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The relationships among microbial parameters and the rate of organic matter mineralization in forest soils, as influenced by forest type

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ABSTRACT

Vegetation type influences the rate of accumulation and mineralization of organic matter in forest soil, mainly through its effect on soil microorganisms. We investigated the relationships among forest types and microbial biomass C (MBC), basal respiration (R_B), substrate-induced respiration (R_S), N mineralization (N_{min}), specific growth rate μ, microbial eco-physiology and activities of seven hydrolytic enzymes, in samples taken from 25 stands on acidic soils and one stand on limestone, covering typical types of coniferous and deciduous forests in Central Europe. Soils under deciduous trees were less acidic than soils of coniferous forests, which led to increased mineralizing activities R_B and N_{min}, and a higher proportion of active microbial biomass (R_S/MBC) in the Of horizon. This resulted in more extractable organic C (0.5 M K_2SO_4) in soils of deciduous forests and a higher accumulation of soil organic matter (SOM) in coniferous forest soil. No effect of forest type on the microbial properties was detected in the Oh horizon and in the 0–10 cm layer. The microbial quotient (MBC/C_{org}), reflecting the quality of organic matter used for microbial growth, was higher in deciduous forests in all three layers. The metabolic quotient qCO_2 (R_B/MBC) and the specific growth rate μ, estimated using respiration growth curves, did not differ in soils of both forest types. Our results showed that the quality of SOM in coniferous forests supported microorganisms with higher activities of β-glucosidase, cellobiosidase and β-xylosidase, which suggested the key importance of fungi in these soils. Processes mediated by bacteria were probably more important in deciduous forest soils with higher activities of arylsulphatase and urease. The results from the stand on limestone showed that pH had a positive effect on microbial biomass and SOM mineralization.

Introduction

The ability of forest soil to store and to transform organic compounds is influenced by the quality of soil organic matter (SOM), which depends on whether the forest is coniferous or deciduous (Turbé et al. 2010). The chemical composition of SOM largely results from microbial transformation of tree litter, whose composition in turn influences the activity and diversity of soil microorganisms (Xu et al. 2006; Charro et al. 2010; Perez-Bejarano et al. 2010; Ushio et al. 2010). Understanding of the effects of vegetation on the processes of SOM transformation performed by soil microorganisms is thus important in order to assess the extent to which the soil will serve as a C sink or source (Vesterdal et al. 2013).

Various abiotic soil properties, mainly pH, the C:N ratio and the content of labile fractions of SOM, have been reported as having a significant influence on the differences in SOM turnover between coniferous and deciduous forests. Low values of pH in coniferous forest soils may explain the higher accumulation of organic matter, because acidifying compounds act as inhibitors of microbial activity (Malchair and Carnol 2009). The slow rate of SOM decomposition in coniferous forests was also linked to lower litter quality, characterized by somewhat higher values of the C:N ratio (Zhang et al. 2008). The relationship between the content of SOM and its turnover is not straightforward. Some authors reported significant correlations between SOM and microbial biomass and/or mineralizing activity (Chaer et al. 2009; Perez-Bejarano et al. 2010; Susyan et al. 2011). Others did not confirm these findings, which suggests that the varied chemical composition and structure of organic molecules may have masked these relationships (Sinha et al. 2009; Gömöryová et al. 2010). This hypothesis was supported by Xing et al. (2010) and De Marco et al. (2012), who analyzed tree litter using ^{13}C NMR spectroscopy and found that plant residues from coniferous forests contained more compounds resistant to microbial decomposition than residues from deciduous trees. The lower respiration and N mineralization in coniferous forest soil found by Xu et al. (2006), Kooijman and Smit (2009), Sinha et al. (2009) and by Charro

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et al. (2010) is in agreement with this assumption. According to Smolander and Kitunen (2002) and Perez-Bejarano et al. (2010), soluble organic compounds, which are more easily leached from the leaf litter of deciduous forests, influence microbial growth and activity, rather than the total content of SOM. Low molecular weight compounds, which form most dissolved C, are released into soil during depolymerization of SOM catalyzed by hydrolytic enzymes (Tian et al. 2010), thus enzyme activities are important for nutrient supply to microorganisms (Šnajdr et al. 2008). The content of SOM as well as differences in its properties decrease with the depth of the soil profile (Thoms et al. 2010), which results in more variable microbial activities in the upper horizons (Vesterdal et al. 2008; Prescott and Graystone 2013).

Although a lot of data about the effects of forest type on soil microorganisms have been collected, it is still unclear to what extent soil microbial properties are influenced directly by means of litter composition, and to what extent indirectly through different soil chemical properties in coniferous and deciduous forests (*e.g.* various pH or the content of labil C). The interactions among the litter composition and abiotic properties of a given stand make differentiation difficult. A detailed understanding of litter decomposition is important for appropriate forest management because this process regulates nutrient cycling, C sequestration and primary productivity in forest ecosystems (Wang et al. 2013).

The goal of this study was (i) to assess differences in microbial biomass and activities related to SOM decomposition in the Of and Oh horizon and the 0–10 cm layer of coniferous and deciduous forests, and (ii) to discriminate the effect of forest type from the influence of basic soil properties on the studied microbial parameters.

The content of microbial biomass C (MBC) was determined as a measure of the ability of soil microorganisms to transform SOM into the labile pool, and substrate-induced respiration (R_S) as a proxy of the active microbial biomass. These parameters were expected to reflect the amount of easily decomposable substrate and suitable abiotic conditions for microbial growth. Basal respiration (R_B) and N mineralization (N_{min}) were chosen to estimate overall microbial activity covering most soil microorganisms. The activities of enzymes involved in the decomposition of starch (α-glucosidase), cellulose (β-glucosidase, cellobiosidase), chitine (chitinase) and xylane (β-xylosidase) were measured along with urease, phosphodiesterase and arylsulphatase to reveal what effect the type of vegetation has on selected steps in the C, N, P and S cycle.

Materials and methods

Study sites, soil sampling

Twenty-six stands were chosen in the eastern part of the Czech Republic (Supplementary Material 1), 13 with coniferous and 13 with deciduous forests, covering a wide range of the physico-chemical soil properties of the region (temperate climate zone) (Supplementary Material 2). The proportion of coniferous/deciduous species in the species composition, the criterion used for allocation of forests to either group, was at least 75% on all sites. The data about species composition were taken from the records of forest management plans of the local working-plan area and verified visually on site. The age of the researched stands was at least 60 years. There was a fully developed herb layer on all studied sites. Soil types on the stands were derived from the atlas of soils of the Czech Republic (Kozák et al. 2009). Each stand had an area of approximately 1 ha, within which ten sampling plots (25 × 25 cm) were randomly selected. Samples of the Of horizon were taken manually from the whole area after removal of litter, and samples of the Oh horizon were taken from a square of 10 × 10 cm marked out in the corner of the sampling plot. If the Oh horizon was too thin, the sample was taken from the whole area. The depth of both horizons was recorded. The layer 0–10 cm was sampled from the 10 × 10 cm square using a corer with a diameter of 4 cm in four replicates. Subsamples of individual horizons were pooled and transported to the laboratory. Sampling was performed in mid-spring within 2009–2011. Since the Oh horizon on site 20 was too thick to be sampled from the whole profile (more than 40 cm of which was the depth reached with a corer), the sample there was taken from the 10-cm layer without reaching the underlying mineral horizon.

Samples were sieved (<2 mm), frozen (−20 °C) and thawed for at minimum one week at 4 °C before the analyses started (ISO 10381-6 2009). The estimation of microbial parameters was carried out during the following seven weeks, when soils were stored at 4 °C. The estimation of enzyme activities was done after overnight thawing at 4 °C.

A part of the soil was dried at room temperature before the chemical analyses were done. Organic carbon was determined in the ground samples (<0.25 mm).

Chemical and physical analysis

Soil moisture was assessed as the weight decrease after 6 h drying at 105 °C. Water-holding capacity (WHC) was determined according to ISO 11274 (1998). The clay content was estimated using a pipette method based on the relationship between the sedimentation rate of soil particles and their size (ISO 11277 2009). The pH was determined in 1 M KCl soil suspension (1:5 w:v) according to ISO 10390 (2005). Organic C (C_{org}) was determined by means of the wet digestion of 1 g soil sample with 5 ml 0.27 M $K_2Cr_2O_7$ solution and 7.5 ml concentrated H_2SO_4 at 135 °C for 30 min. The concentration of C was determined spectrophotometrically at 585 nm with glucose as a standard (ISO 14235 1998). Total N (N_{tot}) was determined by treating soil (5 g) with 13 ml of the mineralizing mixture containing 10 g of selenium mixture (Merck) and 10 g salicylic acid in 1 l of concentrated H_2SO_4. After 12 h, H_2O_2 (30%) was added, and the samples were mineralized for 2 h at 420 °C. Soil N was determined after distillation into boric acid (Novozamsky et al. 1983). The coefficient 1.72 was used to convert C_{org} into SOM (Baldock and Nelson 2000). Macroelements were determined after their extraction from soil with the Mehlich III extraction solution (Mehlich 1984). Briefly, soil sample (10 g) was weighed into a sealable plastic container, 100 ml of the extraction solution (Mehlich III) was added and the suspension was shaken for 10 min. After extraction the suspension was centrifuged for 5 min at 3500 rpm. Potassium was determined by atomic emission spectrometry, Ca and Mg by atomic absorption spectrometry. To avoid the spectral interference during determination of Ca and Mg the extract was diluted with lanthanum. Phosphorus was determined by UV spectrophotometry at wavelength 750 nm as phosphomolybdic blue (Murphy and Riley 1962). For estimation of cation exchange capacity (CEC), 10 g of dry soil sample was weighed into the PE container, 100 ml of $BaCl_2$ solution (0.1 M) was added and the suspension was let stand for 16–18 h. After 2 h on a shaker the extract was filtered through the medium dense filter. First part of filtrate was removed. The exchangeable cations (Ca, Mg, K, Na, Al, Fe, Mn) were determined by optical emission spectrometry with inductively coupled plasma. The exchangeable acidity (Al + H) was determined by titration. CEC was calculated as the sum of the concentrations of the exchangeable cations and the exchangeable acidity (Mehlich 1953). Macroelements and CEC were estimated only in soil from the 0–10 cm layer.