the phenomenon of plant invasion. It has been proposed that non-mycorrhizal alien plants, such as *Hakea* spp. that form cluster roots, could gain a P-acquisition advantage in ecosystems normally dominated by mycorrhizal plants (Allsopp and Holmes 2001; Sousa et al. 2007). For alien plants like *Centaurea maculosa* in Northern American grasslands, a different mode of action has been proposed: invasive plants could tap into the existing mycorrhizal networks and divert the flux of resources, such as P, to individuals with this ability (Batten et al. 2008; Callaway et al. 2004; Zabinski

et al. 2002). Another scenario is that the native mycorrhizal fungi could be suppressed by (non-mycorrhizal) alien plants (e.g., *Alliaria petiolata* in North America), which would in turn result in suppression of the native plants that rely on their (native) mycorrhizal symbionts (Callaway et al. 2008; Mitchell et al. 2006). Although experimental evidence for these processes and their importance in the phenomenon of ecological invasions is still equivocal, it is becoming more and more apparent that underground processes, nutrient balances, and associated microflora including mycorrhizal fungi might all be important players in plant invasions, and thus relevant topics for further study.

6.6 Conclusions

Mycorrhizal symbioses are highly diverse in their relationships with plants and ecosystems and are involved in different P cycling processes. Although evidence is accumulating to suggest that some ECM fungi have the capacity to induce or accelerate weathering of P-bearing minerals, thus affecting P inputs into the biological P cycling within the ecosystems, these processes are still largely unquantified. It is therefore unclear how important these processes are compared to the other input pathways such as aerial deposition, sedimentation, abiotic weathering, and anthropogenic inputs via fertilization and pollution, and how they vary amongst different ecosystems and on different geographic and time scales (Johnson et al. 2010; Newman 1995; Smits et al. 2008). To solve these issues, precise measurement of P fluxes should be continued, using radio- and stable isotope methods (Frossard et al. 2011), as well as complementary modeling efforts and long-term observations (Rosling 2009; Rosling et al. 2009; Schnepf et al. 2011). Soil and environmental conditions must always be taken into account when interpreting experimental results (e.g., little biotic effect on solubilization of apatite through acidification should be expected for strongly acidic soil conditions). Additionally, chemical reactivity and other relevant information (chemical composition, crystallinity, grain size, provenance) of compounds such as apatite used for experimental studies should always be assessed and reported to allow strict reproducibility of results.

The importance of mycorrhizal symbiosis in plant P uptake has been established in hundreds of studies published over many decades (Smith and Read 2008). This extends from traditional comparisons of small (non-mycorrhizal) and big (mycorrhizal) plants, where the differences in net P uptake are easily demonstrable; through more inconspicuous cases, where improvements of net P uptake are not always translated into improved plant growth (Jansa et al. 2005); to the extreme cases, where net P uptake of the plant is not different between mycorrhizal and nonmycorrhizal plants, although the mycorrhizal P uptake pathway is fully operational (Grace et al. 2009; Smith et al. 2004, 2009).

In spite of some well-documented model cases, the mechanisms governing the transfer of P within the common mycorrhizal networks interconnecting different

plants, as well as the involvement of other soil microorganisms in soil–mycorrhizaplant P transfer are still only fragmentarily understood (Finlay 2008; Lindahl et al. 1999; Toljander et al. 2006). Extensive use of mineral P fertilizers and breeding for high-yielding crop varieties under fertilized soil conditions has seemingly led to selection of genotypes that are less responsive to mycorrhizal symbiosis than some of the older cultivars (Hetrick et al. 1993). This might be of concern regarding efficient use of natural resources in the future (Sawers et al. 2008).

Major differences in the accessibility of different soil P pools to different mycorrhizal types explain the distribution of the different mycorrhizal types in ecosystems: AM fungi appear to access mainly orthophosphate in the soil solution, ECM can access both inorganic and organic P in soil, and ERM appear to be able to access mainly the P contained in organic substrates (Read and Perez-Moreno 2003). Direct acquisition of P from decaying plant biomass or other organic substrates via mycorrhizal pathway effectively short-circuits the soil-plant P cycling, in which the largest portion of P mineralization would otherwise have to be accomplished by free-living soil decomposers before being accessed by roots or the mycorrhizal hyphae (Carleton and Read 1991). Through efficient P recycling from organic forms, capture of P_i from soil solution, and by improving soil mechanical stability, mycorrhizal symbiosis has the potential to contribute substantially to reduction of P loss through leaching and erosion (Asghari et al. 2005; Cardoso and Kuyper 2006). On the other hand, in agricultural and other ecosystems where management includes intentional removal of some plant products (green biomass, grains, fibers, wood), mycorrhizal symbiosis alone cannot guarantee long-term sustainability of production without adequate P inputs in the form of mineral or organic fertilizers, plant residues, or products resulting from wastewater treatment or municipal waste processing (Jansa et al. 2006; Oberson et al. 2011).

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