Limnologica 67 (2017) 20–26

Contents lists available at ScienceDirect

Limnologica

journal homepage: www.elsevier.com/locate/limno

The typology of Polish lakes after a decade of its use: A critical review and verification

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ARTICLE INFO

Keywords:
Lake types
Typology
Abiotic criteria
Water Framework Directive
Annex II

ABSTRACT

The abiotic typology of Polish lakes, compliant with the requirements of the EU Directive 2000/60/EC (Water Framework Directive), was developed in 2004 under the commission of the Ministry of the Environment. Based on the combination of the obligatory typological criteria from Annex II of the WFD and one additional factor, all Polish lakes larger than 0.5 km^2 were assigned to 13 abiotic types. This typology reflected the diversity of abiotic conditions and was assumed to be tested and validated for its ecological relevance on the basis of biological data obtained thereafter. The first ten years of its application proved the high usability of this typology for lake classification of ecological status on the one hand, while on the other hand, it allowed to gather experiences and to detect deficiencies and errors considered in the verification process. In 2015, the abiotic typology of Polish lakes was verified. Above all, verification involved an analysis of the justification of lake eco-regional division and refinement of water stratification criteria, as these two issues have been questioned in the current typology. Data on physicochemical properties, phytoplankton, macrophytes and benthic diatoms obtained from the state lake monitoring conducted in the years 2007–2013 were used to verify the ecological relevance of the typological criteria. Typological criteria used in the verified lake typological scheme were essentially the same as those used in 2004. However, the number of lake types has been reduced from the previous 13 to seven, mainly due to the withdrawal from the lake division based on eco-regions. Moreover, the more specific criteria for determining the water mixing type were established. The new lake typology is expected to be implemented in routine monitoring and water management in Poland in the forthcoming River Basin Management Plan 2021–2027.

1. Introduction

Distinguishing ecotypes that describe the diversity of physicochemical and hydromorphological conditions and of associated biological assemblages, i.e. phytoplankton, macrophytes, phytobenthos, zoobenthos and fish, expected under circumstances of no more than 'very minor' anthropogenic distortions, is one of the prerequisites when establishing modern bioassessment systems as required by the Water Framework Directive (2000/60/EU; WFD). For such discriminated ecotypes, the type-specific reference conditions, against which the ecological status of a waterbody is determined, have to be derived. During the first decade of the provision of the WFD, the member states have elaborated and implemented typologies of their surface waters, including lakes (Buraschi et al., 2005; Free et al., 2007; Irvine et al., 2002; Pilke et al., 2002; Solheim, 2002; Wasson et al., 2002; Nykänen et al., 2005; overview also in Kolada and Soszka, 2004). The common practice was to develop a typology based on the variety of abiotic parameters prior to the biotic parameters, which in most European countries, was determined by the lack of appropriate biological data in the early 2000s. It was clear, however, that typological systems based on pre-defined abiotic criteria will hardly fully overlap with the diversity of biological assemblages, and the criteria and level of habitat division appropriate for one biological community are usually inappropriate for another (Pollard and Hauxham, 1998). Therefore, the European countries were challenged to verify their typological schemes based on the biological data collected thereafter.

Water typology, although not a new concept, returned as a weighty issue when it came to establishing environmental objectives for waters, as required by Article 4 of the WFD. In general, environmental objectives for surface waters aim at achieving a good status, both ecological and chemical. In simple terms, they usually reflect values of the assessment criteria anticipated in good environmental conditions that an ecosystem should achieve. Such an approach, although extremely anthropocentric and administrative, provides a pragmatic guideline for water management and protection. However, to ensure a high effectiveness of actions undertaken, environmental goals need to be

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http://dx.doi.org/10.1016/j.limno.2017.09.003
Received 5 June 2017; Received in revised form 10 September 2017; Accepted 19 September 2017
Available online 12 October 2017

achievable in given environmental circumstances (abiotic conditions). Inadequately set environmental goals may lead to measures being improperly designed and resources being unjustifiably spent. If this is the case, the environmental target may be missed not because no action was undertaken, but because the target was unrealistic.

The abiotic typology of Polish lakes, compliant with the WFD requirements, was developed in 2004 under the commission of the Ministry of the Environment. Based on the combination of the obligatory typological criteria from the annex II of the WFD and one additional factor (Schindler's ratio, SR; Schindler, 1971), all Polish lakes larger than 0.5 km^2 were assigned to 13 abiotic types (Kolada et al., 2005). The typological criteria were set as follows: two physico-geographical regions (North European Plain and Eastern Baltic-Belarusian Lowlands provinces), one class of altitude (< 200 m above sea level), one size class (> 0.5 km^2), two classes of calcium concentrations (soft-water lakes with Ca ≤25 mg l^{-1} and hard-water lakes with Ca > 25 mg l^{-1}), two classes of Schindler's ratio values (lakes less exposed to the influences of the catchment with SR ≤2 and lakes more exposed to degradation with SR > 2) and two types of water mixing (stably stratifying during summer period = stratified and permanently mixed = polymictic) (Table 1). This typology system utilised a set of predefined abiotic criteria without direct reference to the diversity of biological assemblages (an *a priori* typology). Its ecological relevance was assumed to be tested and validated in the future on the basis of biological data, which in 2004 were lacking. During much of the past decade, appropriate data were collected within the State Monitoring Programme, including biological data on phytoplankton, macrophytes and phytobenthos (Kolada et al., 2016). Moreover, the first 10 years of the use of the abiotic typology allowed to gather experiences and to detect deficiencies and errors considered in the verification process.

In 2015, the abiotic typology of Polish lakes was verified. Above all, verification involved the analysis of justification of lake ecoregional division and refinement of water stratification criteria, as these two issues have been mostly questioned in the 'old' typology. Subsequently, the new lake typology was developed and is expected to be implemented in routine monitoring and water management in Poland in the fourth River Basin Management Plan 2021–2027.

Table 1
Characteristics of the abiotic lake types according to the typology developed in 2004 (Kolada et al., 2005); for geographic units see Fig. 1; #lakes indicates number of lakes of a surface area greater than 0.5 km^2 identified within a type in the country; #reference lakes indicates number of lakes assessed as in high ecological conditions in the study database; SR – Schindler's ratio (Schindler, 1971), S – stratified, NS – non-stratified.

Type code	Description	# lakes[a]	# reference lakes
Province: North European Plains (Western European Unit)			
Lake districts on postglacial deposits			
1a	lowland, Ca ≤25 mg l^{-1}, S	15	2
1b	lowland, Ca ≤25 mg l^{-1}, NS	12	3
2a	lowland, Ca > 25 mg l^{-1}, SR ≤2, S	112	16
2b	lowland, Ca > 25 mg l^{-1}, SR ≤2, NS	11	3
3a	lowland, Ca > 25 mg l^{-1}, SR > 2, S	254	11
3b	lowland, Ca > 25 mg l^{-1}, SR > 2, NS	296	11
4	lowland, Ca > 25 mg l^{-1}, influenced by the marine waters, all NS	10	0
Province: Eastern Baltic-Belarusian Lowlands (Eastern European Unit)			
Lake districts on postglacial deposits			
5a	lowland, Ca > 25 mg l^{-1}, SR ≤2, S	97	10
5b	lowland, Ca > 25 mg l^{-1}, SR ≤2, NS	7	0
6a	lowland, Ca > 25 mg l^{-1}, SR > 2, S	133	5
6b	lowland, Ca > 25 mg l^{-1}, SR > 2, NS	80	4
Lake district in the Polesie sub-province			
7a	lowland, Ca > 25 mg l^{-1}, S	5	1
7b	lowland, Ca > 25 mg l^{-1}, NS	12	5

[a] According to the 1st River Basin Management Plan 2009–2015 (n = 1044).

2. Material and methods

Poland is relatively rich in lakes, with about 7000 lakes of a surface area > 0.01 km^2. Similar to other European countries with numerous lakes, a lake surface area of 0.5 km^2 is assumed as a threshold to characterise a water body as a "discrete and significant element" (Annex II to the WFD). This allows for a rational planning of the monitoring and the water management. The list of significant water bodies (SWBs) in Poland comprises about 1000 lakes of an area greater than 0.5 km^2. To be precise, based on the historical sources, 1042 lake SWBs were determined in 2004 (Kolada et al., 2005), while in 2015, 1017 lake SWBs were identified based on the reference database 'Hydrographic Map of Poland' at a scale of 1:10,000 (MPHP10; Barszczyńska et al., 2013). All these lakes are located in the lowlands, within two physico-geographical units (Fig. 1), and the majority of them are hard-water ecosystems, while only 27 are soft-water lakes (Table 1).

To verify the validity of the typological criteria, the data obtained from the lake monitoring surveys conducted in the years 2007–2013 were used. In the monitoring dataset, data from a total of 483 lakes and 830 lake-surveys (including repeated surveys) were collected. Of these lakes, 71 were selected (no replicate surveys were included; for lakes surveyed more than once, the most recent study was used), which represented a high ecological status based on all the assessment criteria. The latter were considered lakes with non-impacted biological assemblages and were further referred to as reference lakes. Out of the 71 reference lakes used to verify the typology, 46 belonged to the Western Unit (current types 1, 3 and 4–Western types) and 25 to the Eastern Unit, including the Polesie sub-province (5, 6 and 7 – Eastern types), while 45 were stratified (all types indexed with 'a') and 26 were polymictic ecosystems (indexed with 'b'). Four of the reference lakes were soft-water ecosystems, with a calcium content below 25 mg l^{-1}, while all other lakes were highly alkaline hard-water lakes (Table 1). Data on hydromorphological and water physicochemical parameters for all the reference lakes were available, while biological data on phytoplankton were available for 65 lakes, data on macrophytes for 47 lakes and data on phytobenthos for 31 lakes (Table 2).

Lakes were sampled for physicochemistry and phytoplankton three or four times during the vegetation season, from March to October. During the summer stagnation period, integrated water samples were collected from the epilimnion layer and in spring and autumn, from the euphotic layer. In non-stratified lakes, integrated samples were taken from the layer between 0 and 5 m. The quantitative analyses of phytoplankton followed the Utermöhl method (1958). Phytoplankton biomass was determined using a harmonised national protocol by Hutorowicz (2009). For macrophytes, lakes were investigated once a year, at the peak of the vegetation season (from mid-June to mid-September), using the unified field survey procedure based on the belt transect method (Ciecierska and Kolada, 2014; Kolada et al., 2014a). Within the phytolittoral of each lake, the maximum colonisation depth, the mean vegetation coverage and the relative cover of all the aquatic and emergent plant communities were determined. For phytobenthos (benthic diatoms), lakes were sampled once a year using a standardised procedure (Picińska-Fałtynowicz and Błachuta, 2010; Kelly et al., 2014). Samples were usually taken in late summer/autumn from one sampling site, located at the edge of the littoral zone, from emerged macrophytes or stones, at least 30 cm below water level. Diatom valves were identified to the species level or lower and were counted until about 400 valves were enumerated.

Firstly, canonical community ordination techniques (ter Braak and Šmilauer, 2002) were applied to recognise main gradients in biological data matrices in non-disturbed conditions. The detrended correspondence analysis (DCA) was used to estimate the length of a gradient in the standard deviation of primary data turnover, i.e. summer biomass of 406 phytoplankton taxa (species and genera; data square-root-transformed; rare species downweighted), relative abundance of 90