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Opinion Paper

Use of phytoplankton assemblages for monitoring ecological status of lakes within the Water Framework Directive: the assemblage index

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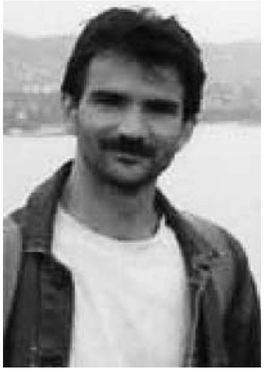

Judit Padisák (1955) graduated at the Eötvös Loránd University of Budapest (1979; PhD: 1990; habilitation: 1998; DSc: 1999), biologist, Professor of Limnology. She is dealing with phytoplankton community ecology since the beginning of her professional career; editor of *Hydrobiologia*, guest editor of five special volumes (*Developments in Hydrobiologia* 81, 100, 150, 170 and *Hydrobiologia* 506–509). Co-authors of this paper are her ex- or recent students. **Gábor Borics** (1967) graduated at the University of Debrecen (1991; PhD: 2002), biologist, working on biological monitoring. He focused his studies on algal flora and assemblages of small lakes (e.g. *Hydrobiologia* 424: 79–90; 502: 145–155). **István Grigorszky** (1967) also graduated at the University of Debrecen, (1991; PhD: 1998), biologist, lecturer. Most of his works concern taxonomy and ecology of dinoflagellates (e.g. *Hydrobiologia* 502: 349–355; 506–509: 203–207 and 209–212). **Éva Soróczki-Pintér** (1977) graduated at the University of Veszprém (2001), teacher of chemistry and environmental science; assistant lecturer. She performed experiments on sinking resistance of phytoplankton (*Hydrobiologia* 500: 243–257) and is working on implementation of the Water Framework Directive in Hungary.

Abstract

On basis of recent developments in phytoplankton ecology an assemblage index, Q, was developed to assess ecological status of different lake types established by the Water Framework Directive (WFD). Since $5 \geq Q \geq 0$, the developed index can provide 5-grade qualification required by WFD. Case studies from very different lake types support the usefulness of the developed index. Straights and weaknesses of the Q index for monitoring purposes are discussed. Without arguing for the superiority of the assemblage index in comparison with any other measures of ecological status of lakes, we aim to open a discussion about its possible applications.

Introductory remarks

The use of phytoplankton species or higher taxa for water quality assessment has a long history. Saprobic and trophic indicator species are listed in many works (Thunmark, 1945; Nygaard, 1949; Järnefelt, 1952; Teiling, 1955; Heinonen, 1980; Rosén, 1981; Kümmerlin, 1990; Gulyás, 1998; Lepistö & Rosenström, 1998; Lepistö, 1999). Numerous indices were developed (Thunmark, 1945; Nygaard, 1949; Hörnström, 1981; Brettum, 1989; Tremel, 1996; Schönfelder, 1997); however, none became widely accepted. This originates from many reasons but most importantly from disregarding (i) the assemblage concept; (ii) the dynamic feature of phytoplankton assemblages; (iii) the habitat diversity of freshwater ecosystems and (iv) the phycogeographical differences.

While community approach and succession theories, most notably the 'superorganism' approach of Clements (1916) and the individualistic concept of Gleason (1927), for terrestrial habitats date back to the beginning of the 20th century, similar theories were lacking for phytoplankton for a long time. The early attempts (for example, Pankin, 1941, 1945) to classify algal associations have never received wide acceptance (see the review by Symoens et al., 1988). Nevertheless, there has been no doubt about the fact that phytoplankton associations exist even if the phytosociological terminology (usage of the '*-etum*' suffix after characteristic genera; see Transley, 1935; Tüxen, 1955; Braun-Blanquet, 1964) has not been applied. Sommer (1986) found a high level of similarity in species composition and seasonal sequences in alpine lakes. Reynolds (1980, 1997) applied the classical phytosociological approach to a series of phytoplankton data from NW England lakes and differentiated numerous reproducible species associations. Recently, Reynolds et al. (2002) published an extended description of phytoplankton assemblages that can be understood as functional groups: groups of species with more or less precisely defined demands for several different combinations of physical, chemical and biological properties (depth of mixing layer, light, temperature, P, N, Si, CO_2 and grazing pressure) of the lake environment. At present, 33 functional groups are described (Reynolds et al., 2002; Padisák et al., 2003b), each having alphanumerical symbols (codons).

Phytoplankton communities undergo significant changes during individual years. In the process, called "seasonal succession" competitive arena is changing (Sommer et al., 1986) and dozens of generations of individual species are involved. The process is directional (Sommer, 1991), and competitive exclusion results in selection of assemblages dominated by only one (or few) species if repeated disturbances do not reverse the process. These disturbances allow maintaining high compositional diversity (Padisák et al., 1993; Scheffer et al., 2003).

Lake morphologies together with recurrent seasonal cycles of combination of main environmental variables provide repetitive competitive arenas that allow the "best adapted" species dominate repetitively in certain periods of the seasonal succession. This way, the assemblage concept of phytoplankton is closely related to development steady-state assemblages. Detailed analyses (Naselli-Flores et al., 2003a) have shown that steady-state phytoplankton assemblages develop rarely, however, if develop, they consist of strongly K-selected species that correspond to certain functional assemblages (Naselli-Flores et al., 2003b).

The Water Framework Directive (hereafter WFD; EC Parliament and Council, 2000) was proposed by the European Community in 1997 (COM(97)49) and was established in 2000 (2000/

60/EC, OJ L327 22.12.2000). The WFD is a broad concept and aims to develop sustainable management strategies for ground and surface waters in Europe. From the point of view of this paper three features of the WFD have to be stressed:

(i) it defines ecoregions (that can be divided to subregions). This way, it allows us to consider phycogeographic regions;
(ii) surface waters (lakes and rivers) are grouped based on their typological characters and number of types may vary in different ecoregions; moreover
(iii) phytoplankton is one of the five groups suggested for ecological status assessment of surface waters.

It is easy to recognize that recent developments in phytoplankton ecology and demands of the WFD overlap since the latter requires functional grouping of biota that correspond to requirements of ecological status assessment.

In each member and associate state of the EU numerous attempts have been made to find phytoplankton attributes and/or indicator groups that fulfill demands of the WFD. Numerous projects and workshops have been organized at national and international consortium levels, however, accessibility of written documents is limited since almost each represent “grey literature”. The systematic overview of this rather large body of existing knowledge is beyond the scope of this paper.

In the following sections we describe the monitoring and evaluation methods of phytoplankton that is used in Hungary. The method has a sufficiently solid theoretical basis (see the short summary above and the papers cited there), and therefore, it can be applied for ecological status assessment without geographic limitations.

Data collecting, assemblage index and lake typology

The phytoplankton assemblage index (Q) developed for WFD ecological status assessment includes the relative share (p_i, where $p_i = n_i/N$; n_i biomass of the i-th functional group; N: total biomass) of functional groups in total biomass and a factor number (F) established for the i-th functional group in the given lake type:

$$Q = \sum_{i=1}^{n} p_i F$$

Evaluation of the Q index is possible after the following steps:

(1) *Sample collection:* Integrated sample from euphotic zone of deep lake, tube sampler integrating for the whole depth in shallow lakes (tubes can be handled in cases when water depth < 3.5 m).
(2) *Sampling frequency:* Circulating information about phytoplankton sampling frequency for the WFD suggests 8 samples per year. This number “8” is inappropriate to handle temporal changes of phytoplankton since it can be shown that minimum requirement is the two-fold of the doubling time, which, in this case, would need a minimum of weekly sampling, that comes to 52 samples per year (Honti et al., in press). Such a frequency is unrealistic, although scientifically supported, on basis of cost-benefit considerations of any monitoring system. Because we are looking for functional assemblages that represent, with at least some probability, competitively selected equilibrium phases we may reduce the number of samplings per year save that we manage to find the periods when such equilibria are the most likely to develop. This approach must be optimized to the monitoring value of these stages in a seasonal sequence and their independence of real timing. In most lakes (warm mono-, di- and polymictic, latter either continuous or discontinuous), phytoplankton reaches at least four well distinguishable phases in any individual year: (i) winter time (with ice or without); (ii) spring diatom phase; (iii) clear water phase and (iv) late summer phase.

(i) Winter phase is usually long lasting (depends on latitude), and therefore easy to monitor (practically at any time between November and February in the northern temperate region). However, phytoplankton is dominated by small flagellates belonging mostly to groups **X1** and **X2**, independently from lake type and therefore monitoring value is questionable.

Table 1. Typology of Hungarian lakes

Type no	Hydro-geographical features[a]	Water depth (m)	Surface area (km^2)	Persistency of water[b]	Type, English and Hungarian name, example
1	calcareous	3–15	>100	persistent	Large lake/nagy tó (Balaton)
2	alkaline	1–3	>100	persistent	Large lake/nagy-tó (Fertő)
3	alkaline	1–3	10–100	persistent	Large lake/nagy tó (Velencei-tó)
4	calcareous-alkaline	<1	10–100	persistent	Alkaline lake/szikes tavak
5	calcareous-alkaline	<1	0.5–10	persistent	Alkaline lake/szikes tavak
6	calcareous-alkaline	<1	>0.5	temporary	Alkaline lake/szikes tavak
7	calcareous	<4	>0.5	persistent	Oxbow outside flood control dams/mentett oldali holtágak
8	calcareous-alkaline	<4	>0.5	persistent	Oxbow outside flood control dams/mentett oldali holtágak

Notes: [a]Calcareous: lakes on calcareous bedrock (even when siliceous covering layers are present); Alkaline: with $NaHCO_3$ ion-dominance and high conductivity.[b]Coverage with water can be considered as permanent if the covered surface is >0.5 km^2 in long-term average. Temporary waters dry out periodically but if covered by water their surface area >0.5 km^2.

(ii) Spring diatom development and composition reflect many characteristic feature of different lakes (=typology-conform), among them trophic state, however, its timing is strongly dependent on presence/absence and extension of winter ice cover [not talking about cases when under-ice development of diatoms occur, see examples of *Stephanocostis chantaicus* (Scheffler & Padisák 2000) and *Aulacoseira baicalensis* (Kozhov, 1963)]. Without questioning the importance of this phase, it cannot be suggested for assessments because of its unexpectedness and relative shortness (when to sample?)

(iii) The appearance of clear water phase depends on collapse of the spring diatom bloom (therefore, in most cases, on ice-break), moreover, it is rather uniform in different lake types (with assemblages of **X1**, **X2, Y, E** sometimes **U)**. Additionally, clear water phase in polymictic lakes is often difficult to recognize. In this case logistic problems, uncertainty of occurrence and assessment reasons support rejection as characteristic phase for WFD.

(iv) Stable summer conditions offer sufficient time for development of steady-state communities and timing is expectable (based on observed accelerated deepening of the thermocline or simply from trend-like drop of air temperatures). Moreover, late summer assemblages integrate the preceding successional events (including onset of stratification and disturbances) and strongly determined by the competitive arena that any given lake type offers. For this reason, only one sampling per year is recommended (appropriate also for member states where the number of lakes/expert person ratio is very high, for example, Finland) but this only sample should be taken in the late summer – early autumn period (for Hungary: 15 August–15 September).

(3) *Sample handling and preservation* (Lugol's Iodine) should follow international recommendations.

(4) *Pytoplantkon counting* should be performed with the well established, inverted microscope technique (Utermöhl, 1958) without any preconcentration method. Additional picophytoplankton counting for much type of lakes is impossible to avoid and this should follow the widely accepted standards (Padisák et al., 1999). There has been an ongoing project within the EC to standardize methods and its future recommendations may also serve as directive. Specific to the above minimized sample number/per year is the need of counting at least 1000 settling unit in each sample to

Table 2. Factor numbers of the Q index for the 8 types (see Table 1) of Hungarian lakes

Codon	TYPE							
	1	2	3	4	5	6	7	8
A	5	5	5	5	5	5	5	5
B	3	2	2	1	1	1	5	5
C	5	4	4	3	2	1	5	5
D	2	4	4	4	2	2	3	3
N	5	2	2	2	2	2	5	5
P	5	5	5	5	5	5	5	5
T	5	5	5	5	5	4	5	5
S1	0	0	0	0	0	0	0	0
S2	2	3	3	3	3	4	2	2
$\mathbf{S_N}$	0	0	0	0	0	0	0	0
Z	5	–	–	–	–	–	5	5
X3	4	5	4	4	4	4	4	4
X2	3.5	3.5	4	3	3	3	3.5	3.5
X1	4	5	3	3	3	3	3	3
$\mathbf{X_{Ph}}$	3.5	3	3	2.5	2	2	3.5	3.5
Y	2	3	3	3	3	3	3.5	3.5
E	2	2	2	2	2	2	5	5
F	5	5	3	5	4	3	3	3
G	1	0	0	0	0	0	4	4
J	1	2	2	2	1	0	3	5
K	2	4	5	5	5	5	2	2
H1	1	1	1	1	1	1	1	1
H2	3	3	3	3	3	3	3	3
U	0	0	0	0	0	0	5	5
$\mathbf{L_O}$	5	3	3	1	1	1	5	5
$\mathbf{L_M}$	0	0	0	0	0	0	0	0
M	0	0	0	0	0	0	0	0
R	–	–	–	–	–	–	–	–
V	0	0	0	0	0	0	0	0
W1	0	3	3	5	5	5	2	2
W2	0	3	3	5	5	5	3	3
$\mathbf{W_S}$	0	0	0	0	0	0	4	4
MP	5	5	5	5	5	5	3	3
Q	0	0	0	0	0	0	4	4

Codons (first row) of algal assemblages are described and analyzed in Reynolds et al. (2002) and Padisák et al. (2003), assemblage **MP** is defined in the text.

cover as much of biodiversity (flora) as possible. Taxonomic resolution of sampling should be as detailed (mostly species level) as application of light microscopy allows. Widely used, updated identification manuals should be used.

(5) Since the assemblage approach is based on resource partitioning principle, estimation of phytoplankton biomass based on density data (point 4) is impossible to avoid. Biomass estimation is based on multiplication of density data with volume (closest geometric form) supposing specific gravity of 1.00 g cm^{-3}. Sizes of different species have to be established in each sample; application of pre-determined volumes should be avoided. In case of handling historical density data, volumes in Padisák & Adrian (1999) can be used.

(6) Species with their p_i data have to be labeled with codons according to existing literature (Reynolds et al., 2002; Padisák et al., 2003b) and summed to functional groups.

(7) Factor numbers (F) have to be pre-determined according to the existing typology and knowledge. This is the most critical part of the assessment which, at present, cannot be described in general terms. See case studies for understanding.

(8) The index number (Q) that results from the equation given above ranges between 0 and 5, of which, according to the WFD's five grade evaluation system can be understood at 0–1: bad; 1–2: tolerable; 2–3: medium; 3–4: good and 4–5: excellent. Because of late summer sampling, the system reflects, at all probability, the possible worst status of most lakes. Therefore, even for reference lakes occasional "excursions" from excellent to good, or even medium, status is allowed.

Case studies

Used typology

In terms of the WFD, Hungary belongs to one single ecoregion, and according to the status 2004 October, lakes (surface area > 50 ha) are sorted into 8 types (Table 1). The factor numbers for different types were established on basis of existing data and expert knowledge (Table 2). Differentiation between some types (4–6) is dubious and existing data allow detailed testing the *Q* index to analyze only for types 1, 2 and 7.

In the inorganically turbid Hungarian shallow lakes a special kind of plankton, meroplankton, exists and sometimes dominate. Species that belong to this group are almost exclusively large diatoms (for example, certain species of *Surirella, Cymatopleura, Aulacoseira* and *Fragilaria*) that are kept in suspension by wind induced turbulence (Padisák & Dokulil, 1994). This kind of plankton was not considered by Reynolds et al. (2002) and therefore, a new group with codon **MP** was introduced to fill this gap. Periphytic diatoms that occasionally occur in lake plankton were also grouped to **MP**.

Lake Balaton

Lake Balaton is the largest (surface area: 593 km^2) shallow (average depth: 3.2 m) lake in Europe. Phytoplankton data are available since the early 20th century. Bibliography of phycological literature (Padisák & Szabó, 1997) and history of eutrophication and restoration (Padisák & Reynolds, 1998; Padisák et al., in press; Istvánovics et al., in press) were published. Since the assemblage based *Q* index is new in ecological status assessment and analyses need comparisons, we compared the index values to the algal biomass based evaluation categories published by Mischke et al. (2002). Balaton corresponds to Type 14 in the system described by Mischke et al. (2002) and boundary data for phytoplankton fresh weight (mg l^{-1}) are: ≤ 1: excellent; >1–4: good; >4–8: medium; >8–16: tolerable; >16 bad. For this analysis data from 2 years, 1994 and 2003 were used. The data were recorded in the western basin of the lake that receives bulk of the external load. Extended algal blooms (*Cylindrospermopsis raciborskii*) have been observed last time in 1994 and, due to management measures going on since the early 1980s, eutrophication was reversed and by the 2000s algal biomass remained largely at mesotrophic levels.

In the hypertrophic period (here 1994) *Q* indicated tolerable/bad status over the vegetation period; in the cold season moderate to good status was established (Fig. 1a). In terms of biomass tolerable/bad quality prevailed in July–October, otherwise excellent to medium status was recorded (Fig. 1c). In 2003, *Q* indicated good or medium quality all over the year except a short period in August–September when medium/tolerable status established (Fig. 1b). In terms of biomass water quality was excellent or good all over the year with only one record slightly in medium category (Fig. 1d). Obviously, *Q* indicates an ecological status one category lower than the classical biomass-based qualification. Detailed analyses have shown that, despite algal biomass in terms of chlorophyll-*a* has been remained low since 1995, the lake still has not been recovered fully from serious eutrophication. Summer storms may mobilize sedimentary phosphorus that results in almost immediate local algal blooms (Istvánovics et al., 2004). Parallel with the eutrophication of the lake, some alien species (mostly Nostocales and Oscillatoriales) appeared in the flora, among them *Cylindrospermopsis raciborskii*. These species reached an annual average contribution to the total biomass of 36% in 1994 with summer records constantly over 70% (Fig. 1e). These species have been still present in the flora and their share can be quite high ($>50\%$) in August–September, although annual average decreased to about 12% in 2003 (Fig. 1f). This observation together with other floristic records (for example, species like *Attheya zachariasii, Urosolenia eriensis, Diplopsalis acuta* can hardly be recorded or if so, only at level of presence) allows to conclude that ecological status estimation of the *Q* index is more realistic than that of quantitative mass variables (biomass or chlorophyll-*a*). Cross-correlations between phytoplankton biomass, %contribution of alien species to total biomass and *Q* are highly ($p < 0.1\%$) significant for 1994 and 2003, respectively (Table 3).

As a consequence of ontogenetic reasons and catchment morphology, a trophic gradient characterizes Lake Balaton with tendency for eutrophic conditions in the western basin. The gradient can be traced both by Q and algal biomass and with one category discrepancy between their quality estimation (Fig. 2).

Lake Fertő (Neusiedlersee)

Lake Fertő (Neusiedlersee) is a large (surface area is 300 km^2) shallow (mean depth of 1.3 m) lake on the Austrian–Hungarian border. The mesotrophic lake has a high salt-content, is alkaline and is very turbid. Conductivity ranges in 2000–3500 μS cm^{-1}, pH is 7.5–10. Secchi transparency in the open water is characteristically about 0.2 m (range: 0.06–0.8 m; higher

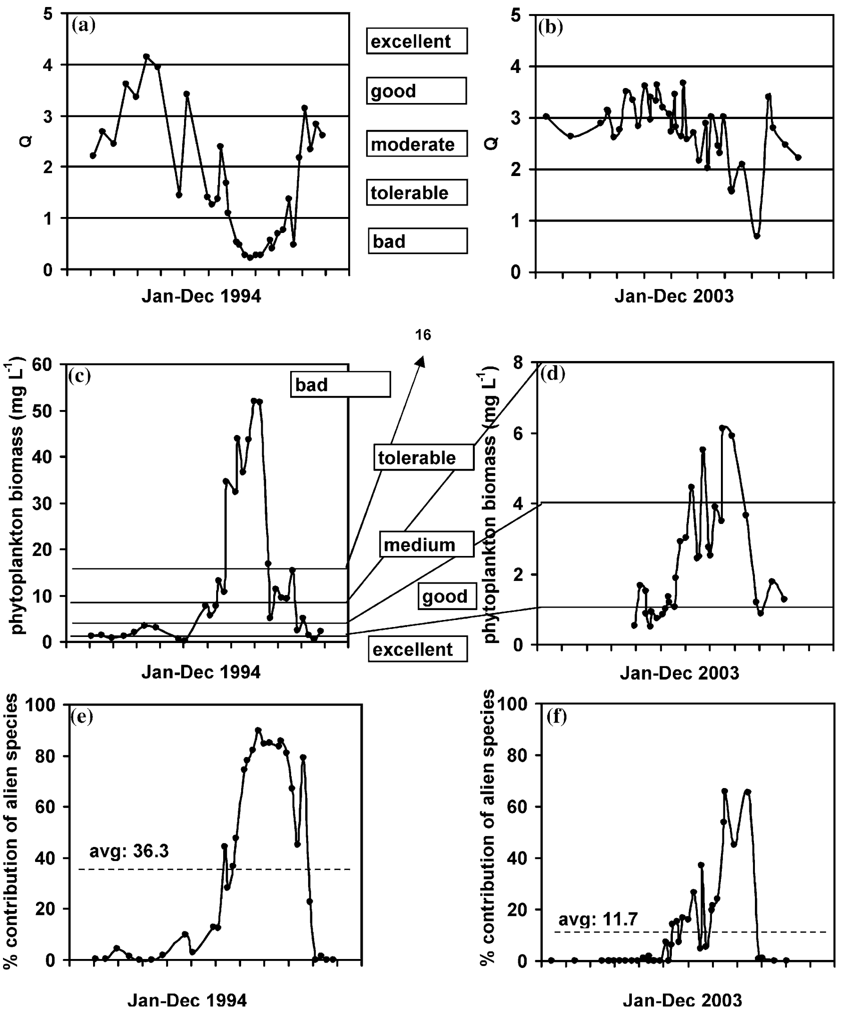

Figure 1. Ecological status assessment by the Q index (a: 1994 and b: 2003) and phytoplankton biomass (c: 1994 and d: 2003), moreover the percentage contribution of alien species (appeared parallel with the eutrophication in the flora; e: 1994 and f: 2003) in the western basin of Lake Balaton.

values occur only under ice). The lake sediment is characterized by small, slowly settling, and fine grained inorganic particles. The average seston content of the lake is very high (80–100 mg l^{-1}; comparably high values are found only in flooding period of large lowland rivers). The lake is located in the Danube valley between the Alps and the Carpathian mountains and its longitudinal axis parallels the main wind direction. The number of windless days is <1 in long term average. Therefore, the fine-grained sediments of the lake are almost continuously stirred up.

The water is rich in silica, and the concentrations of dissolved N and P (average for 1987–1992: 297 μg N l^{-1} and 15 μg P l^{-1}) are usually higher than the level that is considered limiting in other lakes. Because of the inherently high turbidity of the open water of the lake, growth of phytoplankton is presumably subject to frequent or continuous light limitation, with only brief exceptions (Padisák, 1998).

These conditions set a manifold stress on phytoplankton and its main features are the following:

(i) The salinity that is relatively high for freshwaters delimit number of species in the phytoplankton flora and the number of floral

Table 3. Correlation matrix of *Q* index, phytoplankton biomass and % of alien species

	Q	biomass	% alien
Q	x	**−0.57**	**−0.81**
Biomass	*−0.69*	x	**+0.76**
% alien	*−0.89*	*+0.76*	x

Data printed in italics refer to year 1994 and bold printing indicates data for 2003. For each correlation $p < 0.1\%$

elements are further reduced for shade tolerant species. Dominant species are

- colonial blue-greens with picoalgal cell sizes (*Aphanocapsa, Aphanothece*; codon **X3**);
- meroplanktonic species (for example, *Surirella peisonis, Campylodiscus clypeus* and long chains of *Fragilaria construens*; codon **MP**) that are kept in suspension by frequent resuspension but from time to time sink fast to the bottom. According to estimates, one specimen spends half of its life-time in the euphotic region that allows photosynthesis (see detailed description in Padisák & Dokulil, 1994);
- green algal species with thick mucilageous envelop that provides good floating abilities (*Oocystis* spp; *Planktosphaeria gelatinosa, Coenochlorys* sp., *Lobocystis planktonica*; codon **F**) and
- chloroccalean species with elongated cells that are effective ''light antennae'' (c. f. Reynolds, 1988) like *Monoraphidium* and *Koliella* spp. (codons **X1, X3**).

At annual average, 2/3 of the biomass is given by **MP** assemblage.

(ii) In long-term average, horizontal differences along the longitudinal axis cannot be observed (there is no trophic gradient).

(iii) Despite the contribution of phytoplankton biomass to total seston is very low (0.54%; Padisák & Dokulil, 1994), the correlation between these two variables, unlike rivers, is a significant and positive one because both are driven by periodic resuspensions.

(iv) Short-term (weekly) changes of the phytoplankton appear rather chaotic (Fig. 3), since frequent phytoplankton peaks result from periodic resuspensions and not from sudden growth. However, growth periods can be traced by application of statistical methods (Padisák & Dokulil, 1994).

(v) Annual average phytoplankton biomass (Fig. 3.) reflects quite stable carrying capacity.

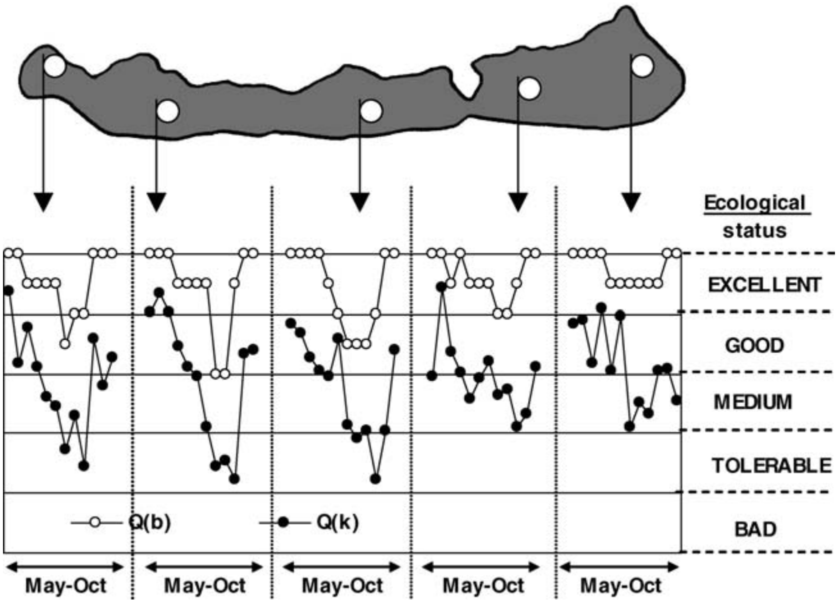

Figure 2. Ecological status estimation by the index based on phytoplankton assemblages [Q(k) = Q on other figures and in the text] and phytoplankton biomass [Q(b)] in five sampling stations along the longitudinal axis of Lake Balaton in May–October, 2003.

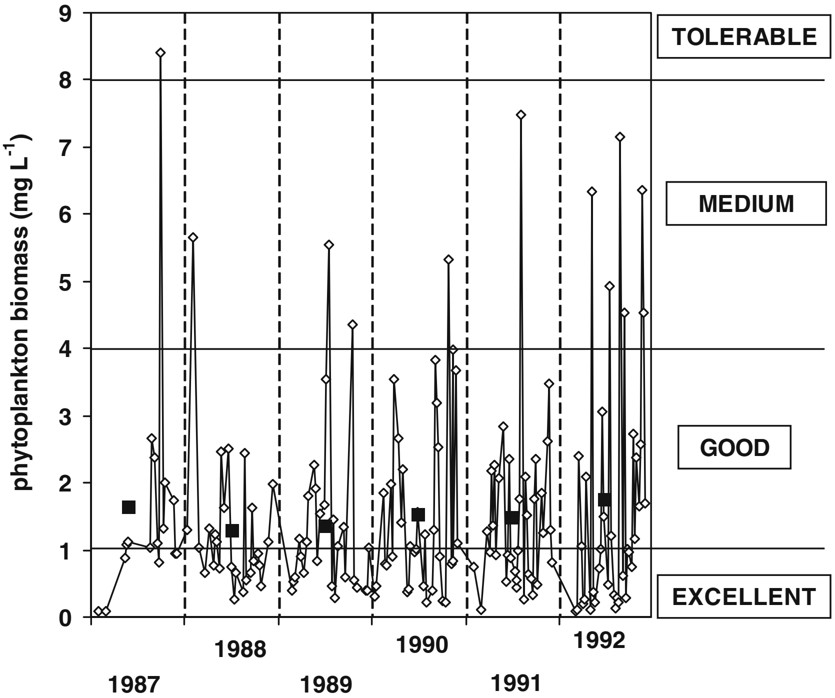

Figure 3. Changes (sampling: weekly to monthly) in phytoplankton biomass (open circles) in Lake Fertő between 1987 and 1992). Closed squares indicate annual averages. Excellent/good/medium tolerable qualification was established according to Type 14 in Mischke et al. (2002).

On level of ecological status assessment by traditional methods (phytoplankton biomass, chlorophyll-*a*), this rapidly changing nature of Lake Fertő's phytoplankton results in rapidly changing qualifications (Fig. 3). Since ecological conditions cannot be qualified as basically different in yes/no wind periods, traditional methods are apparently inadequate for status assessment. The qualification on basis of the *Q* index (Fig. 4) reflects the long-term stability of ecological conditions since the phytoplankton assemblage based index results in permanently excellent qualification (Fig. 4a) and within year changes in ice-free periods are minor (mostly excellent, sometimes good status (Fig. 4b). As a consequence of increased growth of flagellates (mostly *Cryptomonas, Rhodomonas, Gymnodinium*) in the period when the lake is covered by ice results in good/moderate qualifications this way indicating a poorer quality than really exists (since **MP** species are settled on the surface for a long period, however, they are still there).

It is important to note here that Lake Fertő/Neusiedlersee crosses the border between Austria and Hungary, therefore the responsible Austrian authorities also qualified the ecological status of the lake. Their analysis agrees in establishing the present status of the lake as "excellent" (F. Szilágyi, personal communication), and therefore the assessment by phytoplankton assemblage index survives intercalibration test.

Alkaline lakes (types 4, 5, 6, data of type 8 [Szelidi-tó] was included since this type can be considered as degraded version of the types 4–6)

Alkaline lakes on the Hungarian plain are typically very shallow (water level fluctuates considerably within the year, many dries out at an annual basis), their transparency is very low (aphotic and euphotic zones differentiate at water levels of only 30 cm), conductivity may increase up to 30,000 μS cm^{-1}, inorganic N and P concentrations are far above the level that can be limiting for phytoplankton growth. Therefore, double stress (low light and high salinity) explained in the case of Lake Fertő applies for these lakes at a more extreme level. Salinity and diversity correlates

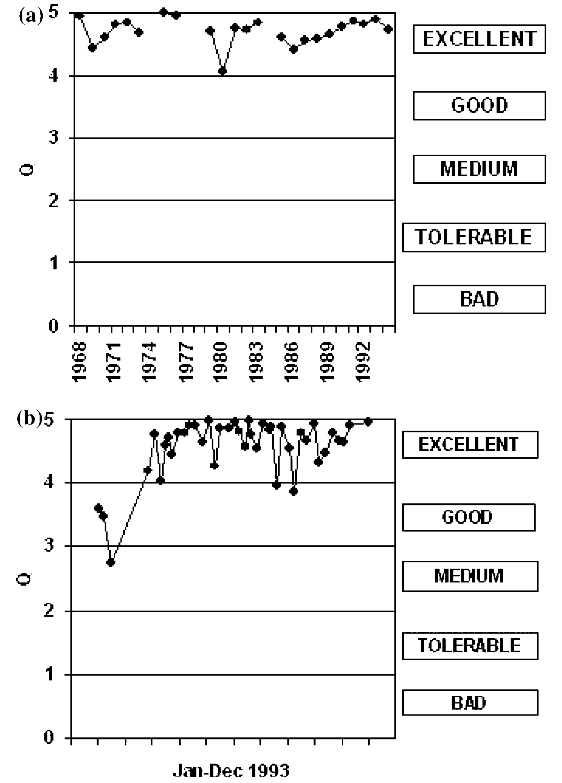

Figure 4. Ecological status assessment by the Q index on interannual (a) and annual (b) scales in Lake Fertő.

negatively (Fig. 5a) as commonly observed for different kinds of biota (see for example Kalff, 2002).

The most important threat for these lakes is stabilization of water level by balancing evaporation with water supply from deep-layer aquifers (because, for example, tourism needs more stable water balance than natural). This might lead to a shift towards freshwater conditions and meantime specialized flora and fauna is lost. On the basis of a very limited data set (Padisák et al., 2003b), however, the phytoplankton assemblage index seems to be appropriate to reflect degraded state (conductivity lower than 4–5000 μS cm^{-1}) of these ecosystems (Fig. 5b).

Oxbows (type 7)

Despite number of large oxbow lakes in Hungary is large, quantitative data on phytoplankton, especially from systematic samplings, are very rare.

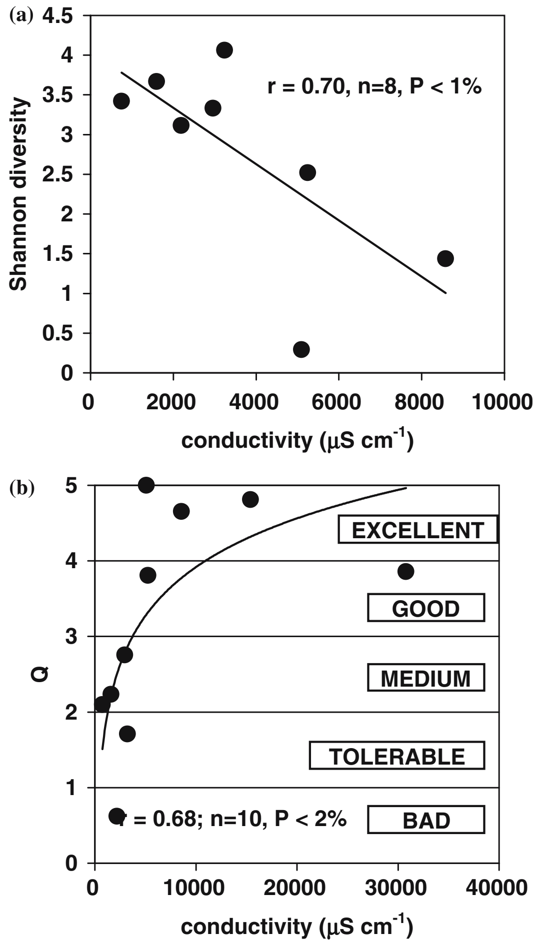

Figure 5. Relationship between conductivity/Shannon-diversity (a) and conductivity/Q index (b) in some alkaline lakes in Hungary.

The oxbow Boroszlói-holtág of river Tisza is considered to be in pristine stage and along its longitudinal axis "successional age" is changing: being one of the edge (sampling site 1) the oldest. Dense submerged macrophyte cover opens in sequence of sampling stations; form 4th point on open water is dominant. Qualification by traditional quantitative methods provides good or excellent ecological states except successionally old areas (sampling sites 1–3) in May (Fig. 6a). In other words, this approach "punishes" old successional phases despite they obviously belong to natural ontogenesis of such ecosystems. The assemblage-based qualification (Fig. 6b) results in an overall good/excellent qualification that is in

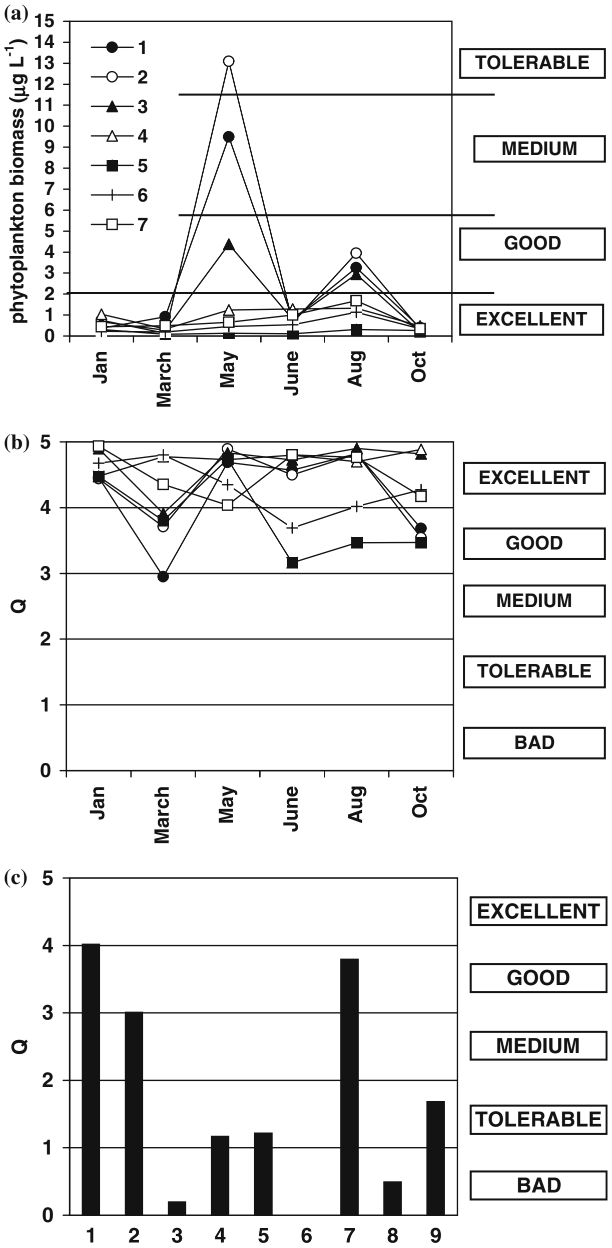

Figure 6. (a) Annual changes of phytoplankton biomass in the oxbow Boroszlói-holtág of the River Tisza. Status assessment follows Mischke et al.'s (2002) recommendation for lake type 11 in the cited paper. (b) Ecological status assessment by the Q index on for the oxbow Boroszlói-holtág of the River Tisza. (c) Ecological status assessment by the Q index based on late summer algal assemblages in 9 different Hungarian oxbow lakes. 1 – Boroszlói-holtág; 2 – Peresi-Holt-Körös; 3 – Endrőd-Középső-Holt-Körös; 4 – Rakamazi-Holt-Tisza; 5 – Óhalászi-Holt-Tisza; 6 – Siratói-Holt-Körös – for this oxbow data exist but the value of the Q index = 0; 7: Kecsegés-zugi-Holt-Körös; 8 – Tiszadobi-Holt-Tisza; 9 – Klágya-Duna.

agreement with the view of experts in nature conservation (pristine).

In cross-oxbow comparison (Fig. 6c) the Boroszlói-holtág is the only one that reaches the "excellent" level (each data from late summer), all the others are characterized by different level of degradation. This comparison shows that the *Q*-index is able to "tolerate" different natural successional stages but sufficiently sensitive to human impacts. Latter are various: lack of floodplain forests (therefore the oxbow in more wind-exposed than it ought to be), too intensive recreational use, household or agricultural sewage ingress, etc.

Discussion

As demonstrated on basis of case studies from many different lakes (with one exception each Hungarian WFD lake types were represented) the assemblage index (*Q*) proves to be a promising tool to assess of ecological status of lakes using phytoplankton as indicator group.

Straights of the method can be summarized as:

(i) The ecological basis of the assemblage index is a robust one that is suitable to follow developments in basic science. This provides the necessary flexibility of applied monitoring use to follow progress in academic science.

(ii) The base of qualification can be adjusted to any given ecoregion that allows flexibility declared in the WFD without basic conceptual changes. This feature of the assemblage index allows incorporation of biogeographical considerations.

(iii) Use of assemblage index is not restricted to ecoregions of Europe. Most assemblages can be found along wide latitudes and, even at original form (Reynolds et al., 2002), at least one basically tropical assemblage is included (**S2** with dominance of *Spirulina*, *Arthrospira*, *Rhaphidiopsis*). Since there is a growing interest driven by the need for appropriate indices for environmental status assessment all over the world, this feature of the assemblage index makes it potentially suitable to use globally.

(iv) Unlike any other previously developed indices, the assemblage does not give preference to any particular human impact (changes in trophic state, acidification, saprobic state, desalination/salination, etc.) that tremendously increases fields of application.

(v) Since assemblages are defined on basis of their preferences to certain combination of habitat properties, the *Q* index might be suitable to test habitat-based monitoring systems (for example the habitat suitability concept that has been under development in the US; R. T. James, personal communication).

Weaknesses of the method:

(i) Officers of monitoring agencies commonly greet the method since they percept the significant reduction of categories (only some 30 instead of hundreds of species) with the hope that taxonomic knowledge of the monitoring staff can be reduced. This is a basic misunderstanding since the method needs identification and biomass estimation at species level and sorting the species to different assemblages is only a second step.

(ii) The method is differently sensitive for misidentifications. Most (but not each) genera of chlorococcalean green algae are grouped into codon **J**, while dominant filamentous cyanoprokaryotes are sorted into different assemblages. As manuals for Cyanoprokaryota have been still in "in preparation – submitted – in press" status (except Chroococcales; Komárek & Anagnostidis, 1999) identification of this group supposes detailed knowledge in taxonomic literature.

(iii) Identification manuals for some groups exist but habitat preferences of the species are poorly known. Best examples for this case are dinoflagellates with accepted identification system (Popovsky & Pfiester, 1990) and poor knowledge in the autecology of species (Grigorszky et al., 2003).

(iv) Probably the most important deficiency of the assemblage index is that factor numbers (see Table 2), at least at present, are established exclusively on basis of expert knowledge. More knowledge in autecology of species and more statistical analyses on existing data bases are necessary to establish a more objective qualification system; however, the developed system is ready to accept changes if academic knowledge increases. Reynolds' (2000) "phytoplankton designer" game supports sufficiently the adaptability of expert knowledge in such cases.

(v) Sorting of different species into a particular functional assemblage is not automatic. The most striking example is probably

Cylindrospermopsis raciborskii that, in most cases belongs to codon $\mathbf{S_N}$. However, if the species occurs in deep layer chlorophyll maxima (Padisák et al., 2003a), labeling as **R**-species is correct. This means that more than one kind of habitat types can suffice requirement of a given species. At level of basic science this appears as minor problem but certainly does not make easy compiling technical guidelines for monitoring staff.

In this paper we do not argue for the superiority of the assemblage index as monitoring tool of ecological status assessment within the WFD. However, the developed index has some remarkable features that meets the philosophy of requirements of habitat-based qualifications, and, here we aim to open a discussion about its possible applications.

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References

Braun-Blanquet, J., 1964. Pflanzensoziologie. Springer, Wien 865 pp.

Brettum, P., 1989. Alger som indikatorer pa vannkvalitet i norske innsjoer. Planteplankton. NIVA, Blindern, Oslo.

Clements, F. E., 1916. Plant Succession: An Analysis of the Development of Vegetation. Carnegie Institute, Washington.

EC Parliament and Coincil, 2000. Directive of the European Parliament and of the Council 2000/60/EC Establishing a Framework for Community Action in the Field of Water Policy. European Commission, Luxembourg. PE-CONS 3639/1/100 Rev 1.

Gleason, H., 1927. Further views on the succession-concept. Ecology 8: 299–326.

Grigorszky, I., G. Borics, J. Padisák, B. Tótmérész, G. Vasas, S. Nagy & G. Borbély, 2003. Factors controlling the occurrence of Dinophyta species in Hungary. Hydrobiologia 506–507: 203–207.

Gulyás P., 1998. Szaprobiológiai indikátorfajok jegyzéke. [List of saprobic indicator species] Vízi Természet- és Környezetvédelem, Környezetgazdálkodási Intézet, Budapest. [in Hungarian].

Heinonen P., 1980. Quantity and composition of phytoplankton in Finish inland waters. Publication of the Water Research Institute.

Honti, M., V. Istvánovics & A. Osztoics, in press. Stability and change of phytoplankton communities in a highly dynamic environment – the case of large, shallow Lake Balaton (Hungary). Hydrobiologia.

Hörnström, E., 1981. Trophic characterisation of lakes by means of quantitative phytoplankton analysis. Limnologica 13: 249–361.

Istvánovics V., A. Clement, L. Somlyódy, A. Specziár, L. G.-Tóth & J. Padisák, in press. Updating water quality targets for shallow Lake Balaton (Hungary), recovering from eutrophication. Hydrobiologia.

Istvánovics, V., A. Osztoics & M. Honti, 2004. Dynamics and ecological significance of daily internal load of phosphorus in shallow Lake Balaton, Hungary. Freshwater Biology 49: 232–252.

Järnefelt, H., 1952. Plankton als Indikator der Trophiegruppen der Seen. – Annales Academia Scientarium Fennica A IV. Biology 18: 1–29.

Kalff, J., 2002. Limnology. Prentica Hall, Upper Sadle River, New Jersey.

Komárek, J. & K. Anagnostidis, 1999. Cyanoprokaryota. Chroococcales. Süβwasserflora von Mitteleuropa 19/1. Gustav Fischer, Jena.

Kozhov, M., 1963. Lake Baikal and its Life. Dr. W. Junk Press, The Hague.

Kümmerlin, R., 1990. Plankton-Gemenischaften als Bioindikatoren für Stehgewässer. Ökologie und Naturschutz 3: 227–241.

Lepistö L., 1999. Phytoplankton assemblages reflecting the ecological status of lakes in Finland. Monographs of the Boreal Environment Research 16, Tampere.

Lepistö, L. & U. Rosenström, 1998. The most typical phytoplankton taxa in four lakes in Finland. Hydrobiologia 369/370: 89–97.

Mischke, U., B. Nixdorf, E. Hoehn & U. Riedmüller, 2002. Möglichkeiten zur Bewertung von Seen anhand des Phytoplanktons. Aktueller Stend in Deutschland. Aktuelle Reihe 5/02: 25–37. Brandenburgische Technische Universität, Cottbus.

Naselli Flores L., J. Padisák, M. T. Dokulil, (eds), 2003a. Phytoplankton and Equilibrium Concept: The Ecology of Steady-State Assemblages. Hydrobiologia 502, Developments in Hydrobiology 172.

Naselli-Flores, L., J. Padisák, M. T. Dokulil & I. Chorus, 2003b. Equilibrium/steady-state concept in phytoplankton ecology. Hydrobiologia 502: 395–403.

Nygaard, G., 1949. Hydrobiological studies on some Danish ponds and lakes. Pert II: The quotient hypothesis and some little known plankton organisms. Kongelige Danske Videnskabernes Selskab Biologiske Skrifter 7: 1–293.

Padisák, J., 1998. Sudden and gradual responses of phytoplankton to global climate change: case studies from two large, shallow lakes (Balaton, Hungary and the Neusiedlersee Austria/Hungary). In George, D. G, J. G, Jones, P. Puncochar, C. S. Reynolds, & D. W. Sutcliffe (eds.) Management of Lakes and Reservoirs during Global Change. Kluwer Academic Publishers, Dordrecht, Boston. London: 111–125.

Padisák, J. & R. Adrian, 1999. Biovolumen. In Tümpling, W. & G. Friedrich (eds.) Methoden der Biologischen Wasseruntersuchung 2. Biologische Gewässeruntersuchung, Chapter 5.1. Gustav Fischer Verlag, Jena: 334–367.

Padisák, J. F., A. R. Barbosa, R. Koschel & L. Krienitz, 2003a. Deep layer cyanoprokaryota maxima are constitutional features of lakes: examples from temperate and tropical regions. Archiv für Hydrobiologie, Special Issues Advances in Limnology 58: 175–199.

Padisák, J., G. Borics, G. Fehér, I. Grigorszky, I. Oldal A. Schmidt & Z. Zámbóné-Doma, 2003b. Dominant species and frequency of equilibrium phases in late summer phytoplankton assemblages in Hungarian small shallow lakes. Hydrobiologia 502: 157–168.

Padisák, J. & M. Dokulil, 1994. Meroplankton dynamics in a saline, turbulent, turbid shallow lake (Neusiedlersee, Austria and Hungary). Hydrobiologia 289: 23–42.

Padisák, J., L. Krienitz & W. Scheffler, 1999. Chapter 3.6.1. Phytoplankton In Tümpling, W. & Friedrich, G. (eds), Methoden der Biologischen Wasseruntersuchung 2. Biologische Gewässeruntersuchung Gustav Fischer Verlag Jena: 35–52.

Padisák, J., G. Molnár, É. Soróczki-Pintér, É. Hajnal & D. G. George, (in press), Four consecutive dry years in Lake Balaton (Hungary): consequences for phytoplankton biomass and composition. Verhandlunger der Internationale Vereinigung für Limnologie Vol. 29.

Padisák, J. & C. S. Reynolds, 1998. Selection of phytoplankton associations in Lake Balaton, Hungary, in response to eutrophication and restoration measures, with special reference to cyanoprokaryotes. Hydrobiologia 384: 41–53.

Padisák, J., C. S. Reynolds & U. Sommer, 1993. Intermediate Disturbance Hypothesis in Phytoplankton Ecology. Developments in Hydrobiology 81/Hydrobiologia 249. Kluwer Acad. Publ, Dordrecht, Boston, London.

Padisák J. & I. Szabó, 1997. Botanikai kutatások: alacsony- és magasabbrendû növények [Botanical results: lower- and higher plants] In Salánki J. & J. Nemcsók (eds), A Balaton kutatás eredményei 1981–1996 [Results of scientific research in Lake Balaton between 1981 and 1997): 97–136. VEAB, Veszprém, ISBN 963 7385 452 [in Hungarian].

Pankin, W., 1941. Die Vegetation einiger Seen und Umgebung von Joachimsthal. Bibliotheca Botanica, H 119: 1–162.

Pankin, W., 1945. Zur Entwicklungsgeschichte der Algensoziologie und zum Problem der 'echten' und 'zugehörigen' Algengesellschaften. Archiv für Hydrobiologie 41: 92–111.

Popovsky, J. & L. A. Pfiester, 1990. Dinophyceae (Dinophyta). Süβwasserflora von Mitteleuropa 6. Gustav Fischer, Jena.

Reynolds, C. S., 1980. Phytoplankton assemblages and their periodicity in stratifying lake systems. Holarctic Ecology 3: 141–159.

Reynolds, C. S., 1988. Functional morphology and the adaptive strategies of freshwater phytoplankton. In Sandgren, C. D. (ed.) Growth and Reproductive Strategies of Freshwater Phytoplankton. Cambridge University Press, Cambridge: 388–433.

Reynolds, C. S., 1997. Vegetation Processes in the Pelagic. Ecology Institute, Oldendorf/Luhe, Germany.

Reynolds, C. S., 2000. Phytoplankton designer – or how to predict compositional responses to trophic-state change. Hydrobiologia 424: 123–132.

Reynolds, C. S., V. Huszar, K. Kruk, L. Naselli-Flores & S. Melo, 2002. Towards classification of the freshwater phytoplankton. Journal of Plankton Research 24: 417–428.

Rosén, G., 1981. Phytoplankton indicators and their relation to certain chemical and physical factors. Limnologica 13: 263–290.

Scheffer, M., S. Reinaldi, J. Huisman & F. J. Weissing, 2003. Why plankton communities have no equilibrium: solutions to the paradox. Hydrobiologia 491: 9–18.

Scheffler, W. & J. Padisák, 2000. *Stephanocostis chantaicus* (Bacillariophyceae): morphology and population dynamics of a rare centric diatom growing in winter under ice in the oligotrophic Lake Stechlin, Germany. Archiv für Hydrobiologie 98/Algological Studies 133: 49–69.

Schönfelder, I., 1997. Eine Phosphor-Diatomeen-relation für alkalische Seen und Flüsse Brandenburgs und ihre Anwendung für die paläolimnologische Analyse von Auensedimenten der unteren Havel. Dissertaiones Botanicae 283. Cramer J, Berlin.

Sommer, U., 1986. The periodicity of phytoplankton in Lake Constance (Bodensee) in comparison to other deep lakes of central Europe. Hydrobiologia 138: 1–7.

Sommer, U., 1991. Phytoplankton: directional succession and forced cycles. In Remmert, H. (ed.) The Mosaic-Cycle Concept of Ecosystems. Springer Verlag, Berlin: 132–146.

Sommer, U., M. Gliwicz, W. Lampert & A. Duncan, 1986. The PEG model of seasonal succession of planktonic events in fresh waters. Archiv für Hydrobiologie 106: 433–471.

Symoens, J.-J., E. Kusel-Fetzmann & J.-P. Descy, 1988. Algal communities of continental waters. In Symoens, J.-J. (ed.) Vegetation of Inland Waters. Kluwer Academic Publishers, Dordrecht: 183–221.

Teiling, E., 1955. Some mesotrophic phytoplankton indicators. Verhandlunger der Internationale Vereinigung für Limnologie 12: 212–215.

Thunmark, S., 1945. Zur Soziologie des Süsswasserplanktons. Eine methodisch-ökologische Studie. Folia Limnologica Skandinvaica 3: 1–66.

Transley, A. G., 1935. The British Isles and their Vegetation. Cambridge University Press, Cambridge.

Tremel, B., 1996. Determination of the trophic state by qualitative and quantitative phytoplankton analysis in twi gravel pit lakes. Hydrobiologia 323: 1–38.

Tüxen, R., 1955. Das Systeme der nordwestdeutschen Pflanzengesellschaft. Mitteilungen der Floristisch-soziologische Arbeitgemenschaft 5: 1–119.

Utermöhl, H., 1958. Zur Vervollkommnung der quantitativen Phytoplankton-Methodik. Mitteilungen der internationale Vereinigung der theoretische und angewandte Limnologie 5: 567–596.