Plant community mycorrhization in temperate forests and grasslands: relations with edaphic properties and plant diversity

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Keywords
Arbuscular mycorrhiza; Biotic interactions; Ellenberg; Moisture; Mycorrhization index; Nitrogen; pH; Plant communities; Plant diversity; Plant richness; Vegetation

Abbreviations
AM = arbuscular mycorrhiza; EcM = ectomycorrhiza; ErM = ericoid mycorrhiza; OrM = orchidoid mycorrhiza; OM = obligatorily mycorrhizal; FM = facultatively mycorrhizal; NM = non-mycorrhizal; MI = mycorrhizal index; AMI = arbuscular mycorrhizal index.

Abstract
Questions: Mycorhizal symbiosis plays a key role in plant communities. Its prevalence in plant communities (mycorrhization) at larger spatial scales has so far been mostly qualitative, while quantitative studies incorporating the mycorrhizal traits of plant species are scarce. This study aims to: (1) determine the variation in general and arbuscular mycorrhization in temperate forests and grasslands, (2) study the effects of soil N, pH and moisture on mycorrhization, and (3) determine the relationships between mycorrhization and plant diversity.

Location: Temperate forests and grasslands in Estonia, Northern Europe.

Methods: To quantify mycorrhization we used a plant community mycorrhization index – community mean of mycorrhizal status weighted by plant species abundances. The effects of edaphic factors characterized by cumulative Ellenberg values on mycorrhization were analysed using linear mixed models, and the relationship between mycorrhization and diversity was evaluated with partial correlation and variance partitioning.

Results: General mycorrhization was higher in forests and lower in grasslands, opposite to arbuscular mycorrhization. Soil N, pH and moisture negatively impacted general mycorrhization, whereas arbuscular mycorrhization was positively affected by soil pH and negatively by soil N and moisture. Plant species richness was negatively correlated with general mycorrhization in forests, whereas arbuscular mycorrhization was positively associated with plant species richness, Shannon and Simpson indices in forests and across ecosystems.

Conclusions: Mycorrhization is highly dependent on soil conditions and related to plant diversity, showing its importance for vegetation science. The plant community mycorrhization index used in this study is a promising tool for quantifying the prevalence of mycorrhizal symbiosis along environmental gradients.
mycorrhizal types. However, most quantitative community studies incorporating mycorrhizal traits have focused on mycorrhizal types (Barni & Siniscalco 2000; Piotrowski et al. 2008; Phillips et al. 2013). Only in one community study were both traits (mycorrhizal status and type) analysed in parallel; Cázares et al. (2005) described the changes in abundances of plant species with different mycorrhizal types and of FM plants during succession. In addition, mycorrhization has been investigated from a fungal perspective. Treseder & Cross (2006) quantified the AM abundance in biomes using live root biomass and AM fungal root colonization intensity, however, without accounting for plant community composition. So far, there are no quantitative community studies addressing the prevalence of mycorrhizal symbiosis, using mycorrhizal status.

We measured the variation in plant community mycorrhization in temperate forests and grasslands in Estonia, using a community mycorrhization index (MI) proposed by Moora (2014). To address plant communities as whole systems and estimate the proportion of the community dependent on mycorrhizal symbiosis regardless of mycorrhizal type, we calculated general community mycorrhization index across all mycorrhizal types, based on plant species ability to form mycorrhizal symbiosis. In addition, as the majority of evidence of the benefits of mycorrhizal symbiosis comes from the research on AM plants, we also calculated an arbuscular mycorrhizal index (AMI), dependent on plant capability to form specific AM. In particular, we aimed to: (1) perform the first quantitative analysis of the variation in plant community mycorrhization among different forest and grassland vegetation types; (2) analyse the relationships between community mycorrhization and three edaphic characteristics, i.e. soil fertility, reaction and moisture content; and (3) study the relationship between community mycorrhization and plant diversity.

Methods

Study sites and sampling

The study was carried out in six forest and five grassland community types in Estonia, Northern Europe (Appendix S1). Plant community types were chosen to encompass a wide array of edaphic gradients (soil fertility, pH and moisture content). Each community type was represented by two replicate sites, where in summers 2012 and 2013 ten 1 m² plots were sampled randomly within a roughly 50 m × 50 m uniform area. Altogether the study contained 22 sites and 220 plots. All field layer vascular plant species in each plot were recorded and their percentage cover estimated. The woody canopy cover was estimated for a larger area surrounding field layer plots. For some sites data were obtained from previously conducted studies.
Edaphic condition estimates

Edaphic conditions were described using Ellenberg indicator values (Ellenberg et al. 1992), which are generally highly reliable and can complement – or even replace – direct measurements of environmental variables (Diekmann 2003). They can be more stable compared to measured environmental data, as the latter can strongly fluctuate in space and time and hence diverge from the long-term average conditions (Cain et al. 1999; Ozinga et al. 2013). In order to use Ellenberg values at the community level, we calculated cumulative Ellenberg indicator values for soil N, soil pH (reaction value according to Ellenberg) and soil moisture for each plot by summing the proportional cover-weighted Ellenberg values. As Ellenberg soil P content is not available, soil N value was taken as a proxy for soil fertility. The species without available Ellenberg values were excluded from calculation of the cumulative Ellenberg values.

Plant community diversity estimates

To determine the associations between community mycorrhization and plant community structure, we used vascular plant species richness per plot, exponential Shannon diversity (exp $H'$) and inverse Simpson dominance index ($\lambda^{-1}$), all operating with species as units, allowing their easier interpretation. To evaluate the equality of the abundances of plant species, evenness was calculated following Alatalo (1981) as:

$$\text{evenness} = \frac{\lambda^{-1} - 1}{\exp H' - 1}.$$  

where $\lambda = \sum p_i^2$ and $H' = -\sum p_i \ln p_i$ are calculated based on relative cover ($p$) of plant species $i$ in a given plot.

Plant community mycorrhization estimates

We used the community mycorrhization index (MI) and arbuscular mycorrhization index (AMI) (Moora 2014) as quantitative estimates for plant community mycorrhization to evaluate the prevalence of mycorrhizal symbiosis in ecosystems. Mycorrhizal status used to calculate the general mycorrhization index was: (1) obligatorily mycorrhizal (OM), (2) facultatively mycorrhizal (FM), and (3) non-mycorrhizal (NM). To calculate the AMI, plants were first categorized based on mycorrhizal type and then according to status: (1) obligatorily AM, (2) facultatively AM, or (3) never AM plants. The last group included plant species forming only some other type of mycorrhiza as well as NM plants. Mycorrhizal status was assigned to plants using the MycoFlor database (Hempel et al. 2013), Harley & Harley (1987), Wang & Qiu (2006) and Akhmetzhanova et al. (2012). In addition, three plant species with previously unknown mycorrhizal status (Festuca sabulosa (Andersson) H. Lindb., Geranium palustre L. and Sagina nodosa (L.) Fenzl) were studied for the occurrence of mycorrhiza (Appendix S2). The plant species of unknown mycorrhizal status, which were mostly rare (in terms of presence and abundance), were discarded from the calculation of MI and AMI.

To calculate the mycorrhization indices, all mycorrhizal statuses were given a numerical value from 0 to 1. NM or non-AM status obtained a value 0, and OM statuses were assigned a value 1. For FM status, the numerical coefficient was calculated in two ways. First, ‘fine-scale’ resolution of FM status was calculated based on literature data as a proportion of empirical observations of all reports with respect to the mycorrhizal status of the plant species (example of calculation in Appendix S3). Second, for ‘coarse-scale’ resolution, the FM status received a numerical coefficient of 0.5, an average of the values assigned to OM and NM status (example of calculation in Moora 2014). Using two different approaches gives an estimate of how they differ in ability to describe plant community mycorrhization and whether the more robust method could replace the more demanding fine-scale approach. As results were similar, only the results of the fine-scale approach and differences with the coarse-scale approach are reported in the main text, while the coarse-scale results are provided in Appendix S4.

Mycorrhization indices MI and AMI were computed as mycorrhizal status weighted by plant species proportional cover summed together. For statistical analyses those indices were expressed as logit functions:

$$(A)\text{MI} = \ln \frac{\sum (p_i \times M_i)}{1 - \sum (p_i \times M_i)}.$$  

where $p_i$ is the proportional cover of species $i$ in a plot and $M_i$ is the numerical coefficient of mycorrhizal status of species $i$.

Statistical analyses

To test whether plant community and ecosystem types differ in their mycorrhization, we used linear mixed effects models (LMM) followed by post-hoc multiple comparisons with Bonferroni correction. MI and AMI were included as response variables, community type (11 types) or ecosystem type (grassland or forest) as predictors, and site as a random factor.
Similarly, LMMs were used to study the influence of edaphic factors on MI and AMI. Cumulative Ellenberg indicator values for soil N, pH and moisture were used as predictors in models across ecosystems and for grasslands. Due to strong collinearity between N content and pH ($r = 0.8$) in the forest model, we ran two parallel models with pH and moisture and with N and moisture as predictor variables, respectively. Site was included as a random factor.

We used partial correlation to determine the association between mycorrhization and plant diversity, while accounting for the effects of possibly confounding edaphic factors. We used LMMs, and generalized linear mixed models (GLMMs) with Poisson distribution in the case of plant richness to determine the effect of soil factors on plant community mycorrhization and diversity, with site as random factor, and calculated the correlation between the residuals of both models.

We carried out variation partitioning to estimate the sole and joint effects of mycorrhization and soil factors on plant diversity using marginal $R^2$ ($R^2_m$), as recommended for fixed factors in (G)LMMS (Nakagawa & Schielzeth 2013). The results of variation partitioning are shown in Appendix S5.

All analyses were performed using R (v 3.1.1; R Foundation for Statistical Computing, Vienna, AT). The assumptions for all models were checked visually. Exp($\hat{H}'$) and $\lambda^{-1}$, were ln-transformed prior the analyses to meet model assumptions.

Results

Mycorrhization across community and ecosystem types

A total of 313 vascular plant species were recorded across both ecosystems, of which 178 and 236 species occurred in forests and grasslands, respectively. Of all plant species, 292 (93%) species (173 (97%) from forests and 218 (92%) from grasslands) were assigned a mycorrhizal type and status: 231 species formed AM, 28 species formed EcM, ErM or orchidoid mycorrhiza (OrM), 112 species were OM, 147 FM and 33 species were NM. Plant species with dual mycorrhization were counted as either AM or EcM, based on the number of mycorrhizal reports. EcM, ErM or OrM plant species dominated in forests, comprising 69% of the community (Appendix S6a), and AM plant species prevailed in grasslands, making up 79% of the community when plant species abundances were taken into account (Appendix S6b).

Community mycorrhization (MI) showed a significant difference between plant community types ($F_{10,198} = 7.38$, $P = 0.001$). Floodplain meadows had significantly lower MI than any other plant community type, whereas the highest MI was recorded in oligotrophic paludifying forest, followed by dry boreal forest (Fig. 1a). Arbuscular mycorrhization (AMI) also differed between plant community types ($F_{10,198} = 6.29$, $P = 0.003$). The lowest AMI was observed in dry boreal forest, whereas it was highest in alvar grasslands (Fig. 1b).

Regarding broad ecosystem types, forests showed significantly higher MI ($F_{1,198} = 13.62$, $P = 0.001$; Fig. 1c) and lower AMI values ($F_{1,198} = 25.58$, $P < 0.0001$; Fig. 1d) than grasslands.

Edaphic factors

The LMM including both ecosystem types showed that MI was negatively dependent on cumulative Ellenberg N value (Table 1). When ecosystems were modelled separately, MI was significantly negatively related to cumulative Ellenberg N value in forests and in grasslands. AMI was significantly negatively related to cumulative Ellenberg N value in grasslands but not in forests (Table 1).

Cumulative Ellenberg reaction value was significantly negatively related to MI in forests, and when both ecosystems were addressed together (Table 1). However, a positive relationship between AMI and cumulative Ellenberg reaction value was evident across ecosystems and in forests (Table 1).

A significant negative relationship of cumulative Ellenberg moisture value with MI, and on AMI, was found only in grasslands (Table 1). No significant relationship between MI or AMI and cumulative Ellenberg moisture value appeared in forests or across ecosystem types (Table 1).

Plant community diversity

Partial correlation across both ecosystem types showed no relationship between MI and species richness (Fig. 2a). However, when analysed separately, a negative relationship between MI and plant species richness was found in forests, but not in grasslands (Fig. 2b,c). Conversely, AMI correlated positively with plant species richness across both ecosystem types and in forests, but not in grasslands (Fig. 2d,e,f). Exponential Shannon diversity and inverse Simpson dominance were positively correlated with AMI across ecosystems (Appendices S7d and S8d) and in forests (Appendices S7e and S8e) but not in grasslands (Appendices S7f and S8f). Evenness was not correlated with MI or AMI (Appendix S9). Variation partitioning suggested that MI and AMI alone and together with soil explain roughly equal amounts of variance, while soil alone is responsible for the bulk of variance in plant diversity (Appendix S5).
Table 1. Summary of results of linear mixed models. Effects of soil nitrogen content, soil reaction and soil moisture content, calculated as community-weighted Ellenberg means, on community mycorrhization index (MI) and arbuscular mycorrhization index (AMI). Models for forests included either soil reaction and moisture or nitrogen and moisture. Numerical coefficient for facultatively mycorrhizal status in calculation of mycorrhizal indices represents the proportion of mycorrhizal reports (fine scale).

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df shown for model, residuals.
*Result differs between fine- and coarse-scale approach.
Statistically significant P-values are shown in bold.

Fig. 1. Levels of community mycorrhization (a, c) and arbuscular mycorrhization (b, d) in plant community types and in temperate forests and grasslands. Bar heights and spreads denote mean ± 1SE, letters indicate statistically significant differences after Bonferroni correction. Statistical significances are obtained from LMM with site included as a random factor. A- dry boreal forest, B- alvar forest, C- eutrophic boreonemoral forest, D- mesotrophic boreonemoral forest, E- mesotrophic paludifying forest, F- oligotrophic paludifying forest, G- alvar grassland, H- mesic boreonemoral grassland, I- moist boreonemoral grassland, J- floodplain grassland, K- coastal grassland. Numerical coefficient for facultative mycorrhizal status in calculation of mycorrhizal indices represents the proportion of mycorrhizal reports (fine scale).
Mycorrhization in temperate forests and grasslands

Fine- and coarse-scale approach

The MI and AMI, calculated using a fine-scale approach, were highly correlated to the respective mycorrhization indices calculated using the coarse-scale method (Fig. S1 in Appendix S4). The results of LMMs addressing the influence of edaphic conditions on mycorrhization were mostly consistent (Table S1 in Appendix S4). However, cumulative Ellenberg N value showed a significant negative relationship with coarse-scale AMI; cumulative Ellenberg reaction value showed no significant relationship with coarse-scale MI across ecosystems and a positive relationship with MI and AMI in grasslands. The partial regression and variation partitioning results using two ways of calculating community mycorrhization also remained unchanged (Appendix S4).

Discussion

We quantified mycorrhization in temperate forest and grassland ecosystems using a novel plant community mycorrhization index (Moora 2014) that uses mycorrhizal status of plant species, i.e. how often plant species are found to be mycorrhizal, as well as their relative abundance. General community mycorrhization was higher in forests than in grasslands, while arbuscular mycorrhization showed the opposite pattern. Community and arbuscular mycorrhization was influenced by edaphic conditions and associated with plant diversity.

Mycorrhization in temperate forests and grasslands

Forest ecosystems exhibited higher community and lower arbuscular mycorrhization than grasslands, possibly attributable to different mycorrhizal types dominating in forests and grasslands. Community mycorrhization is high when the ecosystem is dominated by obligatorily mycorrhizal plants, among which are ecto- and ericoid mycorrhizal species. The tree layer in temperate forests usually consists of ectomycorrhizal trees, whereas the field layer often consists of ericoid mycorrhizal shrubs (Read 1991). The majority of soil N and P in such forests is sequestered in organic forms, which cannot be acquired directly by plants, but is possible with the help of EcM or ErM fungi (Read & Perez-Moreno 2003). Indeed, the accumulation of organic matter and high C:N ratio favour EcM fungal colonization (Soudzilovskaia et al. 2015). AM plants are less abundant in temperate forests because AM fungi are typically incapable of using organic compounds as resources for plant nutrition. Also, AM fungal abundance has been previously shown to be highest in temperate grasslands and lowest in temperate and boreal forests (Treseder & Cross 2006), further supporting our results. In addition, EcM and ErM can have antagonistic effects on AM plants (Kovacic et al. 1984; Genney et al. 2001), possibly contributing to lower arbuscular mycorrhization in temperate forests. In contrast, the conditions for mineralization in grasslands are usually favourable, making N and P available for direct plant acquisition and hence lowering the C:N ratio. Therefore, AM fungal colonization is promoted (Soudzilovskaia et al. 2015) and either plants with AM, which is often a facultative mycorrhizal type (Hempel et al. 2013), or non-mycorrhizal plants can succeed, rendering grasslands in general less mycorrhizal and more arbuscular mycorrhizal than forests.

Edaphic gradients

Plant community mycorrhization decreased with increasing soil N content, adhering to the mutualism–parasitism continuum concept (Johnson et al. 1997; Johnson & Graham 2012). It is not cost-effective for plants to maintain mycorrhizal fungi and provide them with C that could otherwise be used for their own growth in nutrient-limited...
conditions. Therefore, plant species dominating fertile habitats are often either non-mycorrhizal or facultatively mycorrhizal. Also, FM plants may be able to control the degree of mycorrhizal colonization in roots (Koide & Schreiner 1992), enabling them to compete successfully in habitats with plentiful nutrient supply and, in addition, providing a competitive advantage over NM plants when soil nutrients are less abundant. OM plants are more common in infertile soils, whereas FM plants are more often found in fertile soils (Hempel et al. 2013). In addition, experimental addition of N has been shown to lower the abundance of mycorrhizal fungi (Treseder 2004), making decreasing community mycorrhization with increasing soil fertility expected. AM symbiosis is mostly thought to benefit plants through P and not N acquisition (Smith & Read 2008), being consistent with the findings of Peat & Fitter (1993), who observed no relationship between arbuscular mycorrhization and soil fertility (measured using Ellenberg N values) at the species level. However, in our study arbuscular mycorrhization was negatively influenced by soil N content, suggesting the important role of N in determining arbuscular mycorrhization at the plant community level.

General community mycorrhization was negatively and arbuscular mycorrhization positively associated with soil reaction. This supports the general understanding that EcM and ErM plants prevail on acidic while AM plants are favoured on neutral or slightly alkaline soils (Read 1991; Read & Perez-Moreno 2003; Soudzilovskaia et al. 2015). Analyses of British (Peat & Fitter 1993) and Central European (Hempel et al. 2013) flora confirm that AM plants tend to be associated with higher soil pH. Increasing pH induces changes in fungal:bacterial ratio towards bacterial dominance (Rousk et al. 2009; Strickland & Rousk 2010), which increases mineralization rates, resulting in elevated N availability and dominance of AM plants. In addition, increasing pH reduces P availability in the soil, while AM considerably enhances P uptake. In contrast, acidic conditions promote accumulation of organic matter and dominance of EcM or ErM fungi capable of acquiring nutrients from this substrate (Read & Perez-Moreno 2003; Nilsson et al. 2005; Soudzilovskaia et al. 2015).

Mycorrhization in grasslands increased with decreasing soil moisture. Plant trait-based analyses have shown that plant species growing in drier habitats are more often mycorrhizal than those growing in moist conditions (Peat & Fitter 1993; Hempel et al. 2013). Plants are expected to benefit from mycorrhizal symbiosis in dry soils indirectly due to improved nutrition (Neumann & George 2004), as thinner fungal hyphae are physically more able to reach soil pores still filled with soil solution in dry conditions (Smith et al. 2010). In addition, plants could benefit from mycorrhizal fungi due to direct hyphal uptake of water, although so far this has only been shown for EcM fungi (Plamboeck et al. 2007; Lehto & Zwiazek 2011). Another reason for lower mycorrhization in wet habitats could be unsuitable conditions for mycorrhizal fungi (Helgason & Fitter 2009). Frequent waterlogging can induce anaerobiosis in soil, which inhibits fungal growth and the ability to colonize plant roots (Le Tacon et al. 1983). This might be the case in frequently waterlogged floodplain meadows, which are usually dominated by the generally non-mycorrhizal genus Carex (Muthukumar et al. 2004).

Plant species richness

The importance of mycorrhizal symbiosis in determining plant diversity has fascinated ecologists for decades (Zobel et al. 1997; van der Heijden et al. 2008; Kllronomos et al. 2011), yet the results have been highly variable, depending on plant communities or mycorrhizal types investigated.

In a temperate forest ecosystem, we found a negative association between community mycorrhization and plant species richness. Forests dominated by obligatorily EcM trees are known to harbour fewer plant species than AM-dominated ecosystems, and several underlying explanations have been proposed (Alieni et al. 1995; Dickie et al. 2014). One of the most likely explanations claims that EcM and ErM fungi produce proteolytic enzymes, allowing them greater access to organic N sources than AM fungi. This gives an advantage to ectomycorrhizal and ericoid mycorrhizal host species (Averill et al. 2014; Dickie et al. 2014), having an inherently smaller species pool (i.e. number of plant species with EcM and ErM) than that of AM host species (Brundrett 2009).

Conversely, arbuscular mycorrhization in forests, as well as across ecosystem types, was positively related to plant species richness and other diversity estimates. Earlier observations suggest that an increase in the abundance of AM trees is associated with higher field layer diversity (Newman & Reddell 1988), and a decrease in mycorrhizal fungal activity is associated with a decrease in richness of understorey vegetation (Zobel et al. 1999). AM symbiosis is thought to enhance plant diversity by promoting seedling recruitment (van der Heijden 2004; Koorem et al. 2012) or through a negative plant–soil feedback (Bever et al. 2010). In addition, AM may enhance plant species co-existence by amplifying intraspecific and balancing interspecific competition (Moora & Zobel 2010). AM fungal mycelium in the forest soil may act as an extention of the AM host plant root system, following the spatial heterogeneity of P concentration and presumably contributes to the maintenance of rich AM vegetation in EcM-dominated ecosystems (Koorem et al. 2014).

We found no relationship between community arbuscular mycorrhization and plant species richness in
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grasslands, although the current understanding that AM symbiosis is essential to maintain plant diversity originates largely from grassland systems (van der Heijden et al. 1998; Klironomos et al. 2011; Dostálek et al. 2013). However, other studies report that AM symbiosis decreases (Hartnett & Wilson 1999; O’Connor et al. 2002) or has no effect (Stampe & Daehler 2003; Rowe et al. 2009) on plant diversity in grasslands. These findings indicate that the impact of AM symbiosis on plant diversity in grasslands remains speculative and context-dependent. However, because all the aforementioned studies are local, coming from very different study systems, it is possible that the diversity of grassland types in this study was not extensive enough to encompass the full range of environmental gradients. Therefore, the role of AM symbiosis in grassland diversity patterns at larger scales remains to be studied (Hartnett & Wilson 2002).

Also, the arbuscular mycorrhization index might be too coarse to reveal the true small-scale impact of AM symbiosis in grasslands. Hart et al. (2003) postulated ‘coarse-scale effects’ and ‘fine-scale effects’ of AM fungi on plant coexistence. Coarse-scale effects apply in vegetation succession, where low abundance of fungal symbionts might limit host plants and hence growth of non-host plant species is favoured. Fine-scale effects (e.g. host/fungus specificity, AM fungal multifunctionality, feedback among plants and AM fungi, shared mycelial networks) apply if AM fungi are abundant and in contact with the roots of most plants. A positive relationship between AM fungal and plant diversity (van der Heijden et al. 1998; Hiiesalu et al. 2014) suggest that fine-scale effects may play an important role in grasslands. Consequently, increasing arbuscular mycorrhization is associated with higher plant diversity in conditions in which other mycorrhizal types are abundant, such as temperate forests with predominance of EcM trees and ErM understorey. In predominantly arbuscular mycorrhizal grassland vegetation, the variation in plant diversity might be driven through more specific processes not fully incorporated in the arbuscular mycorrhization index.

The plant community mycorrhization index helps to quantify the importance of mycorrhizal symbiosis at large scales. However, its explanatory power decreases with decreasing scale, at which more specific interactions might determine plant community structure. The community mycorrhization was consistent, regardless of whether a fine- or coarse-scale method was used. Nevertheless, it should be noted that calculating mycorrhization for regions with less studied floras should be considered with caution, as many plant species are not sufficiently investigated with respect to their mycorrhizal traits. Future research should determine the variations in mycorrhization and plant diversity, applying this approach to larger data sets, and possibly using directly measured soil characteristics to explain community mycorrhization. Incorporating the concept of mycorrhization to elucidate plant diversity could help to improve our knowledge of the significance of mycorrhizal symbiosis in structuring plant communities.

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References


### Supporting Information

Additional Supporting Information may be found in the online version of this article:

**Appendix S1.** Overview of plant communities used in the study.

**Appendix S2.** Arbuscular fungal colonization of *Festuca sabulosa*, *Geranium palustre* and *Sagina nodosa*.

**Appendix S3.** Example of calculating numerical coefficient for facultatively mycorrhizal plants.

**Appendix S4.** The results of coarse-scale analyses of plant community mycorrhization.

**Appendix S5.** Results of variation partitioning.

**Appendix S6.** Proportional abundance of plants with different mycorrhizal types and status in grasslands and forests.

**Appendix S7.** Partial correlations between general community mycorrhization, arbuscular mycorrhization and exponential Shannon diversity index.

**Appendix S8.** Partial correlations between general community mycorrhization, arbuscular mycorrhization and inverse Simpson dominance index.

**Appendix S9.** Partial correlations between general community mycorrhization, arbuscular mycorrhization and evenness (Alatalo index).