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Spatial and temporal variability of biological indicators of soil quality in two forest catchments in Belgium

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A B S T R A C T

Biological indicators, measurements based on the presence and activity of soil organisms, are increasingly being considered in assessments of forest soil quality. In addition to chemical indicators, such as soil organic carbon and pH, biological indicators can provide an early diagnosis of changes in soil quality and processes in response to environmental change and forest management actions. We investigated the spatial and temporal variability of selected bio-indicators in the forest floor of two catchments over three years. We further evaluated the sensitivity of these indicators to changes in the dominant tree species following reforestation and natural regeneration. Indicators of microbial abundance and activity (microbial biomass, potential respiration) and carbon maintenance costs (metabolic quotient) were higher under young spruce and mixed deciduous stands than pure stands of oak and beech. Our results indicate a greater microbial activity in autumn but a wider range of carbon substrate utilisation in spring. Assessment of seasonal differences in bio-indicator values is vital for the evaluation and planning of long-term studies and the development of reference values for forest soils in Belgium. Our results highlight the usefulness of these bio-indicators in identifying changes in soil quality, particularly in response to management activities, at small spatial scales.

1. Introduction

Soil biological indicators are increasingly being considered in assessments of forest soil quality in the context of sustainable forest management and to assess the impacts of forest management actions (Lemanceau et al., 2016; Stone et al., 2016). Biological indicators typically include soil fauna (earthworms, nematodes, collembola; Cluzeau et al., 2012; Pérès et al., 2011) and microorganisms (Raiesi and Beheshti, 2015), through measures of their abundance (biomass), activity (processes) and diversity (community structure). The importance of soil microorganisms in driving major biogeochemical processes (Delgado-Baquerizo et al., 2016; Schloter et al., 2003; van der Heijden et al., 2008) and their rapid growth under favourable conditions make them useful indicators of soil quality (Bauhus and Khanna, 1999; Kennedy and Stubbs, 2006), as they can provide early indications of changes in the functioning of soils (Kennedy and Stubbs, 2006; Schloter et al., 2003). In recent years, forest management practices have shifted from a reliance on single species stands to patches of different tree species or mixed stands (Bauhus et al., 2013). This increased heterogeneity of forest stands may increase the resilience of soil functioning at the landscape scale (van der Plas et al., 2016). Thus, for the sustainable management of forest soils it is essential to be able to identify changes in soil functioning at the scale of the forest patch.

Soil quality has been defined as the continued capacity of the soil to function as a vital living system, within ecosystem and land use boundaries (*sensu* Karlen et al., 1997). It depends on the physical, chemical and biological properties and processes in the soil as well as their interactions (Karlen et al., 2003), and its characterisation requires indicators that are sensitive to changes in vegetation and hydrology, and that are related to soil functions and processes (Allen et al., 2011). Traditionally in forestry, tree biomass and growth were used as the main indicators of soil quality (Schoenholtz et al., 2000). Soil physical properties, such as texture and bulk density, and chemical properties, such as total organic content, pH and nutrient levels, are also widely used (Zornoza et al., 2015). However, changes in these parameters over time are relatively slow (Kirschbaum, 2000) and thus they provide limited indication of short-term changes in soil quality. Bio-indicators, such as microbial biomass and activity, can be more easily related to soil processes than physical and chemical indicators because it is the direct activity of microorganisms that drives many soil functions and processes (Table 1).

The quality and quantity of plant residues in soil are key drivers of the biomass and activity of soil microbial communities, as heterotroph microorganisms rely on plant-derived carbon as their energy source

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Table 1
Description of each soil quality bio-indicator assessed in the study and the details on the ecosystem function to which it relates. The mean and 75% percentiles of each bio-indicator across both forest catchments is also included.

Bio-indicator	Description	Relation to ecosystem functioning	Mean (25 and 75 percentile)
Microbial biomass carbon (MBC, mg kg^{-1})	Weight of carbon in soil microorganisms	Decomposition, nutrient cycling, soil fertility, soil structure	3461.0 (2719.2, 4633.2)
Potential respiration (PR, μg g^{-1} h^{-1})	Soil CO_2 emissions from microbial respiration	Decomposition	5.3 (3.9, 7.3)
Microbial biomass nitrogen (MBN, mg kg^{-1})	Weight of nitrogen in soil microorganisms	Rates of biogeochemical cycling and indicator of soil fertility	542.7 (411.0, 734.2)
Net nitrogen mineralisation (Nmin, mg kg^{-1} d^{-1})	Net release of inorganic N from soil organic matter	Biogeochemical cycling, soil fertility	9.3 (6.4, 12.0)
Microbial quotient (qmic, MBC:Corg)	Soil microbial biomass carbon per unit of soil carbon content (Organic carbon availability/quality)	Decomposition	0.9 (0.7, 1.2)
Metabolic quotient (qCO_2, μg C-CO_2 mg^{-1} MBC h^{-1})	Soil respiration per unit of microbial biomass carbon	Indicator of the efficiency of carbon utilisation	1.5 (1.1, 1.9)
Metabolic potential (MP, %)	Percentage of 31 carbon substrates used by soil bacteria (Metabolic diversity of soil bacteria)	Decomposition	48.4 (37.9, 58.1)

(Zak et al., 2003). The quantity and biochemical composition of litter and root exudates can vary considerably between tree species (Binkley, 1996; Binkley and Giardina, 1998; Hättenschwiler et al., 2005; Scheibe et al., 2015), such that nutrient availability for microorganisms can be very different in forest stands of varying species composition and stand age (Brant et al., 2006; Carnol and Bazgir, 2013; Grayston et al., 1996; Wardle et al., 2004). Mixed species stands may provide a greater variety and a reduced temporal variability in nutrient availability for microorganisms (Hättenschwiler et al., 2005) thereby improving soil quality compared to single-species stands.

The use of biological indicators for assessing soil quality is relatively new, partly due to a better understanding of the functional role of soil organisms in soil processes (Stone et al., 2016). Their use has additional challenges to more conventional indicators because bio-indicators are more variable in space and time than physical and chemical indicators (Doran and Zeiss, 2000; Parkin, 1993). As such, their novelty and inherent variability means that currently there is a lack of base-line values for many indicators, especially in forest ecosystems, and there is a clear need to define reference values under different environmental conditions and locations (Cluzeau et al., 2012; Lemanceau et al., 2016; Pulleman et al., 2012). In addition, it is important to assess the temporal scales over which bio-indicators respond to changing environmental conditions in order to decide on sampling strategies and data interpretation (Thoms and Gleixner, 2013).

There is a wealth of studies looking at individual indicators in isolation, yet little is known on the relative abilities of different indicators to detect changes in soil quality on similar soil types over relatively short time frames with limited disturbances (Bending et al., 2004). Here we assess a range of bio-indicators (Table 1) measured over three years in two forest catchments, located on the same geological substrate. Although molecular indicators of biodiversity can provide important information on soil quality (Stone et al., 2016), here we focused on 'classical' indicators frequently used in monitoring networks and specifically selected for Wallonia because of their relevance, ease of use, limited cost and interpretability (Malchair et al., 2010; Ritz et al., 2009). Earthworms were not included, because of their limited presence in the acid sites studied. The objectives of this work were to (i) assess the spatial variability of the bio-indicators and to evaluate the sensitivity of selected bio-indicators to changes in the dominant tree species following reforestation and natural regeneration; (ii) assess the temporal variability of the bio-indicators; and (iii) provide data for the development of reference values for forest soils in Belgium.

2. Materials and methods

2.1. Study sites

The study sites were located in two, *ca.* 80 ha, forested catchments (Waroneu, https://data.lter-europe.net/deims/site/LTER_EU_BE_17 and La Robinette, https://data.lter-europe.net/deims/site/LTER_EU_BE_27), in the state forest of Hertogenwald, Belgium (50°33′ N, 6°04′ E). The geological substratum of both catchments consists of quartzites, quartzo-phyllades and Revinian phyllads, covered with acidic brown soils. The structural B-horizon of the soils rests on a horizon with a silty texture called "fragipan", which impedes drainage. The soil is acidic, of type moder to dysmoder and presents a low base saturation. At Waroneu, in the early 1980s, forty percent of the area was covered with hardwoods (beech, *Fagus sylvatica* L., sessile oak, *Quercus petraea* (Matt.) Liebl., and birch, *Betula pendula* ROTH.) and 60% was covered with Norway spruce (*Picea abies* (L.) KARST.), planted in the 1930s. By 2011, the proportions had changed to 38% spruce, 45% deciduous and 17% open areas. Waroneu was limed in 1992 with 3 T/ha dolomite lime and 200 kg/ha potassium sulphate. La Robinette was initially covered with Norway spruce, but following windthrow in the 1990s, forest cover was significantly reduced and a mixed Sitka spruce (*Picea sitchensis* (Bong.) Carrière)-Norway spruce plot was established (hereafter "mixed spruce"). A further 22 ha were clear-cut in 1996, and in 1998 common alder (*Alnus glutinosa* (L.) GAERTN.), pedunculate oak (*Quercus robur* L.), silver birch (*Betula pendula* ROTH.), goat willow (*Salix caprea* L.), noble fir (*Abies procera* Rehd.) and rowan (*Sorbus aucuparia* L.) were planted in alternate rows, spaced by 2.5 m apart within four fenced plots of 2 ha. Alder, rowan, birch and oak were also planted within the catchment (with individual protection against deer damage). Spontaneous regeneration of Norway spruce resulted in thickets across the catchment, with trees in the same age class as the planted deciduous species (Carnol and Bazgir, 2013).

Six 30 by 30 m plots were established in each catchment representing the dominant tree species and drainage combinations (Table 2). The dominant tree height in each plot was defined as the mean height of the three trees with largest circumference at 1.30 m. Measurements of tree basal area and wood volume in each plot were performed according to the permanent inventory of forest resources in Wallonia (Alderweireld et al., 2015). Briefly, the circumference (C; 1.30 m) and the total tree height (VERTEX IV) were measured in circled plots (centred in the middle of the 30 × 30 m square) of 36 m, 18 m, and 9 m diameter for trees with C > 120 m, 70 < C < 119, 20 < C < 69, respectively. In mixed deciduous plots, all trees with C > 20 cm were measured. Wood volume was estimated following Vallet et al. (2006).