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Effects of long-term non-point eutrophication on the abundance and biomass of macrozoobenthos in small lakes of Estonia

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Abstract. The effects of eutrophication on macrozoobenthos were studied in small Estonian lakes. Altogether, 380 sites of 107 lakes sampled repeatedly before and during/after significant non-point eutrophication were compared. The data of 1951–1967 were considered the reference samples, and the data of 1972–1995 were used as the test samples. A total of 66 macro- and megazoobenthic variables were studied. The individual weight of chironomid larvae, as well as the abundance and biomass of several animal groups, had changed significantly between the two periods. The possible reasons for the changes are briefly discussed.

Key words: macrozoobenthos, lakes, eutrophication, Estonia.

INTRODUCTION

The majority of Estonian lakes are of glacial origin, but there occur also a number of relic lagoons on the seashore, raised bog pools, and man-made reservoirs (Raukas, 1995). Most of the lakes are small, with a surface area less than 10 km^2. Human influence on the lakes lies mostly in eutrophication and amelioration. Essential changes took place in many Estonian lakes in the 1970s–1980s when intensive eutrophication occurred due to heavy pollution with fertilizers and waste-waters from farms (Ott & Kõiv, 1999). The pH, alkalinity, and the amount of

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organic matter increased significantly between the periods 1951–1969 and 1970–1989. Water stratification became much stronger, and the content of dissolved oxygen near the bottom decreased by half compared to the earlier period. The area covered with the gravel substrate and shelter favourable for macroinvertebrates (mosses in soft-water lakes, or charophytes in hard-water mesotrophic lakes) decreased sharply. The lakes have become more similar; the oligotrophic limnological type has actually disappeared by now (Mäemets et al., 1994; Ott et al., 1999). The other types of lakes with higher buffering capacity, protecting them against enrichment, have been better preserved. As phosphorus and nitrogen concentrations in water for 1950s–1960s are not available, direct hydrochemical comparisons between these periods cannot be made.

In the 1990s the ecological status of many lakes started to improve rapidly due to declining agricultural production and years with more precipitation. In some eutrophic lakes, the mean total nitrogen concentration of 967 mg m^{-3} in 1970–1989 decreased to 954 mg m^{-3} in 1990–1997. At the same time, the total phosphorus concentration decreased from 76 to 41 mg m^{-3} (Ott & Kõiv, 1999).

Macrozoobenthic animals (with body length >2 mm) as biological indicators for lakes are used in three main directions. First, the profundal communities consisting mainly of chironomid and oligochaete species have been widely used in biomonitoring (Giziński, 1978; Wiśniewski & Dusoge, 1983; Schell & Kerekes, 1989; Johnson & Wiederholm, 1990; Mastrantuono, 1993; Lang & Reymond, 1996; Bazzanti et al., 1998; Dinsmore et al., 1999; Meriläinen et al., 2000; Lang, 2001). In addition, the animal communities of the lake littoral, as well as stream macroinvertebrates, can be used as indicators of anthropogenic disturbance (Brodersen et al., 1998; Johnson & Goedkoop, 2000; Holopainen et al., 2001; Lewis et al., 2001; White, 2002). In North America, several multimetric indices have been elaborated on the basis of single metrics of littoral macroinvertebrates (Gerritsen et al., 1998; Blocksom et al., 2002). In paleolimnological studies, the use of cores of profundal sediment enables to monitor the history of pollution (Kansanen et al., 1990; Brooks et al., 2001).

Long-term changes in hydrochemical parameters and in the plankton and macrovegetation of small Estonian lakes are well described in the conditions of increasing eutrophication (Ott & Kõiv, 1999), but zoobenthos has been insufficiently treated in this context. The first attempt to generalize the existing data was made by Timm et al. (1982). The aim of the current study was to compare macrozoobenthic communities in small Estonian lakes before and during/after a eutrophication period.

MATERIAL AND METHODS

The material for the current study was obtained from the benthological archives of the Centre for Limnology, Estonian University of Life Sciences, as well as from some earlier publications (Mäemets, 1968; Riikoja et al., 1973). Most data on macrozoobenthos have been collected either by the authors themselves, or with their participation, beginning from the 1950s.

For quantitative sampling, two modifications of the Ekman bottom grab (the Boruckij, or the Zabolockij types) were used, each with a surface area of 225 cm^2. As a rule, three replicates taken from a station were treated as a single composite sample. The stations (2–3 in the smaller lakes but more in the larger ones) were distributed over the most representative depths and plant associations (if present) for each lake. The samples were washed on silk sieves (mesh size 0.5 mm) and sorted alive, without magnification. Chironomidae, Oligochaeta, Mollusca, and the other animals were preserved separately in 70% ethanol. Their wet biomass was measured on a torsion balance. Very large Mollusca, considered as megazoobenthos (such as Unionidae, *Dreissena*, and *Viviparus*), were counted and weighed separately and were not taken into account when calculating the total number and biomass of macrozoobenthos. The animals were identified to the species level by the authors, or by other experts whenever possible. Altogether 486 species of macro- and megazoobenthos from the bottom grab samples were registered from small lakes of Estonia during 1971–1996. As data on species level are largely not available for the 1950s and 1960s, only 31 higher taxa, distinguished without magnification, were available for analyses.

According to Mäemets (1977), there are eight accumulation-based limnological types in Estonia. The whole database of macrozoobenthos included 1796 samples from 253 lakes belonging to eight lake types. To find out possible changes that took place, 380 various sites in 108 lakes, sampled both during the reference period and the test period, were selected (Figs 1 and 2). Of the local lakes 73%

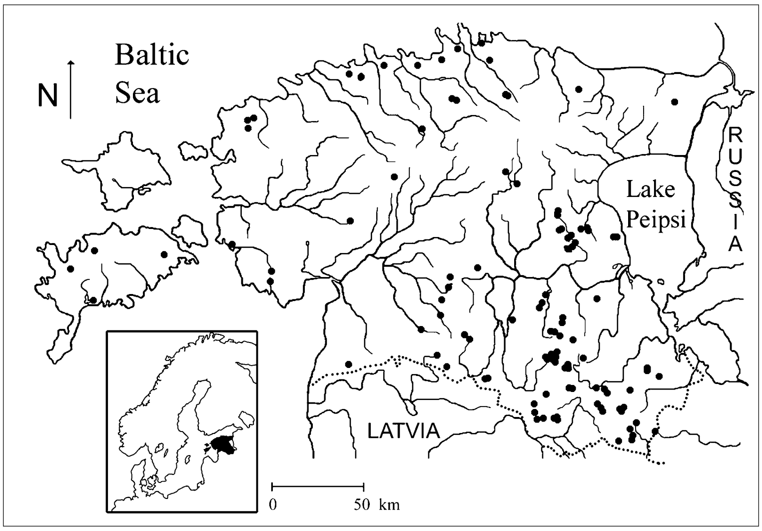

Fig. 1. Study area. Each dot marks a small lake compared between two periods.

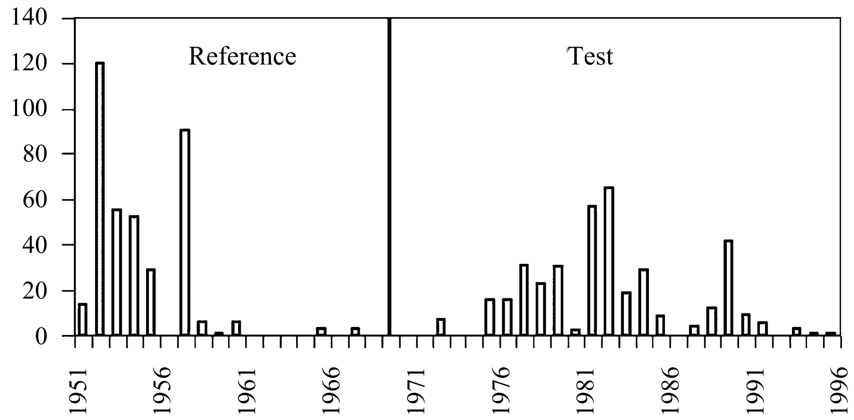

Fig. 2. The number of observations in separate years.

belong either to the eutrophic or to the mixotrophic (eutrophic–humic) type (Ott & Kõiv, 1999). The current data set included four lake types: softwater transparent (18 lakes, 120 samples), softwater brown and acid (4 lakes, 22 samples), common hardwater (both brown and transparent; 84 lakes, 606 samples), and former sea lagoons (2 lakes, 12 samples). The mean surface area of the lakes was 109±31 ha. Sampling depth ranged from 0.1 to 37.0 m, with 3.8±0.3 m as an average. The number of samples collected from soft (muddy) bottom was 526, 142 samples were collected from muddy sand, 86 from sand, and 6 samples from stony bottom. Macrovegetation occurred in 232 sampling sites and was absent in 528 ones. Of all sites 394 were in the profundal, 80 in the sublittoral, and 286 in the littoral.

The same sites were sampled repeatedly in summer (June, July, or August) in similar conditions of depth, sediment and macrovegetation in 1951–1967 (reference samples) and in 1972–1995 (test samples). Altogether 66 variables were tested. These were abundance and biomass of 31 animal groups, total abundance and biomass of macrozoobenthos, individual body mass of chironomid larvae, number of higher taxa per composite sample as a measure of diversity. For detecting the temporal changes in the number of animal groups, only 128 samples from 79 lakes were appropriate, because the groups were not always similarly differentiated.

All variables except body mass of chironomid larvae and mean number of taxa were square-root transformed to meet the requirements of normal distribution. The difference between the two study periods was tested by two-way ANOVA as implemented in GLM procedure in the SAS/STAT package (SAS Institute Inc., 1994). The factors used in ANOVA were ‘Site’ (random factor) and ‘Period’ (fixed factor). The corresponding interaction term was also included and treated as a random effect. Due to the presence of random effects, the mixed ANOVA was employed. If the P-value for the factor 'Period' was less than 0.05, differences in abundance, biomass or individual weight between the compared time intervals were considered significant. As the Bonferroni correction was not used, the comparisons might yield incorrect differences for about 5% of the taxonomic groups. To establish if a measured variable has increased or decreased, the least squares means (generated by the LSMEANS option in GLM) were compared.

RESULTS

Estonian lakes represent all three typical benthic zones: the profundal (with muddy bottom, stratified or not), the sublittoral (sandy, muddy, or detritus bottom), and the littoral (with or without vegetation; muddy, sandy, or gravelly bottom). Water depth in these habitats is lake-specific. Only phantom midge (*Chaoborus flavicans*) larvae could be considered typical inhabitants of the stratified profundal. In fishless humic lakes, a related species (*C. obscuripes*) may also occur in the littoral. Chironomids and oligochaetes were not zone-specific, except regarding their low abundance in the stratified profundal. The same applied to valvatids, sphaeriids, and pisidiids, although they were all less common in the profundal zone than chironomids and oligochaetes. Ceratopogonids reached their highest abundance and biomass in the non-stratified profundal, or in the soft-bottom sublittoral. All other animal groups inhabited mainly the littoral.

Among the different limnological types, well-buffered eutrophic lakes in the broadest sense were comparatively rich in macrozoobenthos. In poorly buffered lakes with transparent water, the average biomass of macrozoobenthos was lower. The lowest abundances for macrozoobenthos were registered from soft-water humic and brackish lakes. Differences between the lake types were distinct in the open region of the lakes, while the abundance of benthos was always relatively high among macrophytes.

Chironomidae constituted the most abundant taxonomic group, both in abundance and biomass (Table 1). Oligochaetes occupied the second place in abundance but they were surpassed by *Bithynia*, sphaeriids, and odonates in biomass values. The biomass of megazoobenthos (large molluscs) exceeded that of macrozoobenthos approximately tenfold. The mean number of higher taxa, distinguished without magnification per triple sample (675 cm^2), was 5.5 calculated for all zones in 253 lakes (Table 1).

At 380 sites sampled repeatedly in 1951–1967 and in 1972–1995, a significant increase (from 4.67 ± 0.39 to 6.61 ± 0.55 g m^{-2}) was observed in the mean body mass of chironomid larvae but not in their abundance or biomass (Table 2). No significant changes occurred in the abundance or biomass of total macrozoobenthos consisting largely of chironomids. However, a significant increase ($P < 0.05$) was observed both in the abundance and biomass of *Chaoborus*, Pisidiidae, and *Bithynia*, in the abundance of Ceratopogonidae, Odonata, Lepidoptera, and in the biomass of Oligochaeta, Mermithidae, and *Valvata*. A significant decrease in both parameters was noted for Hydrachnellae while only the abundance for Ephemeroptera, Trichoptera, Mermithidae, and *Unio*, and only the biomass of Heteroptera and Lepidoptera were significantly lower in the second period. The number of taxa did not reveal any difference between the two periods.

Table 1. The average summer characteristics of zoobenthos (±SE where possible) for 253 lakes. The number of observations was 1566 for the number of groups per sample, 1619 for the average individual weight of Chironomidae, and 1796 for the other measures

Animal group	ind. m^{-2}	g m^{-2}, wet weight
Macrozoobenthos		
Chironomidae	1057 ± 79	2.41 ± 0.16
Oligochaeta	173 ± 10	0.40 ± 0.02
Bithynia	8 ± 1	0.55 ± 0.06
Sphaeriidae	7 ± 1	0.45 ± 0.12
Lymnaeidae	2	0.15 ± 0.04
Pisidiidae	28 ± 4	0.13 ± 0.01
Planorbiidae	3 ± 1	0.12 ± 0.02
Valvata	8 ± 2	0.07 ± 0.01
Odonata	6 ± 1	0.42 ± 0.05
Hirudinea	23 ± 2	0.35 ± 0.04
Trichoptera	21 ± 2	0.32 ± 0.03
Asellus	98 ± 9	0.31 ± 0.03
Chaoborus	65 ± 7	0.25 ± 0.03
Hydrachnellae	35 ± 2	0.02
Total	1658 ± 89	6.45 ± 0.30
Megazoobenthos		
Anodonta	1	23 ± 5
Unio	1	15 ± 4
Dreissena	44 ± 16	9 ± 2
Viviparus	2 ± 1	5 ± 2

Average individual weight of Chironomidae: 5.1 ± 0.4 mg.
Average number of macro- and megazoobenthic taxa per sample (675 cm^2): 5.5 ± 0.2.

Table 2. Comparison of the zoobenthos characteristics for 107 lakes in the periods 1951–1967 and 1972–1995. *P*-values correspond to the factor 'Period', its low value (less than 0.05, in bold) means a significant difference between the two periods. The sign < indicates increase and > indicates decrease

Group	*P*-values of changes in abundance		*P*-values of changes in biomass	
Macrozoobenthos				
Chironomidae	0.3287	>	0.4286	>
Ceratopogonidae	**0.0357**	<	0.0514	<
Chaoborus	**0.017**	<	**0.0011**	<
Ephemeroptera	**0.0001**	>	0.2498	>
Trichoptera	**0.0272**	>	0.8954	>
Odonata	**0.0137**	<	0.5458	>
Sialis	0.06	>	0.974	<
Coleoptera	0.5219	<	0.7773	>
Heteroptera	0.118	>	**0.0016**	>
Lepidoptera	**0.0213**	<	**0.038**	>
Hydrachnellae	**0.0024**	>	**0.0026**	>
Asellus	0.825	>	0.8322	<
Gammaridae	0.4825	<	0.2401	<
Sphaeriidae	0.0545	<	0.2188	<
Pisidiidae	**0.0373**	<	**0.005**	<
Bithynia	**0.004**	<	**0.0005**	<
Valvata	0.0784	<	**0.0336**	<
Lymnaeidae	0.5482	<	0.5191	>
Oligochaeta	0.2395	<	**0.0019**	<
Hirudinea	0.071	<	0.4985	<
Mermithidae	**0.0365**	>	**0.0178**	<
Total	0.8007	>	0.7225	<
Megazoobenthos				
Anodonta	0.7541	>	0.5927	<
Unio	**0.0482**	>	0.3196	<
Dreissena	0.4036	<	0.2993	<
Viviparus	0.5941	>	0.6197	>

Average individual weight of Chironomidae: **0.0043** <.
Average number of macro- and megazoobenthic taxa per sample (675 cm^2): 0.6793 <.

DISCUSSION

The observed changes in macrozoobenthos can well be related to increased eutrophication. The majority of the studied lakes belonged to the common hard-water type, therefore the results should mirror best the changes in this type of lakes.

The significant increase both in the abundance and biomass of *Chaoborus* may indicate decreased oxygen concentrations in the hypolimnion, as well as an expansion of stratified areas to the shallower regions. The increase in small, mostly littoral molluscs (*Bithynia*, *Pisidium*, *Valvata*) can be explained by improved feeding conditions and/or alkalinization of many soft-water lakes (Ott & Kõiv, 1999). The *Valvata* species showed also a marked increase in Lake Peipsi, a large lake between Estonia and Russia, during 1960–1990 (Timm et al., 1996). The significant decline in Ephemeroptera and Hydrachnellae in small lakes coincided with a similar trend in the open-water areas of Lake Peipsi. Ephemeroptera larvae are well-known by their general intolerance of harmful effects (Roback, 1974; Wiederholm, 1984). In open-water areas of Estonian lakes, they are represented mainly by the genera *Caenis*, *Cloeon*, and *Ephemera*. Water mites, on the other hand, are often not considered a group with a high pollution sensitivity. However, the affinity of some species to clean water has been documented in streams (Kowalik & Biesiadka, 1981) and from natural hydrological conditions (Smit & van der Hammen, 1992). Hence, the obvious response of lake-dwelling water mites to eutrophication in Estonia deserves further research.

The increase in Ceratopogonidae, as well as the decrease in Trichoptera and Unionidae abundance, could be explained by increased sedimentation and oxygen deficiency caused by eutrophication. Gerritsen et al. (1998) considered odonates an intolerant group. In our study lakes, the abundance of Odonata, paradoxically, seemed to be favoured by eutrophication. Perhaps it could be explained by the increased abundance of macrophytes in many Estonian lakes during 1950–1980 (Ott & Kõiv, 1999). Mermithidae, the large nematode parasites of chironomid larvae and other aquatic insects, increased in biomass and decreased in abundance after eutrophication. It cannot be excluded that there occurred an uncontrolled introduction of tiny, non-mermithid nematodes in the samples in the reference period. The abundance of Lepidoptera, on the contrary, increased, while their biomass decreased after eutrophication. As both groups were quite rare, such changes can also be occasional.

Chironomids, which constituted the largest part of the total macrozoobenthos, did not reveal any direct response in abundance or biomass to changes in the eutrophication level. Like the total abundance and total biomass of macrozoobenthos, the total abundance and total biomass of chironomids can be considered a relatively poor indicator of moderate eutrophication in small lakes.

However, in the second largest Estonian lake, Võrtsjärv, a general trend towards increase in the biomass of chironomids and oligochaetes has been observed since 1987 (Kangur et al., 1998). At the same time, the biomass of all other macrozoobenthic groups decreased significantly. In small lakes, on the contrary, the body size of chironomids increased significantly after eutrophication. In Lake Võrtsjärv, a similar trend was caused by the increase in the share of large *Chironomus plumosus* larvae in macrozoobenthos (Kangur et al., 1998). Increase in the body size of chironomids due to eutrophication is well consistent with the traditional notion that the profundal of oligotrophic lakes is inhabited by smaller chironomid larvae (for example, Tanytarsini) than the profundal of eutrophic lakes (Chironomini) (Jónasson, 1969). In Lake Esrom (Denmark), eutrophication through the 20th century increased oxygen depletion in the profundal zone. The increased environmental stress resulted in the replacement of chironomids by oligochaetes (Lindegaard et al., 1997).

Although the influence of climate and biotic factors (for example, benthophagous fishes) on macrozoobenthos was not studied, anthropogenic eutrophication was considered to have a negative influence on several intolerant groups. At the same time, no significant changes were observed in biodiversity (the number of taxa distinguished without magnification per single sample remained similar). However, the intolerant groups of macrozoobenthos were largely substituted by tolerant groups.

Benthic invertebrates belong to quality elements for the classification of ecological status in lakes (Water Framework Directive, 2002). For the shallow littoral of Estonian lakes, the preliminary reference levels for some indices based on macroinvertebrates were already estimated earlier by Timm (2003). Unfortunately, most of these indices are not applicable in deeper parts of lakes. The indices proposed for offshore areas of lakes in Sweden (Johnson, 1999) are seldom applicable in Estonian lakes that are often shallow, without hard bottom even in the littoral. Hence, the current results are useable for preliminary judgement which variables are sensitive to eutrophication in open-water areas of small lakes in Estonia. The next purpose is to estimate how the potential indicators are related to lake morphometry, limnological type, sampling depth, stratification, bottom type, and presence and abundance of macrophytes.

CONCLUSIONS

Long-term eutrophication caused by intensive fertilization during the 1970s and the 1980s was probably responsible for changes in the macrozoobenthos of small Estonian lakes.

Eutrophication caused significant changes ($P < 0.05$) in the values of several variables (e.g. an increase in the body size of chironomid larvae). The significant decrease in Ephemeroptera and Hydrachnellae, as well as an increase of *Valvata* snails in small lakes, coincided with similar phenomena in the open-water area of the largest Estonian lake, Peipsi.

Biodiversity, expressed as the number of animal groups per sample established without magnification, did not reveal differences before and after impairment.

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Pikaajalise eutrofeerumise mõju Eesti väikejärvede makrozoobentose arvukusele ja biomassile

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1. Väikejärvede pikaajaline eutrofeerumine, mis tulenes põlluväetiste ohtrast kasutamisest Eestis 1970. ja 1980. aastatel, on nähtavasti põhjustanud muutusi ka põhjaloomastikus (makrozoobentoses).
2. Eutrofeerumine on statistiliselt oluliselt ($P < 0{,}05$) mõjutanud paljude tunnuste väärtusi, sealhulgas suurendanud surusääsklaste vastsete individuaalset kaalu. Ühepäevikuliste (Ephemeroptera) ja vesilestade (Hydrocarina) hulga kahanemine ning sulgtigude (*Valvata*) hulga suurenemine väikejärvedes langeb kokku samasuguste nähtustega Peipsi järves.
3. Taksonirikkus, mida väljendas palja silmaga määratav loomarühmade arv ühe proovi kohta, ei erinenud järvedes enne ja pärast mõjutamist.