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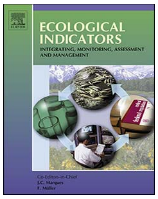

Research paper

Topographic wetness index explains soil moisture better than bioindication with Ellenberg's indicator values

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ABSTRACT

Topography is an important determinant of soil moisture (SM) distribution and thus drives the functioning of terrestrial ecosystems, including vegetation composition and structure. To assess soil water spatial variability, a number of indices have been used. In this study, we compared the ability of the topographic wetness index (TWI) and Ellenberg's indicator values (EIV) for moisture to explain the spatial variation of SM in central European forests. Further, we tested the potential heat load (HL) and soil water capacity (SWC) as additional factors that could improve the regressions between TWI and SM as well as EIV and SM. TWI was calculated using 10 different flow routing algorithms. The average EIV for moisture was calculated on the basis of the presence/absence of plant species. We observed that the flow routing algorithms explain SM variability better than the average EIV. The strongest relationship between TWI and SM was obtained by the MFD-md algorithm. The inclusion of SWC increased the explanatory power of both TWI and EIV. On the other hand, HL did not improve the regressions. The relative increase in the explanatory ability by SWC was particularly pronounced in case of EIV. We interpreted this to be a result of the fact that EIV reflect the synergistic effect of multiple environmental gradients on plant distribution. TWI calculated by any of the flow routing algorithms remains a better explanatory factor of SM than EIV, even if the latter was enhanced by the addition of SWC.

1. Introduction

Soil moisture (SM) is one of the most important environmental factors that determines vegetation composition, structure and functioning of terrestrial ecosystems. Knowledge about the distribution of soil water is therefore crucial for hydrological, ecological, agricultural and environmental studies. Because the evaluation of the SM effect on vegetation is challenging (Kopecký and Čížková, 2010; Häring et al., 2013), many indicators and physical-based hydrological models are used to assess water conditions.

Among the available methods, direct measurements of SM, performed during different seasons according to the research objectives and climatic zones, are often used (Western et al., 1999; Chaplot and Walter, 2003; Zhu et al., 2014). A direct topsoil measurement performed in the driest period of the growing season is relevant to assess the effects of SM on species composition in temperate climate (Güntner et al., 1999; Gilbert and Lechowicz, 2004; Gazol and Ibáñez, 2010; Tölgyesi et al., 2014) as well as forest floor biomass productivity (Axmanová et al., 2012). In the case of vegetation studies, the susceptibility of plants to periodic or occasional drought is generally more important for their long-term performance than their tolerance to occasional periods of high SM (Schaffers and Sýkora, 2000; Huxman et al., 2004; Häring et al., 2013; Vicente-Serrano et al., 2013). However, in a temperate climate, the substantial variation in SM during the growing season is caused by occasional rainfall. Therefore, to obtain comparable results in studies examining vegetation response to SM, the field measurements should be performed within a relatively short period without rainfall (Axmanová et al., 2012). This methodological restriction directly limits the area and time duration of a potential study.

In order to overcome the abovementioned limitations, soil–water relationships can be modelled as a function of soil depth and texture; for example, soil water capacity (SWC: the difference between maximum water content at field capacity and water content at the plant's permanent wilting point) can be assessed (Jamagne et al., 1977; Bergès and Balandier, 2010; Szymura et al., 2014). An alternative method is bioindication based on Ellenberg's indicator values (EIVs) for vascular plants (Ellenberg et al., 1992), which have been widely used in Europe for indications of primary environmental gradients, including SM (Diekmann, 2003; Axmanová et al., 2012; Merunková and Chytrý, 2012; Erdős et al., 2013; Möckel et al., 2016). In this system, the

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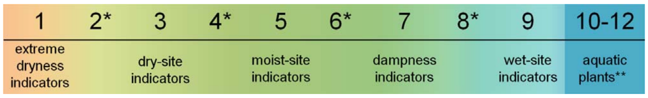

Fig. 1. The Ellenberg's indicator values scale for moisture indicators (Ellenberg et al., 1992) * intermediate value.

realised ecological optima of plant species are expressed as ordinal numbers. The ranking of EIV for moisture are presented in Fig. 1. For bioindication purposes, the average value of ranks weighted by species absence/presence or cover is calculated for the sampling plot and can be considered as a proxy for a particular environmental gradient (Diekmann, 2003). It has been found that in the case of EIV for moisture, the lowest values of SM give better correlations with the average EIV, suggesting that plant susceptibility to drought is more important than its tolerance to occasional periods of high SM (Schaffers and Sýkora, 2000). EIVs were originally designed for central Europe, but due to their usefulness, they have been adapted to other regions of Europe and applied across a wide range of disciplines, e.g. vegetation ecology, forestry and nature conservation. However, the Ellenberg system has been criticised for several reasons. The ranks are not systematically derived from field measurements, but rather from results stemming from plant ecologist field experiences (Diekmann, 2003). Furthermore, EIVs for particular traits are internally correlated, e.g. forest plants typically found in base soils often occur in dry habitats (Ewald, 2003). For short gradients, the mean EIVs do not vary considerably and might be more affected by random fluctuations in species composition than by the environmental gradient (Diekmann, 2003). However, ecologists using the mentioned field methods are obligated to perform expensive and time-consuming fieldwork, requiring equipment and professional knowledge regarding plant identification. Moreover, such data collection offers scattered, plot-scale information, without covering the whole study area and is generally restricted to limited areas (Schmidtlein, 2005; Möckel et al., 2016).

Furthermore, various terrain-based models have been proposed for the quantification of SM. Terrain indices are simple tools derived from a digital elevation model (DEM) and can be efficiently applied at a range of spatial scales (Moore et al., 1991; Gruber and Peckham, 2009; Zhu and Lin, 2011). The most frequently used index is the topographic wetness index (TWI) proposed by Beven and Kirkby (1979). TWI has been frequently tested with measured SM and other soil properties, revealing high correlations (McKenzie and Ryan, 1999; Güntner et al., 1999; Western et al., 1999; Chaplot and Walter, 2003; Zhu et al., 2014). Further, it has been applied in ecological studies devoted to vegetation (Moeslund et al., 2013b; Buchanan et al., 2014; Pielech et al., 2015; Alexander et al., 2016). TWI quantifies the tendency of soil water distribution, which is affected by topography. The index is determined as follows:

$$\mathrm{TWI} = \ln\left(a/\tan\beta\right),$$

where a is the specific catchment area (SCA): the local upslope area draining through a certain point per unit contour length, which is equal to a certain grid cell width, and β is the local slope (Beven and Kirkby, 1979; Sörensen et al., 2006; Gruber and Peckham, 2009). SCA can be evaluated in multiple ways (Gruber and Peckham, 2009; Zhou et al., 2011; Wilson, 2012). Different SCA algorithms can be generally grouped into two categories: single flow direction (SFD) and multiple flow direction (MFD) algorithms, depending on how the potential water flow is apportioned between cells in a DEM grid. SFD algorithms show no divergence in flow direction and are restricted to movement in a downhill direction from one cell to another at a time. MFD algorithms represent a divergence in flow direction, where the flow line is spread to several (two or more) adjacent cells according to the downslope gradient.

The ability of TWI to explain SM distribution patterns is limited because soil humidity does not depend on topography alone. It is also influenced by soil water redistribution, radiation, heterogeneity of soil properties and vegetation cover (Zhu and Lin, 2011; Moeslund et al., 2013a). Moreover, different TWI algorithms produce various results (Zhou and Liu, 2002; Schmidt and Persson, 2003; Wilson, 2012; Gruber and Peckham, 2009; Zhou et al., 2011; Buchanan et al., 2014) with varying credibility for ecological studies in particular landscapes (Güntner et al., 1999; Sörensen et al., 2006; Kopecký and Čížková, 2010; Rampi et al., 2014). Particular TWI algorithms have been compared with the measured SM; however, direct comparison of different algorithms in terms of their ability to explain SM is scarce (Park et al., 2009; Buchanan et al., 2014). The assessment of TWI performance, with respect to both vegetation traits and measured SM, is very rare (Zinko et al., 2005; Sörensen et al., 2006). In ecological studies, the performance of a specific (Häring et al., 2013; Moeslund et al., 2013b) or different (Kopecký and Čížková, 2010) flow routing algorithms has been evaluated by comparing with the bioindication approach. The effectiveness of SM bioindication based on EIV has rarely been tested (Schaffers and Sýkora, 2000; Diekmann, 2003; Szymura et al., 2014). In fact, it is unclear which of these two methods gives more realistic results.

Moreover, it has been suggested that for ecological and vegetation studies, the performance of TWI in SM modelling can be enhanced by incorporating other environmental variables, such as soil traits and proxies for evaporation, e.g. potential heat load (HL) (Kopecký and Čížková, 2010; Zhu and Lin, 2011; Buchanan et al., 2014). However, such procedures are not often applied.

Due to the great importance of SM estimation, knowledge about the effectiveness of different SM indicators is desirable, but the efficiency of different methods has rarely been directly compared. Our objective for this study was to compare measured SM with results of bioindication based on EIV and TWI computed with 10 different flow routing algorithms, commonly available in non-commercial software. This enabled a direct comparison between these two commonly applied methods to test which approach was better able to explain the SM spatial pattern. Secondly, we checked the extent to which the incorporation of additional variables representing evaporation (HL) and/or SWC enhanced the accuracy of SM indication of both tested methods.

2. Materials and methods

2.1. Study area

The study area was located in Sudety Mountains (Silesia, Poland, central Europe) (Fig. 2). The mean annual temperature is approximately 7.0 °C, and the mean total annual precipitation is approximately 634 mm. Approximately 40% of the precipitation falls during the summer quarter, when the average temperature is approximately 16.1 °C (Hijmans et al., 2005). Because of the presence of gullies eroded by water, the land relief was relatively complex. The soil types and depths varied with respect to topographic position, generally with cambisols found on plateaus and rankers on the slopes of hills. The vegetation consists of central European acidophilous and thermophilous oak forests. A detailed description of the vegetation and environmental conditions and visualisation of land relief can be found in Szymura et al. (2015) and Szymura and Szymura (2011).

2.2. Soil sampling, vegetation and environmental data

Eight sites were established in three mountain ranges, with 75 randomly located sampling plots. The plots were located at the bases of the slopes, on mid-slopes and on the summit plateaus of hills at altitudes of 300–580 m a.s.l. The position of each plot was determined using a GPS receiver with a differential correction. The number of plots per site varied from 7 to 14 (Fig. 2). The distance between plots within a particular site varied from 34 to 840 m, with an average of 288 m. All the plots were circular and covered an area of 250 m^2 (8.92 m in