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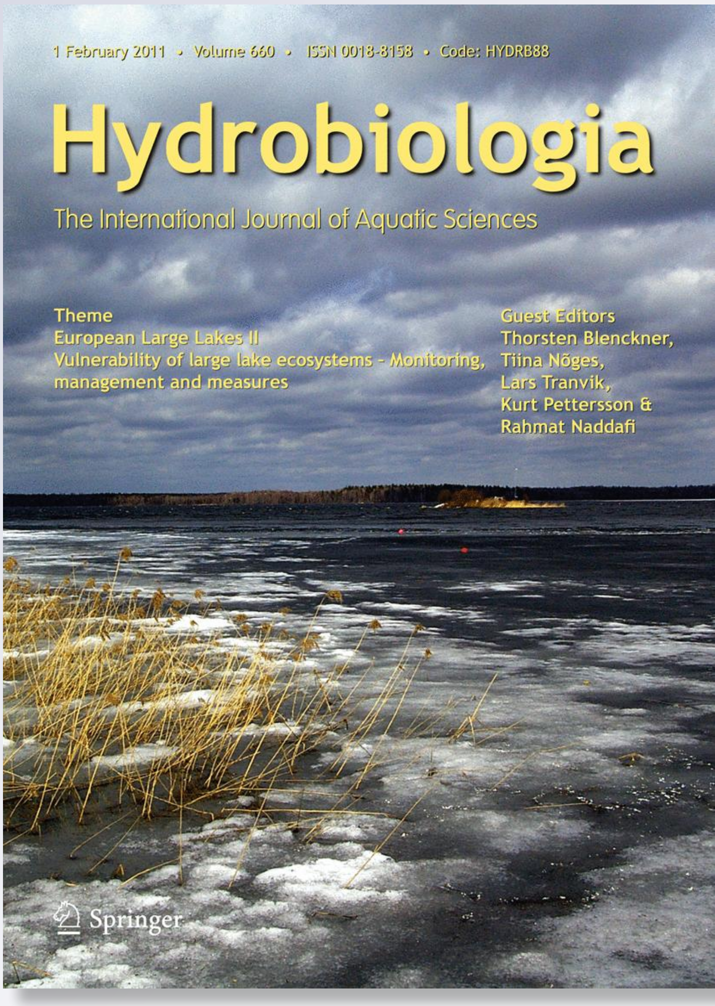

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EUROPEAN LARGE LAKES II

Why do phytoplankton species composition and "traditional" water quality parameters indicate different ecological status of a large shallow lake?

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Abstract Long-term data on phytoplankton species composition in large and shallow Lake Võrtsjärv indicated a sharp deterioration of the ecological status at the end of the 1970s. The more traditional water quality indicators, such as the concentrations of nutrients and chlorophyll a, phytoplankton biomass, and Secchi depth, failed to capture this tipping point or even showed an improvement of the status at that time. As the shift coincided with a large increase of the lake's water level (WL), we hypothesized that direct effect of the changing WL on traditional water quality indicators might have blurred the picture. We removed statistically the direct effect of the WL and the seasonality from the traditional water quality indicators in order to minimize the effects of natural variability. The average of the standardised water quality indicators, used as a proxy for the ecological status, distinguished a period of fast eutrophication in the first half of the 1970s (not captured by the phytoplankton species index), a fast improvement at the end of the 1970s (when the species index showed deterioration) followed by a continuous deterioration trend (when the species index remained rather constant). The causes of this inconsistency are discussed in the light of the alternative stable states theory and the priority of biotic indicators stipulated by the EU Water Framework Directive.

Keywords Water Framework Directive · Phytoplankton taxonomic index · Trophic state indicators · Long-term data · High natural variability · Alternative stable states

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Introduction

The principal legislative tool in the field of water policy in Europe, the Water Framework Directive (WFD; Directive, 2000) defines the status of water bodies by the extent of anthropogenically derived deviation from the reference conditions, i.e. conditions that should occur at sites of any particular type in the absence of human impact. Still, the latter is often overshadowed by the natural variability appearing at longer or shorter time scales (Nõges et al., 2007a, b). The following natural factors may have remarkable influence on parameters commonly used to assess ecological status:

(1) Diurnal changes in the physical, chemical and biological variables, some of which are regular due to daily cycle (e.g. photosynthesis and respiration), still comprise a stochastic component deriving from meteorology.
(2) Seasonal changes take place with well-known regularity from year to year. The randomness is added to them due to differences in the meteorological conditions between years, causing deviations in seasonality and phenology.
(3) The prolonged changes in atmospheric circulation patterns such as the North Atlantic Oscillation (NAO; Hurrell, 1995) or El Nino Southern Oscillation (ENSO; Philander, 1990) affect through different mechanisms physical, chemical and biological properties of water bodies. Both have shown to cause large fluctuations in affluence and lake water levels WLs (Rodó et al., 1997).

To avoid the effect of diurnal differences in the data, the monitoring programmes usually determine a certain sampling time for a water body (Loftis et al., 1991). To remove seasonality from data, several methods of time series analysis such as seasonal decomposition (Cleveland & Tiao, 1976) and seasonal smoothing (Gardner, 1985) exist. However, in practice, large amounts of monitoring data are omitted due to seasonality problems and the status assessment of water bodies is often based on data of a single season only, mostly summer. Using seasonal averages requires regular and comparable data coverage for all years. The decadal scale periodicities caused by atmospheric circulation patterns can be revealed only in really long-term monitoring data and still no standard solution exists to eliminate them.

The sensitivity of lakes to natural variability factors depends strongly on their morphometry, and the role of physical drivers like wind and WL in controlling the ecosystem processes increases with increasing lake area and decreasing depth (Nõges, 2009). While large and shallow polymictic lakes are extremely sensitive to natural physical drivers (Scheffer, 2004; Nõges et al., 2007a, b; Scheffer & van Nes, 2007), in deep lakes, the in-lake biological and chemical factors prevail (Tilzer & Serruya, 1990). Natural variability that in principle should belong to reference conditions exceeds often the variability caused by anthropogenic factors. As the target variables of both the types of variability largely overlap in natural waters, it becomes difficult to disentangle their effects that add a large uncertainty to the status assessment. To illustrate this problem, we have chosen as an example the large and shallow Lake Võrtsjärv (Estonia, 270 km^2, average depth 2.8 m), famous by its huge natural variability caused by fluctuating WLs.

Seasonal and inter-annual WL fluctuations exceeding 3 m and modifying the intensity of sediment resuspension, strongly influence all water quality parameters in Võrtsjärv (Nõges & Järvet, 1995; Nõges & Nõges, 1998, 1999). The lake has been identified as an individual type in the Estonian classification of lake status for state monitoring. In a recent study where four phytoplankton taxonomic indices were tested on the long 44-year time series of phytoplankton data from this lake (Nõges et al., 2010a), all indices showed a unidirectional deterioration of the lake's ecological status with a major stepwise change occurring in 1979. To some extent, it was a surprise for us as the nutrient loadings had a decreasing trend since the end of 1980s (Nõges et al., 2010b), and we expected to see an improvement also in the biotic indices. The traditionally monitored water quality indicators, such as the concentrations of total phosphorus, total nitrogen, chlorophyll a, phytoplankton biomass, and Secchi depth, failed to capture the sudden deterioration at the end of the 1970s or even showed an improvement of the status at that time. As all these indicators are strongly influenced by changes in WL, we hypothesized that this factor could blur the picture and cause the contradictory results not allowing a consistent estimation of the ecological status of this lake.

To clarify the trophic state history of the lake, we applied the traditional water quality indicators in which we statistically removed the direct effect of the changing WL and the seasonality in order to minimize the effects of natural variability. In this article, we analyse what caused the inconsistency in the status assessments based on traditional water quality indicators, on one hand, and on phytoplankton species index, on the other hand, and whether it was caused by ignoring the dynamic reference caused by natural variability in this lake.

Materials and methods

We used long-term data on most common trophic state parameters for lakes such as phytoplankton biomass (BM), chlorophyll a (Chl), total nitrogen (TN), total phosphorus (TP), and Secchi depth (*S*) measured at the main monitoring station in Lake Võrtsjärv from May to October. Time series of BM and *S* started from the year 1965, that of Chl from 1982 and those of nutrients from 1983. The assessment system was created by the following steps:

(1) Data standardization. Concentrations (TN, TP, Chl) and biomass were Ln-transformed in order to achieve normal distribution of the variables. In the case of Secchi depth, we used the reciprocal (1/*S*) to make it increase with the trophic state like the other variables, and used then the natural logarithm values as the best approximation to normal distribution. Further, the transformed variables were used as normal ones in all steps of the analysis (regressions, averaging) and the exponents (i.e. the geometric means of the initial variables) were calculated only at the end to indicate the quality class boundaries in their original units.

(2) To correct the data for the effects of changing WL, we used monthly linear regressions between Ln-transformed variables and the WL of the sampling day (Fig. 1). We corrected the variable values by adding ΔY corresponding to the deviation of the WL from the monthly mean value (ΔWL). We applied this correction step only for those variables and months for which the relationship with WL was statistically significant, otherwise the variables remained unchanged.

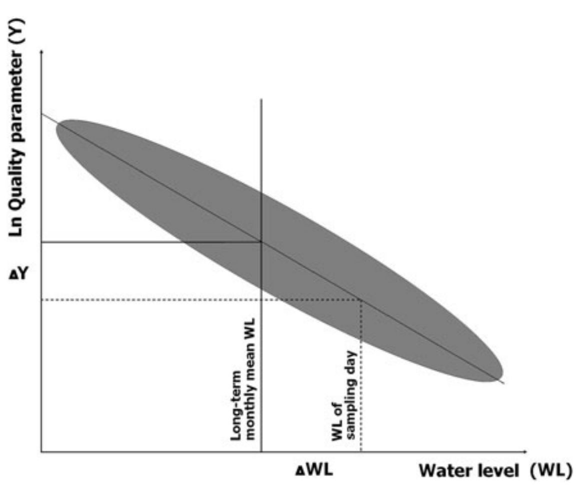


Fig. 1 Principal scheme of correcting water quality parameters for water level changes based on linear regression. *Grey area* marks the data distribution

(3) To remove the seasonality from the data series, we calculated first the monthly and the seasonal (May–October) averages of all corrected variables for all years. We excluded those years from the seasonal average calculations for which less than four of the six monthly values were available. Second, we calculated regressions to derive the seasonal average from single monthly values. Non-significant regressions were omitted. Finally, a new seasonal average was calculated based on those derived from monthly values. In this way, more correct seasonal averages could be calculated for those years for which only few measurements were available. As a result, the new corrected data consisted of approximations of seasonal average values corrected for the effect of WL changes.

(4) To define reference conditions and set the quality class boundaries for individual variables, we supposed, based on historical data and expert opinions (Nõges et al., 2001; Nõges, 2003; Nõges & Nõges, 2006), that for most of the period studied, the lake has been deviating only slightly from the reference conditions described in the second decade of the twentieth century (Mühlen, 1918; von zur Mühlen & Schneider, 1920), i.e., has been in "good" status according to the WFD (Directive, 2000). Hence, we considered that at least one-half of the parameter values (25th to 75th percentile) should indicate "good" status. In line with the guidance document on reference conditions (CIS, 2003), we supposed that the median of the "high" class values (values below the 25th percentile) should describe the site-specific reference conditions for the lake. The upper 10% was considered to characterize the "poor status" (Fig. 2). The analysis of historical biotic changes (Nõges et al., 2001) showed that the lake has never fallen to "bad" status, which, according to WFD (Directive, 2000), is defined by the "…absence of large portions of the relevant biological communities normally associated with the

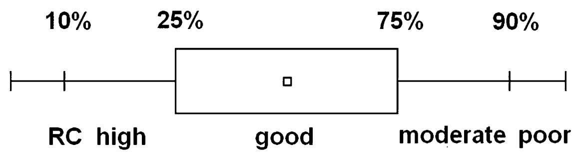


Fig. 2 Classification criteria based on the statistical distribution of geometric mean values of the water quality metrics in Lake Võrtsjärv for the period studied. It was supposed that half of the measurements (25th–75th percentile) should indicate the "good" quality class mostly identified by previous studies. Reference conditions were defined as the median of the "high" class values

surface water body type under undisturbed conditions".

(5) For the final assessment, values from 1 to 4 were given, correspondingly, to the classes "high" (H), "good" (G), "moderate" (M), and "poor" (P). These quality scores of all variables were averaged for the final assessment where the values of 1.5, 2.5, and 3.5 served as the H/G, G/M, and M/P boundaries. Given that the variables were not independent, averaging of them was considered as a pragmatic step to get a summarising status estimate. We expressed the uncertainty of the final estimate as the standard deviation (STDev) of the quality class number.

To characterize the phytoplankton species composition, the PTSI index (Mischke et al., 2008) was calculated for all samples from the study period.

We used the chemical oxygen demand (COD_{Mn}) measured in two periods, 1968–1977 and 1998–2008, as a rough overarching proxy for water colour to find out possible long-term impacts on light conditions.

We used the Mann–Kendall test (Kendall, 1938) for trend analysis and the Worsley likelihood ratio test (Worsley, 1979) to find step-changes in the series.

Table 1 Characteristics of monthly regressions between logarithmic values of trophic state indicators (Y) and water level (cm)

Y	Month	*r*	*P*	Intercept	Slope
LnBM	5	−0.262	0.051 n.s.	2.895	−0.0040
LnBM	6	−0.437	0.001	3.279	−0.0064
LnBM	7	−0.522	0.000	3.318	−0.0077
LnBM	8	−0.409	0.002	3.222	−0.0065
LnBM	9	−0.383	0.003	3.210	−0.0043
LnBM	10	−0.330	0.014	3.238	−0.0037
LnChl	5	−0.249	0.085 n.s.	3.617	−0.0020
LnChl	6	−0.111	0.463 n.s.	3.540	−0.0015
LnChl	7	−0.414	0.004	3.630	−0.0041
LnChl	8	−0.362	0.008	3.714	−0.0040
LnChl	9	−0.161	0.286 n.s.	3.786	−0.0016
LnChl	10	−0.308	0.037	3.900	−0.0025
LnTP	5	0.124	0.427 n.s.	3.658	0.0010
LnTP	6	−0.223	0.157 n.s.	3.868	−0.0024
LnTP	7	−0.108	0.496 n.s.	3.954	−0.0012
LnTP	8	−0.317	0.036	3.967	−0.0052
LnTP	9	−0.455	0.002	4.042	−0.0053
LnTP	10	−0.301	0.050	4.096	−0.0031
LnTN	5	0.234	0.152 n.s.	0.083	0.0019
LnTN	6	0.296	0.063 n.s.	−0.154	0.0024
LnTN	7	0.107	0.530 n.s.	−0.161	0.0015
LnTN	8	−0.071	0.666 n.s.	−0.013	−0.0008
LnTN	9	−0.271	0.075 n.s.	0.104	−0.0022
LnTN	10	−0.023	0.889 n.s.	0.128	−0.0002
Ln1/S	5	−0.184	0.115 n.s.	0.091	−0.0010
Ln1/S	6	−0.405	0.001	0.444	−0.0028
Ln1/S	7	−0.454	0.000	0.425	−0.0026
Ln1/S	8	−0.593	0.000	0.442	−0.0035
Ln1/S	9	−0.612	0.000	0.496	−0.0047
Ln1/S	10	−0.507	0.000	0.405	−0.0034

n.s. Non-significant relationships

Units of initial measurements: phytoplankton biomass (BM) and total nitrogen (TN), mg l^{-1}; total phosphorus (TP) and chlorophyll (Chl), μg l^{-1}; Secchi depth (*S*), m

Results

The values of all selected trophic state indicators except TN were significantly related to the WL during several months of the vegetation period (Table 1), while the relationships were the strongest for the Secchi depth. All significant relationships were negative, i.e. the lake looked significantly more eutrophic at lower WLs. In May, none of the parameters had a significant relationship with WL; in addition, the regression was non-significant for LnChl in June and September, and for LnTP in June and July. In cases when the relationship was non-significant, we used the measured values in the following steps, otherwise the measured values were corrected according to the regression to correspond to the long-term average WL of the month.

On average, the relationships between monthly values of trophic indicators and their seasonal averages (Table 2) were strongest in August and September and weakest in May. The relationship was non-significant for TP in July and for Secchi depth in May.

As a result of the two corrections/transformations, the mean values of the full time series did not change significantly (*P* of *t* test for means between 0.3 and 0.9 for different variables). The correction for WL did not significantly change the total variability of the time series (*P* of *t* test for variance between 0.1 and 0.9 for different variables). The correction for seasonality diminished considerably the variability ranges of all variables (Fig. 3). The total variance decreased fivefold for LnBM, LnChl, and LnTN, nine-fold for LnTP, and 21-fold for Ln1/S.

Despite the unchanged long-term averages, both corrections affected substantially the monthly values (Fig. 4) and the seasonal averages (Fig. 5) as exemplified with phytoplankton biomass.

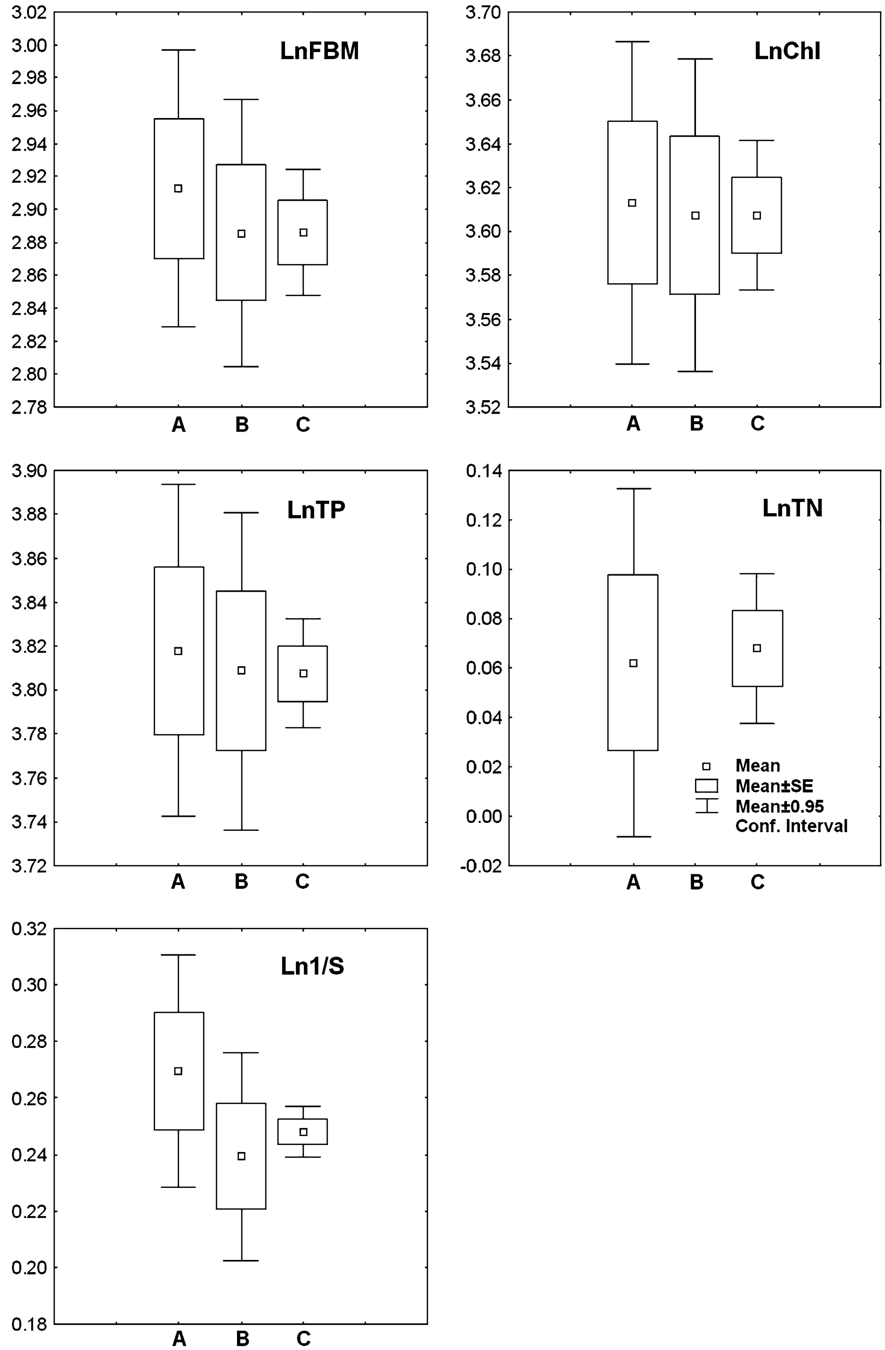
The correction for the effects of WL changes followed closely the dynamics of the mean WL for May–October (Fig. 5) correcting the BM by up to 23% down in low-water years and up to 41% up in high-water years. The seasonality removal had even stronger effect ranging from 28% down to 64% up. In nearly 70% of the cases, the seasonality removal

Table 2 Characteristics of monthly regressions between logarithmic values of trophic state indicators (variable names as in Table 1) corrected for water level changes where appropriate (X) and their seasonal mean values (Y)

X	Y	*r*	*P*	Intercept	Slope
LnBM May	LnBM May–October	0.703	0.0000	1.832	0.438
LnBM June	LnBM May–October	0.589	0.0001	1.516	0.512
LnBM July	LnBM May–October	0.736	0.0000	1.091	0.635
LnBM August	LnBM May–October	0.824	0.0000	1.334	0.485
LnBM September	LnBM May–October	0.784	0.0000	1.045	0.602
LnBM October	LnBM May–October	0.779	0.0000	0.941	0.626
LnChl May	LnChla May–October	0.699	0.0001	1.175	0.702
LnChl June	LnChla May–October	0.703	0.0001	2.415	0.344
LnChl July	LnChla May–October	0.840	0.0000	1.198	0.710
LnChl August	LnChla May–October	0.646	0.0003	1.601	0.556
LnChl September	LnChla May–October	0.773	0.0000	1.545	0.540
LnChl October	LnChla May–October	0.781	0.0000	1.306	0.593
LnTP May	LnTP May–October	0.481	0.0174	2.301	0.404
LnTP June	LnTP May–October	0.711	0.0001	2.458	0.372
LnTP July	LnTP May–October	0.292	0.1657 n.s.	3.156	0.166
LnTP August	LnTP May–October	0.648	0.0005	2.395	0.381
LnTP September	LnTP May–October	0.805	0.0000	1.915	0.491
LnTP October	LnTP May–October	0.604	0.0014	2.493	0.326
LnTN May	LnTN May–October	0.409	0.0426	−0.029	0.316
LnTN June	LnTN May–October	0.600	0.0012	0.063	0.454
LnTN July	LnTN May–October	0.786	0.0000	0.070	0.397
LnTN August	LnTN May–October	0.873	0.0000	0.093	0.563
LnTN September	LnTN May–October	0.600	0.0012	0.052	0.426
LnTN October	LnTN May–October	0.708	0.0001	−0.014	0.532
Ln1/S May	Ln1/S May–October	0.287	0.0802 n.s.	0.251	0.141
Ln1/S June	Ln1/S May–October	0.585	0.0001	0.187	0.279
Ln1/S July	Ln1/S May–October	0.458	0.0084	0.174	0.287
Ln1/S August	Ln1/S May–October	0.543	0.0007	0.137	0.386
Ln1/S September	Ln1/S May–October	0.659	0.0000	0.141	0.330
Ln1/S October	Ln1/S May–October	0.580	0.0003	0.162	0.244

n.s. Non-significant relationships

Fig. 3 The effect of corrections on the mean values and variability ranges of different trophic state variables. *A* uncorrected variables, *B* variables corrected for the effect of water level changes (was not done for TN because of missing relationship), *C* variables corrected both for water level changes and seasonality. Variable names as in Table 1



corrected the data to the same direction with the correction for WL giving a summary effect between 41% down and 65% up.

For the whole 44-year period analyzed, there was no significant trend in the WL (Fig. 5); however, there was a stepwise jump between 1977 to 1978 (Worsley test, $P < 0.01$). Since 1978, the WL had a slight decreasing trend ($P < 0.05$). The corrected biomass series (Fig. 6) indicated an abrupt decrease from 1978 to 1979 followed by a significant ($P < 0.01$) increasing trend since that. The decrease in phytoplankton biomass was reflected also in improved Secchi transparency (corrected series) although the water turned brown in this period obviously due to humic substances carried into the lake during the rainy 1978. The corrected Secchi depth had an increasing trend from 1981 to 1992 and decreased since that ($P < 0.01$; note that in the Fig. 6 there is shown $1/S$ to make S increase with the trophic state like other variables). The time series of total nutrients corrected for the WL effects did not show any significant trend.

The COD_{Mn} levels measured in the period 1968–1977 (10.6 ± 3.2 mg O l^{-1}) were significantly ($P < 0.01$) lower compared with those in 1998–2008 (13.0 ± 1.5). Within both periods, COD_{Mn} had a highly significant ($P < 0.01$) increasing trend.

The average trophic state proxy index calculated by applying the class boundaries of trophic state indicators (Table 3) to the long-term data (Fig. 7) showed two distinctive periods in the changes of the ecological status of the lake: the initial fast eutrophication in the 1970s (assessed only on the basis of BM and S) followed by a temporary improvement in 1979–1980 and a worsening trend afterwards. The latter trend for the whole period was clearly seen also without using the additional data (TP, TN, Chl) for posterior years, and was not caused by the difference

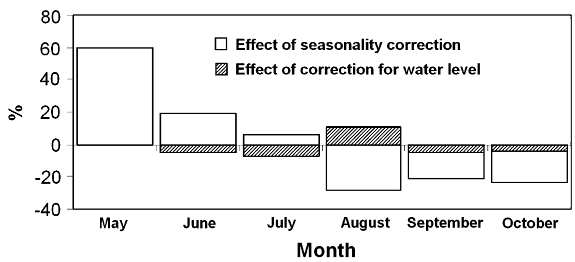


Fig. 4 The effect of corrections on monthly mean values of measured phytoplankton biomass

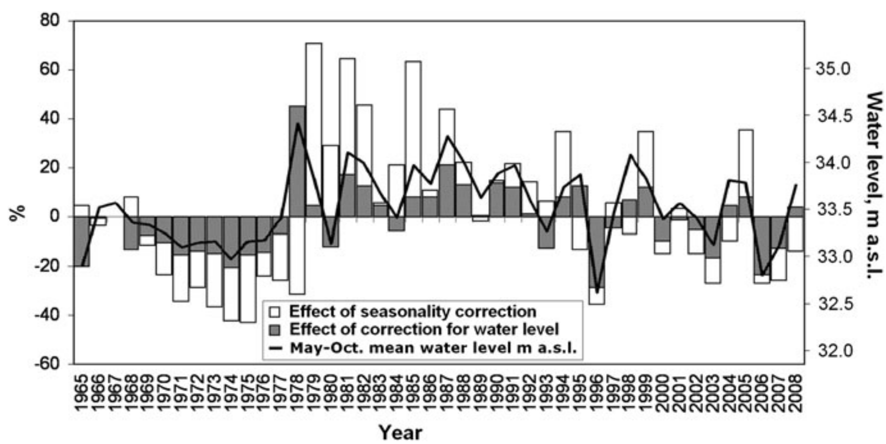


Fig. 5 The effect of corrections on seasonal mean values of measured phytoplankton biomass and the changes in May–October mean water level

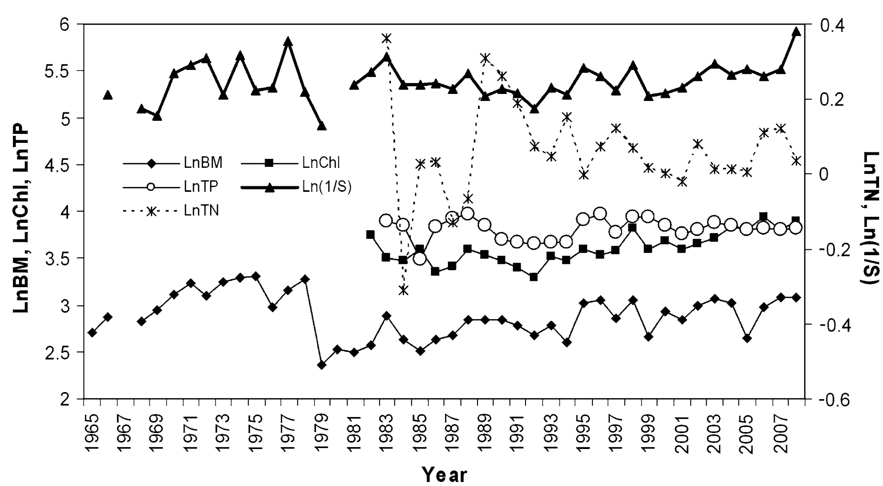


Fig. 6 Long-term changes of the May–October mean values of the common trophic state variables corrected for the changes in the water level in Lake Võrtsjärv. Variable names as in Table 1

Table 3 Reference conditions and class boundaries for the geometric mean values of trophic state parameters (variable names as in Table 1) corrected for seasonal variability and the effect of water level changes in Lake Võrtsjärv

Percentile Class boundary	RC	25% H/G	75% G/M	90% M/P
BM, g m^{-3}	13.4	14.5	23.2	27.2
Chl, μg l^{-1}	30.4	32.3	43.6	48.8
TP, μg l^{-1}	39.3	42.8	50.2	54.2
TN, mg l^{-1}	0.9	1.0	1.2	1.3
S, m	0.90	0.82	0.74	0.70

Reference conditions were calculated as the median value of the "high" class

in data availability. Since 2001, the data show an accelerated deterioration of the status, although also the uncertainty of the estimate has increased in this period.

The German PTSI index (Mischke et al., 2008); however, showed different results for the period since 1979 (Fig. 8) when due to the change in dominant species (earlier *Planktolyngbya limnetica* (Lemm.) was replaced by *Limnothrix redekei* (Goor) Meffert and *L. planktonica* (Wołosz.) Meffert.). The taxonomic index revealed an irreversible drop of the ecological status of the lake. Hence, our hypothesis that the inconsistency of the assessment results based

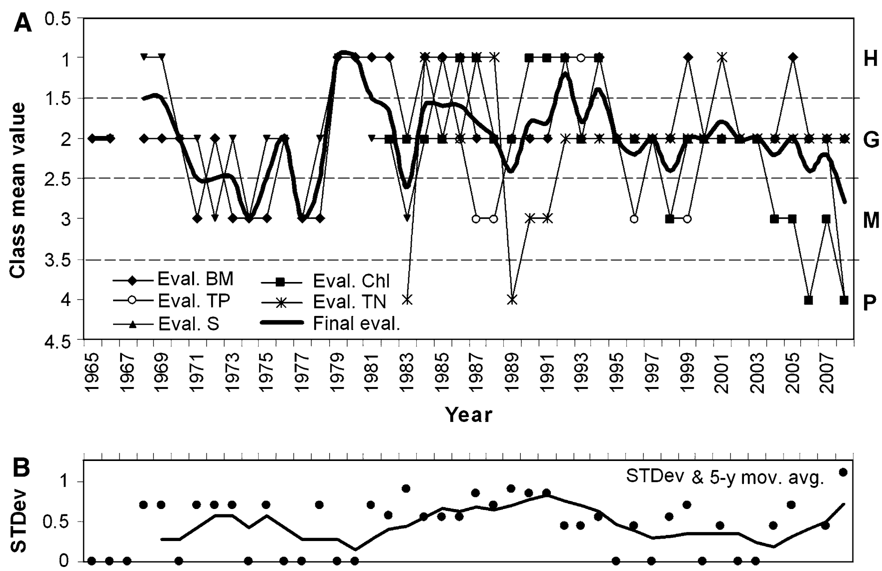


Fig. 7 Long-term changes in the ecological status of Lake Võrtsjärv based on trophic state parameters and corrected for seasonality and water level changes (**A**), and the standard deviation of the final evaluation (**B**). Variable names as in Table 1

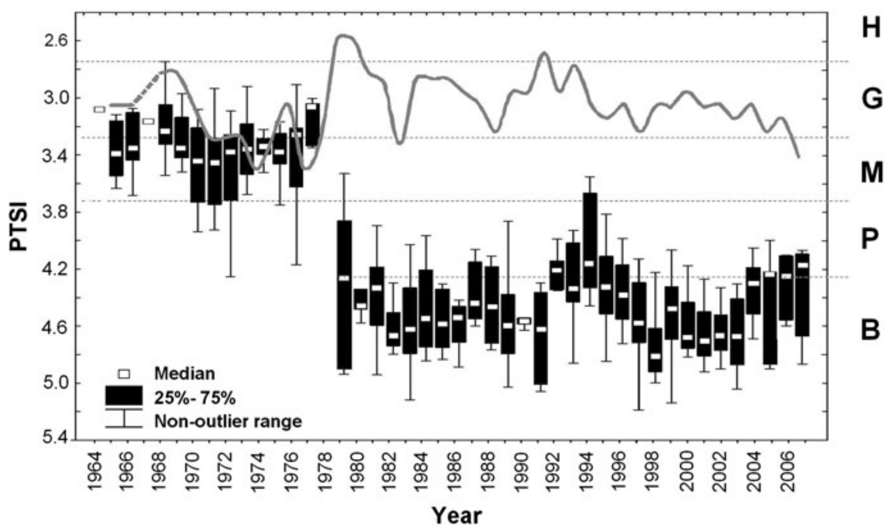


Fig. 8 Long-term changes in the average status index based on traditional trophic state parameters (*grey line*) and the phytoplankton taxonomy-based PTSI index (*box* and *whiskers*) in Lake Võrtsjärv

on phytoplankton taxa and traditional trophic state indicators was caused by the effect of changing WLs, was disproved.

Discussion

The ultimate goal of lake monitoring should be the establishment of a coherent and comprehensive overview of the ecological and chemical status of lakes in nearly real-time regime to enable water managers to take measures if the conditions deteriorate as a result of human impact. Selecting those among the multitude of measurable physical, chemical and biological parameters that reliably reflect the effects of human activities, but remain insensitive to extraneous conditions is a step of extraordinary importance (Karr & Chu, 1997). Given the often strong effects to multiple causal factors operating simultaneously, it is unlikely to find such suitable metrics in ecosystems strongly physically controlled by natural factors. Different ways have been used to disentangle the effects of natural and anthropogenic variability in long-term data ranging from simple detrending (George et al., 2004), applying coefficients for residual adjustment (Reist, 1986), statistical partialling of the effects of other factors and variance decomposition (Rodríguez & Magnan, 1995), to linear (Carstensen & Henriksen, 2009) and nonlinear regression models (Massol et al., 2007).

For simplicity and transparency reasons, we selected the linear adjustment method to statistically account for the effects of the changing WL on the common trophic state variables. In this way, the direct effect caused by WLs deviating from the long-term monthly averages could be eliminated while retaining the meaningfulness and original dimensions of the variables. The considerable correction of single seasonal mean values by more than ±40% shows the high importance of WL changes in shaping these variables. The correction dampened efficiently the effect of the record low WL in 1996 when most of the common water quality indicators were far out of range indicating strong hypertrophy (Nõges & Nõges, 1999).

To cope with the strong seasonality of trophic state variables, often values of only some months, seasonal averages or seasonal maxima are used for the status assessment. In this way, correct assessment cannot be carried out for years with data gaps for relevant months while, on the other hand, part of the seasonal data is omitted from the analysis. In principle, if seasonal monitoring is carried out, the calculated indicator should make use of all available data. Reducing of datasets is acceptable only if proven that a more precise indicator can be obtained from a subset of data (Carstensen, 2007). The method we choose for seasonality removal allows using data from all months in which the variable has a significant relationship with the seasonal mean value. This way of standardizing has three main advantages: (i) each single measurement can be equally used for the assessment purposes, (ii) more correct seasonal averages can be calculated for years with data gaps, and (iii) the obtained values are ecologically meaningful as estimates for the vegetation period mean value.

The calculated summary index revealed two distinct periods in the ecological status of Võrtsjärv: the initial fast eutrophication of the lake in the 1970s followed by a temporary improvement after the high water year of 1978 (Fig. 6) and a slow continuous deterioration trend. The German PTSI index (Mischke et al., 2008) distinguished even more clearly the same periods (Fig. 7). Paradoxically, the taxonomic index indicated a drop of the ecological status from "good" to "moderate" to "bad" where the average trophic index indicated an improvement. Although the scale of the PTSI index was not adapted for Võrtsjärv and the "bad" status was obviously exaggerated, the divergence of the two indices remains a fact. Indeed, during the fast nutrient enrichment in the first half of the 1970s, phytoplankton biomass increased, but no remarkable changes were observed in the species composition. The increase in the WL by more than 1 m in 1978 changed the conditions considerably. Due to less resuspension, the nutrient concentrations and phytoplankton biomass decreased and water became less turbid. However, the nearly 50% increase in the mixing depth due to higher water probably strengthened light limitation and created favourable conditions for two highly shade tolerant *Limnothrix* species, which replaced the previous dominating cyanobacterium *P. limnetica*. The significant difference between the COD_{Mn} values measured in the 1970s and in the later period suggests a possible increase in water colour that could be an additional supporting factor explaining the success of *Limnothrix*. Oscillatoriales, either by *Planktothrix* or *Limnothrix* species dominate in several very shallow polytrophic lakes creating steady-state communities, while a lower phosphorus requirements and a lower light tolerance are the possible advantages for *Limnothrix* over *Planktothrix* (Rücker et al., 1997). Though the WL changed as much from 1996 to 1998 as in 1977–1978 (Fig. 5), the very low level in 1996 did not change permanently the species composition (Nõges & Nõges, 1999). This indicates that the dynamics of cyanobacteria is not well understood.

The German index is calculated as biomass weighed average product of trophic scores showing the average trophic preferences of indicator species, and stenoecy factors showing the indicator power of the species (i.e. how specifically they indicate the given trophic state). A comparison of the values of these parameters applied in the German index for the dominating cyanobacteria species in Võrtsjärv shows that they indicate nearly the same trophic status but the stenoecy factors differ substantially (Table 4). Consequently, the jump in the index can be explained by the lake having reached a high enough trophic state where all three species could coexist, and a strengthening of light limitation at which the more specifically adapted *Limnothrix* species outcompeted the more eurytrophic *P. limnetica.* No reversal has still happened as this would require a reduction in both light limitation and trophic state. As at the moment of the change in dominants triggered by light limitation, no further increase in trophic status was required (the trophic score of *L. redekei* is even lower than that of *P. limnetica*), the traditional trophic state indicators did not reflect it. On the contrary, they showed an improvement caused by the dilution of substances and weaker sediment resuspension caused by the higher WL.

The use of a stenoecy (=specificity) factor, i.e., a weighting factor that describes the degree of constancy with which a taxon can be detected within its proposed preference range, is a common practice in bioindication (e.g., Zelinka & Marvan, 1961; Gervais et al., 1999; Frédéric & Luc, 2005). This factor gives high weights to the specific indicator species which sometimes may be not numerous, like it is often the case with character species used in phytocoenology. The weighting factors, however, overemphasize the importance of dominant species. In our case the PTSI index indicated a sharp deterioration of the status resulting from the switch of the dominants to more shade tolerant species. On the other hand, this change indicates to an aggravation of the situation, as the domination of *Limnothrix* species means a switching of the system to a steady state. This steady state presents a self-induced habitat, in which competitors fail because of low-light conditions are reproduced by the dominants based on efficient exploitation of nutrient resources (Mischke & Nixdorf, 2003). Still, it may be not so clear for Lake Võrtsjärv where the influence of resuspension and humic substances on light conditions is also remarkable and the dominance of *Limnothrix* species is strongly influenced by external conditions as precipitation, temperature and WL.

Table 4 Trophic scores and stenoecy factors of the cyanobacteria species dominating in Lake Võrtsjärv as applied in the German phytoplankton index for polymictic lowland lakes (Mischke et al., 2008)

Species	Trophic score	Stenoecy factor
Planktolyngbya limnetica	5.18	1
Limnothrix redekei	4.68	2
Limnothrix planktonica	5.40	4

There remains the question, should we trust one of these indices to correctly interpret the lake monitoring results or further research is needed to develop a suitable assessment system for this complicated physically driven lake. Both of the indices have their advantages and disadvantages. Due to strong resilience of the phytoplankton community, the taxonomy based index did not almost change during the fast eutrophication in the beginning of the 1970s, neither demonstrated clear patterns in the period after the change of the dominants. The big change occurring in 1979 was brought upon not so much by a change in the trophic state but expressed the tipping point evoked by a disturbance—the sudden increase of the WL. Such behaviour of the index throws doubt upon the popular belief that phytoplankton provides a good indication of lake trophic state and respond quickly and predictably to changes in nutrient status (e.g. Murphy et al., 2002). Also Kaiblinger et al. (2009) who tested phytoplankton indices on three large perialpine lakes to analyze their suitability for trophic classification, concluded that the indices were only appropriate to roughly distinguish lakes of different water quality but were not sensitive enough to track changes that occur within a lake. In several shallow lakes reduction of nutrient loads has not led to discernible recovery. The main causes of delay are phosphorus storage and its subsequent release from sediments (Van Liere & Gulati, 1992). Internal loading and the mechanism of hysteresis, i.e. less nutrient concentrations are needed for recovering the previous better equilibrium state than it was at the time of its decline (Scheffer et al., 1993, 1997; Jeppesen et al., 2007), offer an explanation for the resistance of cyanobacteria dominance in shallow lakes to restoration efforts by means of nutrient load reduction. In that way, phytoplankton indices really reflect the ecological effect of human impact, which can last much longer than the direct impact itself. In addition, usually several other factors like weather conditions or spatial heterogeneity, and even the cyanobacterial dominance itself, play role in developing an alternative regime (Scheffer & van Nes, 2007).

The index based on traditional trophic state variables and corrected for the simultaneous WL changes demonstrated an evolution of the trophic state more consistent with the expert opinion. However, it did not capture the tipping point occurring in phytoplankton, one of the biggest changes ever observed in the plankton community of Võrtsjärv, and without phytoplankton composition data the actual status would have been misinterpreted.

From the point of view of WFD, which gives a priority to biological indicators, the situation is clear: preceding nutrient loadings caused a pressure on the ecosystem resulting in a regime shift when a sudden disturbance (high WL) broke the resilience of the system. The new degraded stable state has demonstrated strong resistance to remediation measures. However, this interpretation has been put together after a critical comparison of both indices and the initial monitoring data. For a more consistent assessment of the ecological status of the lake, other biological elements such as fish, macrophytes and macrozoobenthos should be included in the assessment as suggested by the WFD.

Conclusions

We suggest that correcting of the metric values used in status assessment for the effects of natural variability factors is a necessary step in order to increase the signal/noise ratio and decrease the uncertainty of the estimate. This step is of utmost importance for strongly, physically driven systems such as shallow lakes where the large variation of driving factors may not only mask the effect of human pressures but also the effect of restoration measures.

In cases of high uncertainty of the status estimate (different metrics or different quality elements give controversial results), the causes of the controversy should be analysed and the more appropriate metrics and elements selected before averaging the results or applying the “one out – all out” principle.

As alternative stable states may exist in water bodies making some of the biological response indicators highly resilient, pressure indicators (e.g. nutrient concentrations) could be used in parallel to reflect the trends and get a more awarding system for assessing the managerial efforts.

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