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Assessment of water quality in a large lowland river (Narva, Estonia/Russia) using a new Hungarian potamoplanktic method

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Abstract. Phytoplankton is one of the four biological elements used for the assessment of the water quality of surface water bodies. In rivers phytoplankton-based assessment of water quality has not been conducted in Estonia up to now. The aim of the present study was to test a new Hungarian potamoplanktic method for the assessment of the water quality of the Narva River, a large river on the Estonian–Russian border. For testing the method, algal species in the phytoplankton of the Narva River were classified into functional groups. Then the Ecological Quality Ratio (EQR) was calculated and its value was compared with the corresponding values for different water quality classes given in the literature. The mean value of the EQR for the Narva River revealed seasonal variation: in most cases it indicated 'good' or 'excellent' quality classes in spring and summer and 'very bad' quality class in autumn. The median value for the vegetation period of 2001–2008 indicated 'moderate' quality class. Variation in the functional groups of phytoplankton and in the EQR values reflected the seasonal dynamics of phytoplankton and the impact of Lake Peipsi. Comparison of the results of the assessments made by using the phytoplankton EQR and benthic diatom indices revealed agreement between the two metrics in the summer period: both were sensitive to the water quality and indicated at least 'good' quality class.

The new Hungarian method appears to be suitable for the assessment of water quality in this Estonian large river. However, the numerical boundaries of the EQR for different water quality classes should be specified in the future on the basis of a larger Estonian phytoplankton database.

Key words: phytoplankton functional groups, Ecological Quality Ratio, rivers, water quality assessment method.

INTRODUCTION

Phytoplankton is one of the four biological elements used for the assessment of the ecological status of surface water bodies according to the Water Framework Directive, WFD (EC Parliament and Council, 2000). The WFD prescribes assessment of the ecological status of surface waters using the Ecological Quality Ratio (EQR). The EQR is defined as the relationship between the current observed value and the reference condition for a given biological element. The reference condition reflects a relatively undisturbed state with only minimal human impact.

The Directive provides descriptive definitions for five classes of the ecological status (high, good, moderate, poor, and bad), leaving the numerical boundaries between the classes to be elaborated by each EU country. Numerous attempts have been made to develop phytoplankton metrics for lakes (Kangur et al., 2003; Nixdorf et al., 2003; Padisák et al., 2006). For rivers the WFD does not consider it necessary to elaborate phytoplankton-based quality assessment methods because of the predominance of allochthonous organic matter over autochthonous primary production (Reynolds, 2000). Nevertheless, several studies (Descy et al., 1988; Kiss, 1994; Dokulil, 1996; Borics et al., 2007) indicate development of eutrophic or even hypertrophic phytoplankton assemblages in large lowland rivers, where phytoplankton provides a quantifiable measure of water quality degradation.

In Estonia the Narva is the only river belonging to the 'large lowland river' type (catchment area >10 000 km^2). Its upper course is used for the uptake of drinking water for the town of Narva (70 000 inhabitants). For a river that serves as a drinking water source intensive phytoplankton development is undesirable. The Narva is also important for energy production: a hydropower plant and two thermal power plants have been built along the river. The upper and lower reaches of the Narva River have been included in the Natura 2000 network owing to the habitats of salmon (*Salmo salar* L.), grayling (*Thymallus thymallus* L.), and brown trout (*Salmo trutta* morpha *fario* L.) there.

Previous phytoplankton research on the Narva River has been relatively occasional (Tuvikene, 2003; Nõges et al., 2005; Tuvikene et al., 2005). There has been no regular hydrobiological monitoring up to now, except in the Narva Reservoir where it has been carried out since 2001. The water quality of the Narva River has been evaluated according to hydrochemical criteria and by using diatom indices (Tuvikene et al., 2006, 2009).

Recently, a new Hungarian evaluation technique was elaborated for the assessment of the water quality of rivers (Borics et al., 2007). The method is based on phytoplankton assemblages and has been tested with hundreds of samples from Hungarian rivers.

The main aim of the present study was to test the new Hungarian potamoplanktic method for the assessment of the water quality in the Narva River. The hypothesis is that the method elaborated for the rivers of Hungarian lowlands and the Carpathian ecoregion is valid also for rivers of the Baltic Province ecoregion. Another aim was to compare our results with the results obtained by using benthic diatom indices of the Narva River.

MATERIAL AND METHOD

Study area

The Narva is the largest river in Estonia with respect to discharge; its mean annual volume is 400 $m^3 s^{-1}$ (min 70–80 $m^3 s^{-1}$, max 2000 $m^3 s^{-1}$). The river draws its water from Lake Peipsi, runs along the border of the Republic of Estonia and

the Russian Federation, and falls into Narva Bay, Gulf of Finland (Fig. 1). The area of the Narva River basin is 56 200 km^2, of which 63% belongs to Russia, 30% to Estonia, and 7% to Latvia. The length of the river is 77 km, the mean width and depth in the lower course are about 350 m and 6.0 m, respectively (Loopmann, 1979). In the upper river reach the banks are covered mostly by swamped forests and bogs. Only a few small villages are located in the area. The banks are bordered with *Phragmites australis* (Cav.) Trin. ex Steud. and *Glyceria maxima* (Hartm.) Holmb. In its lower reach the river runs through the towns of Narva/Ivangorod and Narva-Jõesuu. Since 1956 the river has been dammed by the Narva Reservoir, which extends up to 38 km upstream (total surface area 200 km^2; mean depth 1.8 m, maximum depth 15 m). In the reservoir the water turnover is 34–35 times a year (Mištšuk & Jaani, 2000). Several tributaries fall into the Narva, of them the Plyussa in the territory of Russia is the largest (catchment area 6650 km^2, mean water discharge 50 m^3 s^{-1}). Between the towns of Narva and Ivangorod the river flows over the Baltic klint, forming Narva waterfall, which was among the most powerful waterfalls in Europe before the river was dammed (Suuroja, 2005). Kreenholm Islet divides the waterfall into two sections: Kreenholm waterfall located in the western part of the islet (width 60 m and height 6.5 m, with multiple terraces) and Joala waterfall in its eastern part (width 110 m and height 6.3 m). Water is let to flow in the former original channel for a few days every year and the waterfall is dry most of time. The current velocity varies in the lower course from 0.5 to 0.6 m s^{-1} (Jaani, 2000).

Fig. 1. Location of the Narva River and of the sampling site at Narva-Jõesuu.

Fieldwork

The Narva River was sampled 15 times, one to three times in the vegetation periods of 2001, 2003–2006, and 2008 (see Table 1 for the sampling dates). As the Narva River rises from a large lake and its length is only 77 km, sampling was carried out at one site (Narva-Jõesuu, 59°28′ N; 28°02′ E) in the lower reach (Fig. 1).

Quantitative phytoplankton samples (100–200 mL) were taken from a depth of 0.1 m from the thalweg. Samples were preserved in dark glass bottles with Lugol's iodine solution (1% final concentration) according to the European standard EN 15204 (2006). Water for hydrochemical analyses was taken simultaneously with phytoplankton sampling. All samples were kept cool in the dark.

Water transparency (m) was measured by Secchi disk; water temperature (°C), pH, and oxygen concentration (mg L^{-1}) were measured in situ with a multisensor F/SET (WTW Wissenschaftlich-Technische Werkstätten GmbH, Germany). The accuracy of the estimations was in all cases 0.1 of the relevant unit.

Laboratory work

Phytoplankton samples were analysed within one month using inverted bright field microscope (Labovert Leitz, Rockleigh, N.J) according to the European standard EN 15204 (2006). Samples were left to settle in 2.5–10 mL Utermöhl (1958) chambers for 24 h (Olrik et al., 1998). To obtain a statistically acceptable estimation of the number of organisms, approximately 100 individuals from each of the most abundant species, or, in total, at least 500 organisms were counted per sample (Olrik et al., 1998). The species larger than 50 µm were counted over the whole chamber or half of it at magnification 12 × 10. The other species were counted at magnification 12 × 32. The number of counting units (cells, filaments, or colonies, 10^6 L^{-1}) was converted to biovolume (wet weight biomass, mg L^{-1}) using stereometric formulae after Olrik et al. (1998). The following literature sources were used to identify the cyanobacteria and algae: Komárek & Fott (1983), Anagnostidis & Komárek (1988), Tikkanen & Willén (1992), Uherkovich et al. (1995), and Komárek & Anagnostidis (1999, 2005).

Hydrochemical samples were analysed the next day after sampling. The content of nitrogen and phosphorus compounds was analysed from the non-filtered water samples using standard methods (Grasshoff et al., 1999) to 0.01 units. The samples were digested with persulphate to determine total nitrogen (TN, mg L^{-1}) and total phosphorus (TP, mg L^{-1}). The concentration of TN was determined by the cadmium reduction method. The highly coloured azo dye formed was measured by a spectrophotometer at 545 nm (Model 6300, Jenway, UK). The concentration of ammonium nitrogen (NH_4-N, mg L^{-1}) was determined by the indophenol blue method, and the absorbance of the solution was measured by a spectrophotometer at 630 nm. The TP concentration was determined by the ascorbic acid method, and the absorbance of the solution was measured at 880 nm.

The value of BOD_5 was obtained from the difference between the measurements of dissolved oxygen before and after the incubation period (5 days at 20 °C in the dark) and determined with an accuracy of 0.1 mgO L^{-1}.

Application of the new Hungarian potamoplanktic method

The Hungarian potamoplanktic method is based on the functional groups of phytoplankton, elaborated by Reynolds et al. (2002). The functional group of phytoplankton is a group of species with more or less specified demands for different combinations of physical, chemical, and biological conditions (for instance light, temperature, nutrient concentration, grazing pressure, etc.). The groups are mostly polyphyletic, recognizing commonly shared adaptive features (Reynolds et al., 2002). Each group is characterized by an alphanumerical code, habitat template, and representative taxa. The alphanumerical codes are allocated in blocks to reflect seasonal shifts (A–D for vernal blooms, E–H for assemblages at the start of summer stratification, etc.) and within each block, the trophic 'preference' is distinguished (for instance, C→G→M→P indicates the eutrophication process) (Reynolds et al., 2002). The habitat template shows the favourable environment for the corresponding functional group (clear, deep, poor lakes for functional group with code A; shallow turbid waters including rivers for functional group with code D, etc.) (Padisák et al., 2009). Originally, functional groups were used to characterize standing water bodies, the number of the groups was 31, and each group was characterized by one to four algal species (Reynolds et al., 2002). Now the functional classification of phytoplankton has attracted the attention of many phytoplankton ecologists worldwide and it has been updated mainly by Padisák et al. (2003, 2006, 2009), Callieri et al. (2006), Devercelli (2006), Baranazarova & Lyashenko (2007), Borics et al. (2007), Sarmento & Descy (2008), and Souza et al. (2008). The total number of groups is 40 in the last updated version and the number of the representative taxa per each functional group varies from three to more than forty (Padisák et al., 2009).

Elaboration of the new potamoplanktic method needed evaluation of phytoplankton associations in rivers (Borics et al., 2007). Therefore, evaluation of the elements of the phytoplankton assemblages was performed by estimating how far or how close these limnetic assemblages were to those types that can be considered as reference algal assemblages of the rivers (Borics et al., 2007). They focused on those environmental elements that are specific for rivers (short residence time), or are important from the environmental point of view (trophic state). For this, the assemblage index Q, described by Padisák et al. (2006), was applied:

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

where p_i is the relative share of the i-th functional group in biomass, and F is the value of the factor estimated from the following components (Borics et al., 2007):

(i) Trophic status (hypertrophic 0; eutrophic 1; meso-eutrophic 2; mesotrophic 3; oligo-mesotrophic 4; oligotrophic 5);
(ii) Turbulence character; habitats are scored by their turbulence preference, from absolutely standing waters (0) to highly lotic habitats (5);
(iii) Residence time sufficient for the development of the given assemblage. Values range from 1 to 5. The lowest value is assigned to large-celled climax assemblages requiring relatively long time for development. The highest value is assigned to small-celled pioneer assemblages;
(iv) Expert opinion (varies from high risk marked with 0, to low risk, marked with 5) expresses how the occurrence of the given assemblage reflects pollution or toxicity in riverine ecosystems.

The designated values of each component were summed up and the value of the factor F was calculated. All functional groups were provided with the value of the factor F, which ranges from 0 to 5 (Borics et al., 2007). For instance, the sum of the points from 1 to 3 yields 0 for the value of the factor F and indicates that this functional group occurs in the climax or pre-climax phytoplankton assemblages of eutrophic standing water bodies. The sum of the points 18–20 yields 5 for the value of the factor F and indicates that this functional group occurs in true riverine phytoplankton assemblages. Intermediate values of the factor F indicate transition of phytoplankton assemblages from lentic to lotic habitats (Borics et al., 2007).

The value of the assemblage index Q varies also from 0 to 5. The lowest value of Q is characteristic of highly lentic phytoplankton assemblages in hypertrophic lakes, and the highest value is characteristic of benthic diatom assemblages in highly lotic habitats including rivers and rivulets. Along rivers from the upper to the lower reach and below lakes, potamoplanktic assemblages are usually enriched with euplanktonic elements and thus the value of Q decreases in the lower reach (Borics et al., 2007).

In Hungary the values of the EQR have been recommended for different water quality classes. Boundaries between the quality classes were derived from a detailed analysis of large data sets of Hungarian rivers (Borics et al., 2007).

For testing the Hungarian potamoplanktic method in the Narva River, we classified each algal species in the phytoplankton samples into appropriate functional groups according to Borics et al. (2007) and Padisák et al. (2009). In our study the representative taxon was the species with the largest relative share of the total phytoplankton biomass. All taxa except *Stephanodiscus binderianus* that we found in the Narva River belonged to the list of the phytoplankton functional groups. We assigned this diatom to the functional group with code B taking into account its morphological and ecological similarity to other species in this group. Then the relative biomass of each functional group was calculated in the sample and multiplied by the value of the factor F given to the corresponding functional group by Borics et al. (2007). The sum of these scores was taken as the index Q for the phytoplankton of the Narva River. For the assessment of the water quality

of the Narva River, the EQR values were calculated ($EQR = Q/5$) and compared with the EQR values of water quality classes according to Borics et al. (2007): ≥0.8 for 'excellent', 0.7 for 'good', 0.6 for 'moderate', 0.5 for 'poor', and <0.5 for 'bad' quality class.

Statistical analysis

Data processing was performed with the computer program STATISTICA 8.0. Nonparametric Spearman correlation analysis was used to measure the relationship between the relative share of the phytoplankton functional groups and the environmental parameters. The relative share of each functional group from the total phytoplankton biomass was calculated. It ranged from 0 to 1. Cluster analysis, a multivariate technique, was used to group phytoplankton functional groups.

RESULTS

Phytoplankton functional groups and the values of the EQR

Altogether 203 phytoplankton taxa were identified, among them 64 diatoms (Bacillariophyceae), 58 green algae (Chlorophyta), and 47 cyanobacteria (Cyanophyceae). The other groups were Chrysophyceae (12 taxa), Dinophyceae (9 taxa), Cryptophyceae (7 taxa), and Euglenophyceae (6 taxa).

In April–May phytoplankton assemblages were dominated by the X2-group (representative species *Rhodomonas lacustris*) (Table 1). The share of the X2-group in the phytoplankton total biomass was negatively correlated with TP (Spearman $r = -0.58$; $P < 0.05$). In addition, the MP- and B-groups (*Diatoma* sp. and *Aulacoseira islandica* (O. Müller) Simonsen, respectively) were important in 2004 and the Y-group (*Cryptomonas* sp.) in 2006. In June and July the species composition of phytoplankton changed. Although the above-mentioned groups retained their dominance in 2005 and 2006, in 2001 the dominating position was occupied by the D-group (*Stephanodiscus* cf. *hantzschii* Grunow), in 2004 by the P-group (*Aulacoseira granulata* (Ehrenberg) Simonsen), and in 2008 by the J-group (*Pediastrum boryanum* (Turpin) Meneghini). In August 2008 the M-, L_o-, and C-groups (*Microcystis* spp., *Woronichinia naegeliana* (Unger) Elenkin, *Aulacoseira ambigua* (Grunow) Simonsen), respectively) were added to the phytoplankton community. In September–October the phytoplankton community retained its relatively high number of functional groups (Table 1). In these months the K-group (*Aphanocapsa* sp.) dominated in 2001, the Y-group (*Cryptomonas* sp.) in 2003, the J-group (*Crucigenia quadrata* Morren, *Pediastrum* spp.) in 2004 and 2008, and the M-group (*Microcystis* spp.) in 2006.

Ward's method (Euclidean distance) revealed three clusters of the phytoplankton functional groups on the basis of their seasonal occurrence (Fig. 2). The

Table 1. Total number of the functional groups and characteristic algal species for the Narva River in 2001–2008. The codes are given according to Reynolds et al. (2002), Borics et al. (2007), and Padisák et al. (2009). Each functional group is illustrated by one taxon that dominated by biomass in the phytoplankton

Year	Month	Total number of functional groups in phytoplankton	Codes of the dominating functional groups	Dominating taxa by biomass of the functional groups in the Narva River
2001	Apr	11	X2	*Rhodomonas lacustris*
2001	June	9	D+X2+Y	*Stephanodiscus* cf. *hantzschii*, *Rhodomonas lacustris*, *Cryptomonas* sp.
2001	Sep	20	K+M+C	*Aphanocapsa* sp., *Microcystis* spp., *Aulacoseira ambigua*
2003	June	7	T_B	*Navicula* spp.
2003	Sep	12	Y+T_B+C+X2	*Cryptomonas* sp., *Melosira varians*, *Aulacoseira ambigua*, *Rhodomonas lacustris*
2004	Apr	12	X2+MP+B	*Rhodomonas lacustris*, *Diatoma* sp., *Aulacoseira islandica*
2004	July	3	P+K+C	*Aulacoseira granulata*, *Aphanothece* sp., *A. ambigua*
2004	Oct	12	J+L_o+D	*Crucigenia quadrata*, *Snowella lacustris*, *Stephanodiscus* cf. *hantzschii*
2005	June	9	B+X2	*Aulacoseira islandica*, *Rhodomonas lacustris*
2006	May	9	X2+Y	*Rhodomonas lacustris*, *Cryptomonas* sp.
2006	June	8	X2	*Rhodomonas lacustris*
2006	Oct	10	M	*Microcystis wesenbergii*
2008	June	11	J+X2	*Pediastrum boryanum*, *Rhodomonas lacustris*
2008	Aug	15	J+M+L_o+C	*Pediastrum boryanum*, *Microcystis* spp., *Woronichinia naegeliana*, *Aulacoseira ambigua*
2008	Oct	15	J+M+L_o	*Pediastrum* spp., *Microcystis aeruginosa*, *Snowella lacustris*

first cluster included small cryptophytes from the X2-group that occurred mostly in spring and early summer. The second cluster included *Microcystis* species from the M-group that occurred mainly in late summer or autumn. The third cluster included many different functional groups of no seasonal preference.

The values of the EQR in the Narva River showed substantial seasonal variation ranging from 0.6 to 0.8 in spring and from 0.3 to 0.9 in summer (Fig. 3). The median EQR value 0.7 indicated ‘good’ water quality in spring and summer.

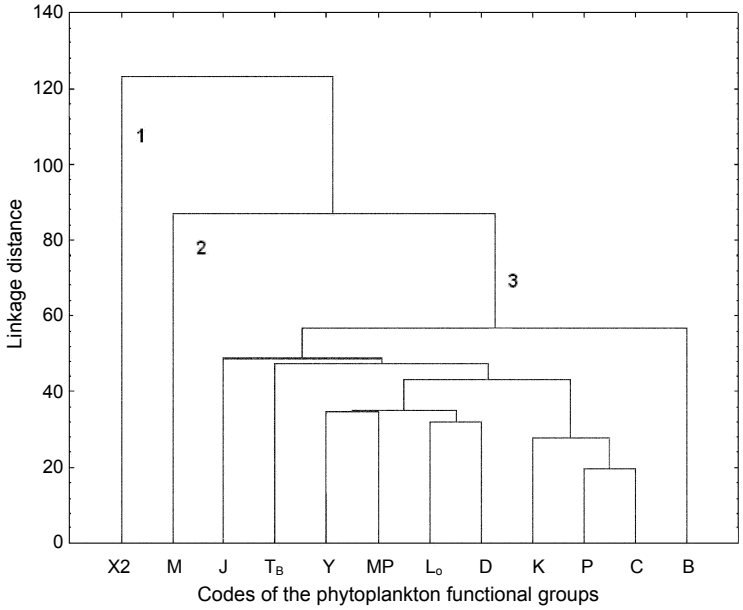

Fig. 2. Dendrogram obtained from the clustering of the phytoplankton functional groups. The codes are given according to Reynolds et al. (2002), Borics et al. (2007), and Padisák et al. (2009). The dominating species for each group are given in Table 1.

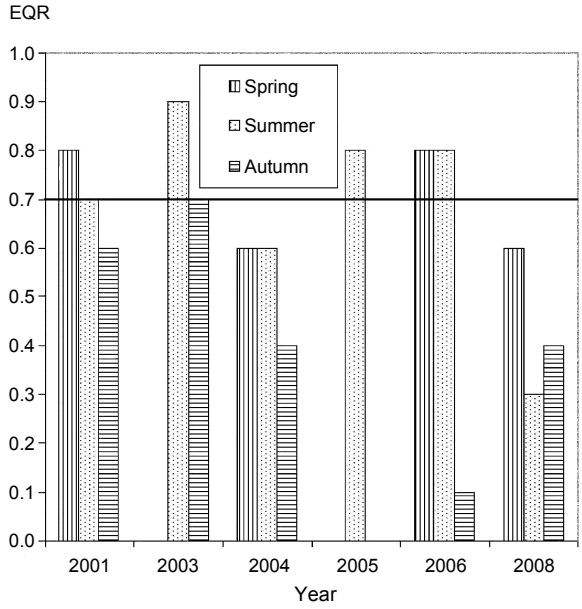

Fig. 3. Ecological Quality Ratio (EQR) of the Narva River in 2001–2008. The horizontal line at 0.7 indicates the good–moderate boundary elaborated by Borics et al. (2007).

In autumn the EQR varied from 0.1 to 0.7, and the median value 0.4 indicated 'bad' water quality. The low EQR values in summer and autumn 2008 were caused by the large share of the cyanobacterium *Microcystis* in the total phytoplankton biomass. The inter-annual median value 0.6 for 2001–2008 indicated 'moderate' quality class.

Water quality in the Narva River on the basis of hydrochemical parameters and benthic diatom indices

Boundary values for different water quality classes of the Narva River have been developed for the concentrations of dissolved O_2, BOD_5, TP, TN, and NH_4-N (Pinnaveekogumite …, 2009). In 2001–2008 the mean values of dissolved O_2 and NH_4-N indicated 'excellent' water quality, the mean values of TN and TP indicated 'good' water quality, and the mean value of BOD_5 indicated 'moderate' water quality (Table 2). According to these criteria (Pinnaveekogumite …, 2009), the final assessment for the lower reach of the river (Narva-Jõesuu) in 2001–2008 varied between 'good' and 'excellent'.

The median values of the diatom metrics IPS (Specific Polluosensitivity Index; Coste in CEMAGREF, 1982), TDI (Trophic Diatom Index; Kelly & Whitton, 1995), and WAT (Watanabe's Index; Watanabe et al., 1990) indicated 'good' or 'excellent' quality class for the lower reach of the Narva River in the summer periods of 2006–2008 (Tuvikene et al., 2006, 2009) (Table 2).

Table 2. Boundaries for different water quality classes of the Narva River worked out on the basis of hydrochemical parameters O_2, BOD_5, TP, TN, and NH_4-N (Pinnaveekogumite …, 2009) and diatom indices (Tuvikene et al., 2006, 2009); their median, minimum, and maximum values, and water quality assessment for the Narva River in 2001–2008

Parameter	Water quality classes for the Narva River				Median and min–max	Water quality class
	Excellent	Good	Moderate	Poor		
Dissolved O_2, %	>70	70–60	60–50	50–40	97 (91–104)	Excellent
BOD_5, mg L^{-1}	<2.0	2.0–2.5	2.6–4.0	4.1–5.0	3.0 (1.7–3.7)	Moderate
TP, mg L^{-1}	<0.04	0.04–0.06	0.07–0.08	0.09–0.1	0.05 (0.02–0.07)	Good
TN, mg L^{-1}	<0.5	0.5–0.7	0.8–1.0	1.1–1.5	0.72 (0.42–0.89)	Good
NH_4-N, mg L^{-1}	<0.10	0.10–0.30	0.31–0.45	0.46–0.60	0.02 (0.01–0.07)	Excellent
IPS	>15.5	15.5–12.0	12.0–9.6	9.5–6.9	13.2*	Good
WAT	>15.9	15.9–12.4	12.4–9.8	9.7–7.1	15.5*	Good
TDI	<75	75–79	80–84	85–90	68.8*	Excellent

* Median values of the diatom indices IPS (Indice Polluosensitivité Spécifique), WAT (Watanabe's Index), and TDI (Trophic Diatom Index) are given for the summer period of 2006–2008.

DISCUSSION

Algal assemblages are useful indicators for environmental monitoring of rivers as they integrate the impact of human activities. In running waters algal communities are influenced by the size of the catchment area (Billen et al., 1994), water residence time (Reynolds et al., 1991), and nutrient concentration (Van Nieuwenhuyse & Jones, 1996; Basu & Pick, 1997; Koch et al., 2004; Piirsoo et al., 2007). For streams and small rivers, benthic diatoms are good indicators (Ács et al., 2003, 2004, 2006; Szabó et al., 2004; Hering et al., 2006; Hlúbiková et al., 2007; Kelly et al., 2009). For large rivers, application of benthic diatoms is problematic owing to the lack of appropriate substrate for algae. For large and slow-flowing rivers, water quality can be evaluated by phytoplankton indices (Borics et al., 2007; Trifonova et al., 2007). Phytoplankton reflects water quality through changes in its community structure, patterns of distribution, and the proportion of sensitive species. The EQR value that is based on the functional groups of phytoplankton is especially informative (Borics et al., 2007) as the functional groups are distinguished by major adaptive features not specific for one or a few phylogenetic groups (Reynolds et al., 2002).

Variation in the functional groups in the Narva River reflected the seasonal dynamics of riverine phytoplankton, on the one hand, and the impact of Lake Peipsi, on the other hand. Peipsi is under strong anthropogenic pressure. Approximately 90% of nitrogen and 95% of phosphorus are transported into Peipsi by rivers (Blinova, 2001). High concentrations of phosphorus are accumulated also in the bottom sediments (Kangur et al., 2003). Hydrological regime, especially water level and water temperature, are also important factors for phytoplankton development in Peipsi (Milius et al., 2005). The moderately calcareous water of Peipsi is favourable for the growth of cyanobacteria (Kapanen et al., 2008). Phytoplankton biomass shows the strongest correlation with TP (Kangur et al., 2002), but the dominance of *Microcystis* species in the low water period is explained by the higher ratio of the mineral forms of nitrogen and phosphorus (Haldna et al., 2008). The mean annual discharge of the Narva River is relatively large (400 $m^3 s^{-1}$), but its length is relatively short (77 km). Therefore, the influence of Lake Peipsi on the Narva River was notable. The impact of the Narva Reservoir on riverine phytoplankton was rather weak because of the rapid turnover in the reservoir, 34–35 times a year (Mištšuk & Jaani, 2000). Monitoring data (Narva veehoidla …, 2001–2008) and our study (results not shown here) indicated that the Narva Reservoir is relatively poor in phytoplankton.

In spring the Narva River is characterized by riverine phytoplankton: planktic cryptophytes from the functional X2-group dominated in the lower reach (Cluster 1 in Fig. 2). Favourable habitats for this functional group are small meso- or eutrophic environments (Reynolds et al., 2002; Padisák et al., 2009). Its relatively high *F* value indicated that riverine conditions were also favourable for this group. Small flagellates are able to maintain a high degree of reproductivity in turbulent river water (Reynolds, 1988). This group is common also for other

Estonian rivers (Piirsoo, 2001, 2003). In summer the phytoplankton of the Narva River changed as a result of enrichment with species from many functional groups (Cluster 3 in Fig. 2). Co-occurrence of different functional groups is a rule rather than an exception (Padisák et al., 2009). Favourable habitats for the functional groups occurring in the Narva River in the summer period (Table 1) are mostly meso- and eutrophic waters or highly lotic habitats (Padisák et al., 2009). In late summer and autumn *Microcystis* species from the M-group were very abundant (Cluster 2 in Fig. 2). This group is typical for Lake Peipsi (Laugaste et al., 2008). Its low *F* value indicates a strong impact of Lake Peipsi on the Narva River.

The system of biological indicators is sophisticated and a single indicator cannot reflect the ecological status of a waterbody. Algae respond rapidly to changes in water quality owing to their high reproduction rate. Benthic diatoms are used for assessing acidity (Coring, 1996; Battarbee et al., 1997) and nutrient enrichment (Kelly et al., 1995; Coring, 1999; Kovács et al., 2006). In this study comparison of assessments on the basis of phytoplankton and benthic diatom metrics revealed their relative agreement in the summer period: both showed at least 'good' water quality in the Narva River. Earlier diatom indices were found to correlate with nutrient concentration in Estonian streams (Vilbaste, 2004; Vilbaste et al., 2007). The phytoplankton community is influenced by phosphorus concentration (Piirsoo et al., 2007). This means that both phytoplankton and benthic diatom metrics are sensitive to water quality. However, as their lifespan is short, algae indicate alterations in water quality during a few days or weeks.

To sum up, the assemblage indices *Q* and EQR are assessment metrics that have been elaborated and tested on a large phytoplankton data set for rivers of Hungarian lowlands and the Carpathian ecoregion. Despite the fact that the values of the factor *F* were partly based on expert judgment and the setting of the boundaries of the EQR was rather subjective (Borics et al., 2007), the method seemed to be suitable also for a large Estonian lowland river where benthic diatoms may be absent due to lack of appropriate substrates. The Narva River is short and its phytoplankton are similar to those of Lake Peipsi (Laugaste et al., 1996, 2008). Therefore, the variation in the functional groups of phytoplankton reflects the seasonal dynamics of riverine phytoplankton on one hand and the impact of the lake on the other hand. Assessment of water quality by using phytoplankton variables should be based on seasonal samples collected during the vegetation period. The numerical values of the EQR for different water quality classes should be elaborated more precisely on the basis of a larger phytoplankton data set of Estonian rivers.

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Narva jõe veekvaliteedi hindamine Ungaris väljatöötatud potamoplanktilise meetodi abil

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On hinnatud Narva jõe veekvaliteeti Ungaris väljatöötatud meetodi abil, mis põhineb fütoplanktoni funktsionaalsetel gruppidel. Arvutati koosluses olevate erinevate funktsionaalsete gruppide suhteline osakaal fütoplanktoni biomassis ja see korrutati igale funktsionaalsele grupile eriomase faktorväärtusega *F*. Korrutiste summeerimisel saadi fütoplanktoni koosluse indeks *Q*. Seejärel arvutati ökoloogilise kvaliteedi indeks (EQR) ja võrreldi selle väärtusi 'suurte jõgede' tüübispetsiifiliste väärtustega erinevates veekvaliteedi klassides. Leiti, et Narva jõe veekvaliteet muutub sesoonselt, kuuludes kevadel ja suvel kõige sagedamini kvaliteediklassi 'väga hea' või 'hea', kuid sügisel kõige sagedamini kvaliteediklassi 'väga halb'. Sügisene halb veekvaliteedi hinnang oli põhjustatud Peipsi järve sinivetikate suurest osakaalust Narva jõe fütoplanktonis. Ungaris väljatöötatud potamoplanktilist meetodit veekvaliteedi hindamiseks võib kasutada suurtes ja sügavates jõgedes, kus bentiliste ränivetikate kasv on takistatud sobivate substraatide puudumise tõttu. Kuna fütoplankterid on lühiealised ja peegeldavad suhteliselt lühiajalisi veekvaliteedi muutusi, peaks üldhinnang põhinema fütoplanktoni proovidel, mis on kogutud erinevatel aastaaegadel kogu vegetatsiooniperioodi jooksul. Veekvaliteedi klassipiirid Eesti jõgede jaoks tuleks täpsustada suurema fütoplanktoni andmebaasi abil.