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The role of phytoplankton diversity metrics in shallow lake and river quality assessment

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ABSTRACT

Ecological water quality problems are frequently connected to increment of phytoplankton productivity and overdominance of some phytoplankton species. Metrics that show monotonously increasing or decreasing tendencies along stressor gradients is recommended for ecological state assessment. Diversity metrics are influenced by various physical disturbances and show high within-year variability; thus, there is no agreement on the usefulness of these metrics as state indicators.

To test the usefulness of phytoplankton diversity in ecological state assessment we investigated the productivity–diversity relationships for lakes and rivers in the Carpathian Basin (Hungary). We demonstrated that the shape of productivity–diversity relationship depends on the investigated water body type. Regarding lakes, hump-shaped relationship was found for all computed metrics. Parallel with the increase in phytoplankton productivity values, diversity metrics showed monotonously increasing tendencies in rhithral and decreasing tendencies in large potamal rivers. We found no systematic relationship in the case of small lowland rivers.

Changes of diversity metrics calculated for species and functional groups showed similar tendencies within the types, only the slopes of regression lines differ each other.

The use of diversity metrics as ecological state indicators should be restricted to water body types where diversity decreases or increases monotonously with phytoplankton biomass. Regarding the lakes the use of diversity metrics is not recommended for ecological state assessment. In rhithral and large potamal river assessment, application of diversity metrics should be strongly considered. We demonstrated that diversity metrics can be useful components of multimetric indices proposed to use by the Water Framework Directive.

1. Introduction

Diversity is indisputably one of the most frequently used quantitative descriptors of communities. Diversity metrics are capable of describing system properties such as complexity, stability, and functioning of ecosystems (Hacker and Gaines, 1997); therefore, they became parts of several multimetric indices used for biological quality assessment (Hering et al., 2006; Stoddard et al., 2008; Carvalho et al., 2013). Applicability of diversity-based approaches needs a clear relationship between stressors and diversity metrics as response variables (European Commission, 2010). The nature of these relationships depends on the types of the anthropogenic disturbances and the responses of biological assemblages. In highly diverse natural assemblages human-caused environmental changes result in a decrease both in functional and species diversity (Gabriels et al., 2010). In the case of benthic diatoms, assemblages exhibit opposite responses to nutrient enrichment. Sonneman et al. (2001) demonstrated that sites with low nutrient concentrations were more species-rich than mildly enriched sites. In contrast Stevenson et al. (2008) found that species diversity of stream phytobenthos increases with phosphorus enrichment.

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These examples indicate that prior to the application of diversity metrics as ecological indicators the possibilities and limitation has to be investigated.

Due to central role in the aquatic food chain, phytoplankton is one of the biological quality elements that have to be monitored and assessed in Europe (European Commission, 2000). Based on quantitative and qualitative characteristics of phytoplankton several types of metrics have been elaborated and used in Europe: e.g. biomass, sensitivity/tolerance, composition and bloom metrics (Carvalho et al., 2013). Among the various water quality problems, eutrophication is indisputably a phenomenon that is closely related to phytoplankton issues. Nutrient enrichment coincides with enhanced phytoplankton production and impoverishment of floristic composition in lakes and large potamal rivers as well (Schmidt, 1994; Thorp et al., 1998; Wehr and Descy, 1998; Borics et al., 2007, 2013a,b; Stankovič et al., 2012). Several human induced alterations of the aquatic ecosystems result in the overdominance of some phytoplankton taxa (Naselli-Flores et al., 2003), which cause the decrease of diversity; therefore, diversity indices as state indicators seem plausible ecological assessment measures.

Phytoplankton diversity is influenced by the fluctuation of the resources (Sommer, 1984) and not by their absolute quantity; thus, diversity metrics cannot be studied by the traditional stressor–metric relationships, as it is proposed in technical guidance (European Commission, 2010), and has been done in the nutrients–sestonic chl-a and nutrients–sensitivity metrics relations. Instead of that, we investigated the changes of diversity along phytoplankton productivity, which is the most robust phytoplankton metric used for quality assessment (Carvalho et al., 2013).

Diversity is frequently studied as a function of productivity in the case of plants, animals or microbes (Adler et al., 2011; Grime, 2001; Chase and Leibold, 2002; Borics et al., 2012; Fridley et al., 2012; Skácelová and Lepš, 2014). Besides its theoretical importance, the shape of this relationship provides useful information on the practical use of diversity metrics as state indicators. Ecological state assessment should be based on clear relationship between the stressor(s) and the indicator metrics, i.e. the metrics should exhibit monotonously increasing or decreasing tendencies with increasing anthropogenic loads (European Commission, 2010). As phytoplankton productivity can be considered as a proximate measure of anthropogenic loads (Borics et al., 2013a,b), investigation of the productivity–diversity relationships, in an indirect way, inform us about the role of diversity in phytoplankton-based ecological state assessment.

The specific objective of this study is to investigate the usefulness of diversity metrics in phytoplankton-based water quality assessment, and to set the limits of their application. Thus, we tested the following hypotheses:

- The shape of the phytoplankton productivity–species diversity curves depends on water body types.
- Functional diversity metrics are more sensitive measures of productivity than those calculated for species data.

2. Material and methods

2.1. Data

For the analyses phytoplankton and chlorophyll-a (chl-a) data were provided by the Hungarian national water quality monitoring system. Data of 25 lakes and 71 rivers were used for the investigations. Lake samples were from the photic layer (2.5 × Secchi depth) of the lakes. In the case of the very shallow lakes (Z_{max} < 2 m) the whole water column was sampled. River samples were collected from the immediate surface layer of the thalweg. There were monthly samplings in the growing season. Samples were fixed by Lugol's solution on the spot. Algal counting was performed using the Utermöhl's settling procedure (Lund et al., 1958). Algae were identified to species level. Standard geometric models (Hillebrand et al., 1999) were used to calculate phytoplankton biovolumes. Sestonic chl-a as a surrogate measure of phytoplankton productivity was used to analyse phytoplankton productivity–diversity relationships.

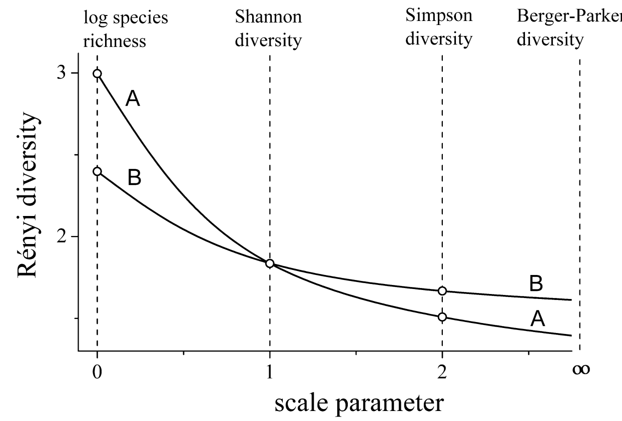

Fig. 1. Diversity profiles of the one-parametric Rényi diversity index family for two hypothetical assemblages, denoted by A and B. Vertical dotted lines denotes the particular values of the scale parameter (measured along the *x*-axis) which provides classical diversity index statistics, like number of species, Shannon, Simpson, and Berger–Parker index of diversity).

2.2. Applied metrics

We characterise the diversity of phytoplankton by four diversity indices (species richness, Shannon index, Simpon index, Berger–Parker index); each of these indices is a member of the Rényi diversity index family Eq. (1) (Rényi, 1961; Tóthmérész, 1998). This is a so-called one-parametric diversity index family: the diversity of an assemblage is characterised by a (scale-dependent) diversity profile instead of a numerical value (see Fig. 1). By increasing the value of scale parameter (α), the contribution of abundant species to the diversity of the assemblage increases, and the contribution of rare species decreases. This is a solution of the classical index choice problem: one may wish the index to be sensitive to the composition of the abundant species but relatively indifferent to that of the rare ones (Peet, 1974). Diversity profiles can be used in a graphical form to visualise the diversity relations of an assemblage as shown in Fig. 1 for the assemblages A and B based on Rényi diversity index family (Tóthmérész, 1995). We have used the following α values: 0, 1, 2, ∞. When the value of the scale parameter is 0, then the value of the Rényi diversity is the logarithm of the number of species Eq. (2). It is extremely sensitive to the presence of rare species: a species present as a single individual has the same contribution to HR(0) as the most abundant species. When the value of the scale parameter is 1, Rényi diversity is identical to the Shannon index of diversity (Shannon, 1948) Eq. (3). It is less sensitive to the rare species than HR(0). When the value of the scale parameter is 2, the Rényi diversity is equivalent to the Simpson diversity Eq. (4), and it is more sensitive to the frequent species than to the rare ones. HR(∞) is the logarithm of the relative abundance of the commonest species, and ignores the others Eq. (5); it is usually mentioned as Berger–Parker diversity (Berger and Parker, 1970).

$$\mathrm{HR}_{\alpha} = \frac{1}{1-\alpha} \log \sum_{i=1}^{S} p_i^{\alpha} \quad \text{where } \alpha \geq 0 \text{ and } \alpha \neq 1, \tag{1}$$