Proc. Estonian Acad. Sci. Biol. Ecol., 1998, **47**, 2, 98–113
https://doi.org/10.3176/biol.ecol.1998.2.02

THE EFFECT OF FLUCTUATING WATER LEVEL ON THE ECOSYSTEM OF LAKE VÕRTSJÄRV, CENTRAL ESTONIA

Peeter NÕGES and Tiina NÕGES

Võrtsjärv Limnological Station, Institute of Zoology and Botany at the Estonian Agricultural University, EE-2454 Rannu, Tartumaa, Estonia; e-mail: tiina@lim.tartu.ee; and Institute of Zoology and Hydrobiology, University of Tartu, EE-2400, Vanemuise 46, Tartu, Estonia

Received 4 December 1997, in revised form 18 February 1998

Abstract. Lake Võrtsjärv (270 km^2) in Central Estonia is the second largest lake in the Baltic countries. It is a shallow and turbid eutrophic lake with a mean depth of 2.7 m and maximum depth of 6 m. The amplitude of water level (WL) fluctuations is large: the annual mean is 1.38 m, annual maximum 2.20 m, and the absolute range 3.20 m. Besides seasonal fluctuations, the annual mean WL of dry and rainy years differs by more than one metre. Long-term WL measurements (starting from 1885, daily measurements from November 1921) show a periodic alternation of low and high water states within about 30-year periods. Due to the shallowness of the lake, low-level periods are accompanied by several unfavourable phenomena like cyanophyte blooms, overgrowing by macrophytes, restricted spawning area for pike, and winter fish kills. Despite a higher external nutrient load in precipitation rich years, the rising WL has a positive effect on the ecological state of the lake. A decrease in phytoplankton and bacterioplankton biomass and in suspended solids concentration, resembling the effect of a reduction of the trophic status, is related to the strengthening of the light limitation effect on phytoplankton and a decline in sediment resuspension.

Key words: water level, shallow lake, light limitation, Secchi depth, phytoplankton biomass, bacterioplankton, long-term investigations, sediment resuspension, water level regulation.

INTRODUCTION

Every time when the water level (WL) in shallow Lake Võrtsjärv has fallen drastically during dry years, the question about regulation possibilities has arisen. The previous disputes, which started in the early 1970s (Eipre, 1971; Mäemets, 1972), resulted in a realistic regulation scheme. Considering the large drainage area, poor outflow conditions, and flat shores, Võrtsjärv represents a complicated case for WL regulation. Outflow calculations (Kaljumäe & Koskor, 1980) showed that it would be reasonable to keep the minimum level close to the long-term mean level of the lake (33.7 m). The regulated maximum WL ought to be 34.8 m, in which case the forced maximum would reach 35.6 m. As a result, the WL dynamics would generally follow the natural pattern at a higher level and with a smaller amplitude.

In the new situation brought about by the changes in land ownership, environmental protection, and economy, a revaluation of the potential consequences of WL regulation to the lake ecology, fishery, water economy, agriculture, recreation, and water transport is urgently needed. In the present paper we show the response of biota to changes in WL, and figure out some ecological criteria for selecting the optimum scenario of WL regulation. Optical properties of water as well as the mixing depth and resuspension rate of sediments, all determined by the WL, come to the forefront in the case of shallow and turbid Võrtsjärv where light limitation plays a major role in controlling phytoplankton growth (Nõges, 1995; Nõges et al., 1997). Despite the rather stable nutrient concentration, the biological indices of the trophic state fluctuate strongly. Changing WL affects various components of biota either directly or through cascading effects in the food chain.

DESCRIPTION OF THE LAKE WITH AN EMPHASIS ON HYDROLOGY

Võrtsjärv (Fig. 1) is a shallow lake in Central Estonia at 58°05'–58°25' N, and 25°55'–26°10' E. Its area is 270 km^2, mean depth 2.7 m, and maximum depth 6 m. Eighteen rivers and streams collect their water from the 3104 km^2 mostly intensively cultivated drainage basin. The lake is highly eutrophic, characterized by mean concentrations of total N (TN) about 2 mg l^{-1} and total P (TP) about 50 µg l^{-1}. The water is alkaline (pH 7.5–8.5) with a great buffering capacity and a high seston content. During the ice-free period, transparency does not exceed 1 m. The outflow of Võrtsjärv, the Suur Emajõgi River, has a small slope, only 0.038‰. The outflow conditions of the lake depend on the level of the Pede River, the greatest tributary of the Suur Emajõgi. The opposite flow in the upper course of the Suur Emajõgi occurs almost every spring, while during about half a month the lake has no outflow (Järvet, 1995).

The lake depression is of preglacial origin. About 2/3 of the lake bottom is covered with mud lying on marl of a total thickness of up to 7.6 m (Veber, 1973). Sediment accumulation is intensive and the lake has become increasingly shallower from year to year. Short-term increases in the WL, observed in the middle of the last century and at the beginning of this century (Sievers, 1854; Mühlen, 1919), intensified shore abrasion and transport of eroded material towards the outflow promoting its further clogging (Orviku, 1973).

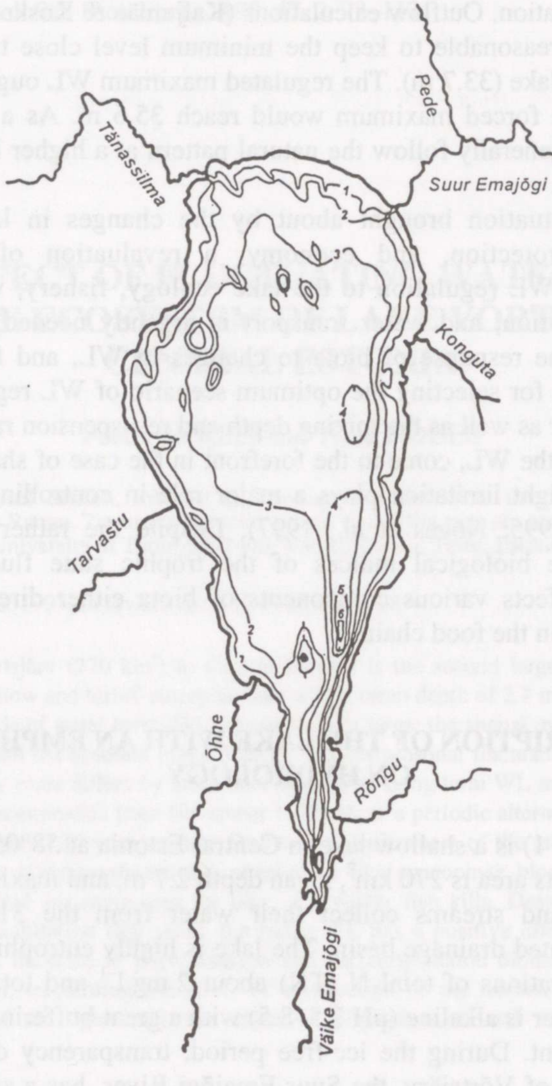

Fig. 1. Lake Võrtsjärv and connected rivers. 1–6, depth at the mean water level.

Increase in the water budget of Võrtsjärv is mostly accounted for by inflow (73–83%) and decrease by outflow (80–91%) (Jaani, 1990). The renewal of water takes from 240 to 384 days. Due to rather flat shores and small depth, changes in the WL are expressed in the mobile shoreline and in large relative differences in volume (Table 1). The long-term mean level of Võrtsjärv, computed over a 74-year period (1922–96) is 33.63 m according to the Baltic Level System. The absolute long-term range of level fluctuations is 320 cm, which corresponds to a 93 km^2 difference in the lake area and to a 0.874 km^3

Table 1

Changes in the area, volume, and mean depth of L. Võrtsjärv depending on the water level (areas taken from Jaani (1973), volume and mean depth recalculated)

Water level, m	Area, km^2	Volume, 10^6 m^3	Mean depth, m
35.28 (abs. maximum)	327	1211	3.70
35.00	309	1121	3.63
34.50	286	973	3.40
34.00	276	832	3.02
33.63 (average)	269	731	2.72
33.50	267	697	2.61
33.00	259	565	2.18
32.50	248	438	1.77
32.08 (abs. minimum)	234	337	1.44

difference in its volume. In the recorded range, the relation between the WL and the mean depth of the lake (z_{av}) can be described by a linear approximation:

$$z_{av} = 0.741 \times \mathrm{WL} - 22.26 \qquad (R^2 = 0.989; p < 0.01). \qquad (1)$$

The absolute range of the mean depth of the lake extended from 1.44 (6 Sep 1996) to 3.70 m (26 Nov 1923).

Intensive and prolonged high water in spring, low water in summer and winter, and a noticeable rise in the WL in autumn are characteristic seasonal features of the lake (Fig. 2). The annual amplitude of fluctuation in the WL has been 1.38 m on an average, ranging from 0.76 to 2.20 m in different years (Jaani, 1990). In 78% of years the highest WL has occurred in spring, and in 22% of years in late autumn.

A long-term sinusoidal fluctuation of the WL with a period of about 30 years is best revealed by using a 7-year moving average (Fig. 3). Water levels for the period 1885–1921 were calculated by Jaani (1990) on the basis of data on the Suur Emajõgi. A rather smooth continuous decrease (1928–40) or increase (1940–57; 1965–90) within the period resembles a long-term trend and can be distinguished from it only in the context of a longer time series. Apparent periodicity is probably associated with large-scale fluctuations in solar activity (Jaani, 1973) and atmospheric processes, since it appears in the same way in large geographic regions. Similar periodicities with a spectral density maximum of 28–32 years were found in the WL dynamics of lakes Saimaa, Ilmen, and Onega (Masanova & Filatova, 1985), as well as for different hydrological elements of Lake Ladoga (20–30 years) (Malinina et al., 1985) and Lake Müggelsee (Behrendt & Stellmacher, 1987). The last increase in the WL can be firmly

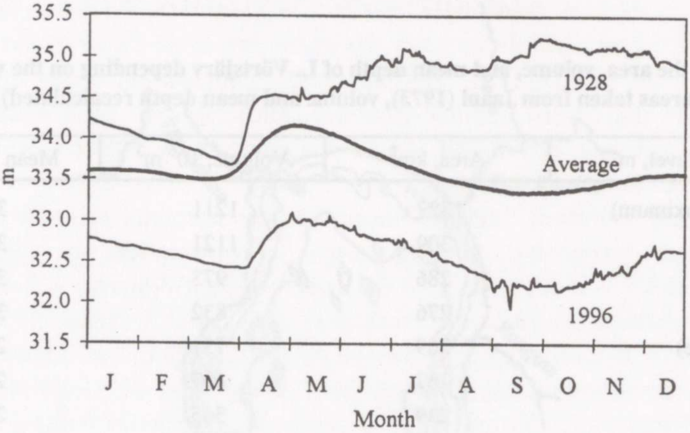

Fig. 2. Mean seasonal dynamics of the absolute water level in Lake Võrtsjärv in 1922–96 and the dynamics in two extreme years.

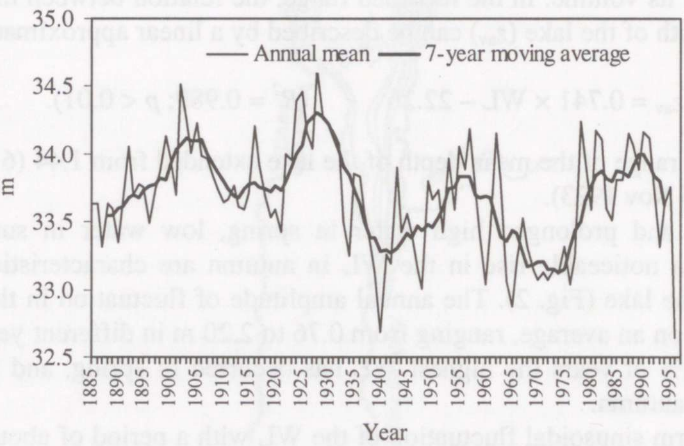

Fig. 3. Long-term changes in the absolute water level in Lake Võrtsjärv (after Jaani, 1990, supplemented with latest data).

attributed to the 25% increase in the annual amount of precipitation in Central Estonia (from 526 to 652 mm) between 1955 and 1986 (Kivi & Russak, 1990).

A linear trend for the period 1964–93 revealed an increase in the mean WL by 1.5 m and the corresponding increase in the mean depth of the lake by 1.1 m (Nõges et al., 1997). A new decrease in the WL was noticeable in 1992–93. In September 1996 the record lowest WL was registered in the lake.

MATERIAL AND METHODS

The data set used in the present paper consists of water transparency measurements and hydrochemical and plankton analyses made at the Võrtsjärv Limnological Station in 1964–96. In 1997 a new survey of aquatic macrophytes was made. Data on daily WL at the outflow for the years 1922–96 were obtained from Tartu Hydrological Station. The other measurements and analyses pertain to the surface layer of the main monitoring station. Routine sampling was performed once a month, in some periods more frequently, 2–4 times per month. Underwater light measurements with a 4-π PhAR collector (Williams & Jenkinson, 1980) were made in 1989 and 1995. Bottom irradiance (I_B) was calculated according to Lambert-Beer's law:

$$I_B = I_0 \exp(-kz), \tag{2}$$

where irradiance in the surface layer (I_0) was taken equal to 100% as its changes in a 30-year period could be neglected. The mean depth of the lake was denoted by z and the attenuation coefficient (k) was replaced by its relation to Secchi depth(s) (Nõges et al., 1997):

$$k = (1.48 \pm 0.05) / S \qquad (\pm \text{ standard error}). \tag{3}$$

The average irradiance in the mixed water column was calculated according to Riley (1957):

$$I_{mix} = I_0[1 - \exp(-kz)] / kz \tag{4}$$

assuming that in Võrtsjärv the mixing depth increases simultaneously with the mean depth of the lake z. The proportion of the euphotic bottom area was calculated on the basis of the bathymetric curve by Kongo (1973) and the penetration depth of 1% of the photosynthetically active radiation (PhAR) at the water surface, commonly used to assess the lower boundary of the euphotic zone (Tilzer, 1987).

The cases of winter fish kills were analysed statistically. The significance of the WL difference from the mean in the years of fish kills was evaluated using the z-criterion (Jalasto, 1978):

$$z = \frac{(x - y)\sqrt{n}}{\sigma}, \tag{5}$$

where x the mean WL for 1922–96;
y the mean WL for the years of fish kills;
n the number of fish kill years;
σ standard deviation of the WL for 1922–96.

RESULTS AND DISCUSSION

Phytoplankton

The phytoplankton of Võrtsjärv is abundant and rich in species. Owing to shallow water and continuous wind-induced mixing the number of benthic and periphytic forms is big. At the end of the 1970s, several changes took place in the species composition: it became generally poorer, while a new species of cyanophytes, *Limnothrix redekei* (Van Goor) Meffert, appeared among dominants. Nowadays the algal community is represented by an association of filamentous algae, the species of the genus *Aulacoseira* in spring, *L. redekei*, *L. planctonica* (Wolosz.) Meffert, *Planktolyngbya limnetica* (Lemm.) Kom. & Cronb., and *Aphanizomenon skujae* Kom.-Legn. & Cronb. in summer and autumn, accompanied by a low variable biomass of small algae, mostly chlorophytes and chrysophytes. In 1996, the year of extremely low WL, the filamentous species were temporarily replaced by fast growing chroococcal blue-green algae from the genera *Cyanonephron*, *Cyanodictyon*, and *Microcystis.*

Changes in phytoplankton biomass were diametrical to changes in WL (Fig. 4). During the previous low-water period in the early 1970s, phytoplankton biomass maxima reached 100 g m^{-3} in a few cases. In spite of heavy nutrient loading in the middle of the 1980s (Järvet, 1997), phytoplankton biomass decreased and seldom exceeded 30 g m^{-3}. The reduced use of fertilizers together with the smaller runoff resulted in a continuous decrease in nutrient loading in the 1990s, but the standing stock of phytoplankton increased again, reaching a peak value of 72 g m^{-3} in 1996.

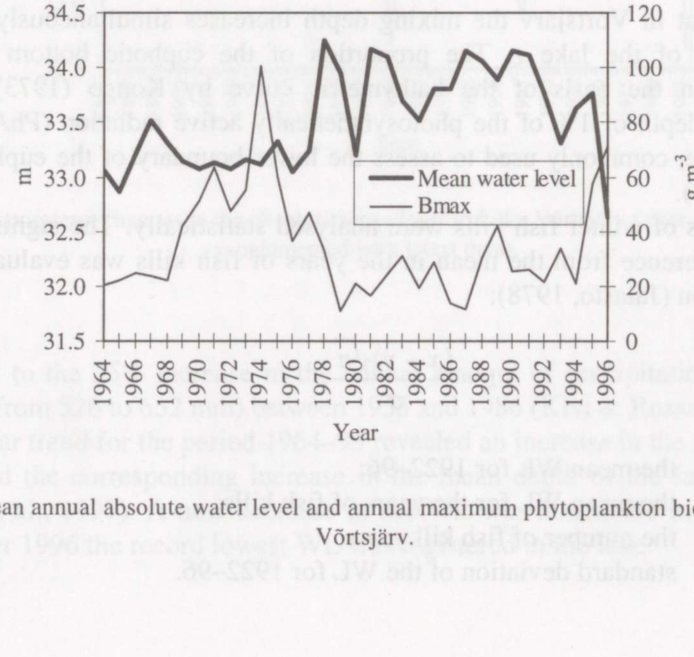

Fig. 4. Mean annual absolute water level and annual maximum phytoplankton biomass in Lake Võrtsjärv.

The inverse correlation between phytoplankton biomass maxima and the mean WL, as well as the succession of species, can be explained by changes both in light and nutrient conditions accompanying the fluctuating WL. In natural water-bodies light limitation is a universal phenomenon, as light-limited phytoplankton is always found below a certain depth or in the dark season of the year (Sakshaug, 1980). In shallow turbid lakes where the mixing depth is determined by the depth of the lake, light conditions are strongly affected by seasonal and long-term changes in the WL. This is a dual effect since not only the mean depth (z) but also the optical properties of water are affected by the WL. During years of increasing WL, water transparency improved, especially in autumn (September–November) at the time of the annual maximum of phytoplankton biomass and detritus content. Autumn Secchi depth increased from 0.6 m in the 1970s to 0.9 m in the early the 1990s. Despite improved water transparency, the calculated water column irradiance (Eq. 4) dropped by about 1/3 during the period 1964–93. Although most of the water column remained within the euphotic zone, deterioration of light conditions resulted in a decrease of primary production, which was reflected in reduced phytoplankton biomass and pH values. By vertical mixing phytoplankton is transported through a light gradient and adapted to an average light intensity in the mixed layer, which Riley (1957) defined as "effective light climate". In Võrtsjärv this mean light intensity was rather low during high water periods. A similar process was observed in Lake Beloe (Rumyantsev et al., 1985) where a 1.5 m increase in the WL had a much stronger influence on the underwater light field than an increase in trophic parameters like mineral phosphorus and nitrates.

Dim light in the water column favoured the development of *L. redekei* in the 1980s. Nicklisch et al. (1981) demonstrated a specific light adaptation of this species, which allows its mass development, compared with other blue-greens, in more turbid and cool water. The replacement of filaments by colonial forms of cyanophytes requiring more light was a response to the drop of the WL in 1996, as the extremely shallow water (mean depth 1.44 m in September) was well illuminated.

Nutrient enrichment caused by sediment resuspension, in its turn, favours phytoplankton growth at low WLs, as not only particulate matter, but also dissolved matter from the interstitial water, is carried back to the water phase. This enables a repetitive use of nutrients for phytoplankton (Galicka, 1992) and makes the system rather independent of the external loading. In Võrtsjärv sediment resuspension enriches water first of all with phosphorus as the TN : TP ratio in the sediment (median 12) is nearly three times lower than in the water (median 35), while the availability of nitrogen can become even worse. As shown by Jensen et al. (1990) for 58 shallow Danish lakes, sediment disturbance may intensify denitrification by bringing nitrate-rich lake water into contact with anoxic sediment layers, which is the important site for denitrification. Probably, the more acute N deficiency has favoured the development of N_2-fixing species (*Aphanizomenon skujae*, *Anabaena* spp.) in recent years.

Bacterioplankton

Bacterial counts in Võrtsjärv have ranged from 1.2 to 14.0×10^6 cells ml^{-1} with a mean value of 4.2×10^6. Their maxima occurred in the first half of the 1970s just as those of phytoplankton biomass, and a similar relation could be observed between the total number of free-living bacteria and monthly mean WLs (Fig. 5).

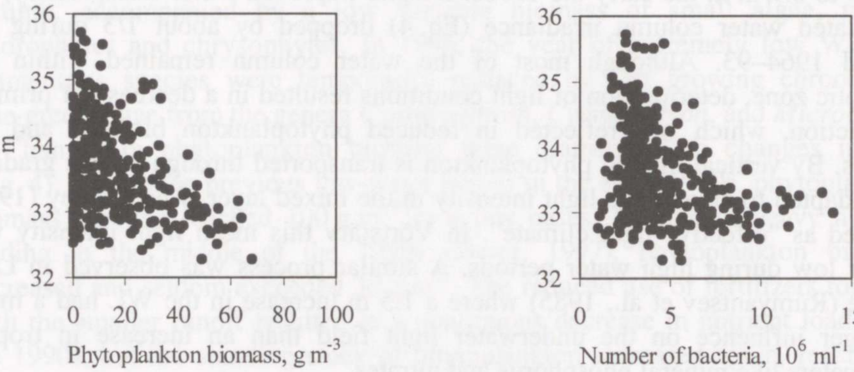

Fig. 5. Scatter plot of monthly mean plankton abundance versus absolute water level in Lake Võrtsjärv.

Three mechanisms may be responsible for a simultaneous decrease in phyto- and bacterioplankton abundance: (1) limitation of bacterioplankton growth by algal substrate under conditions of decreasing primary production; (2) limitation of bacterial numbers by the smaller amount of resuspended sediments in the case of a rising WL; (3) increase in zooplankton grazing pressure on bacteria under the conditions of the growing proportion of filamentous blue-green algae in phytoplankton. The first two mechanisms can be joined under a common denominator “dilution effect”. On the basis of available data it is hard to determine the role of each of these mechanisms, though the leading role of the WL is obvious. Fondén (1969) found a similar negative correlation between bacterial density and the depth of the water column in the large Swedish lakes of Mälaren and Hjälmaren, and associated it with the vertical dilution effect of the deeper water column. During summer, bacterioplankton was influenced by both filter-feeding zooplankton and the amount of available organic carbon. There exists a close link between pelagic and benthic processes in large shallow lakes. In Loosdrecht Lakes resuspended particulate organic carbon exceeds 15–20 times carbon fixation by phytoplankton on windy days (Gons et al., 1986). Wainwright (1987) and Ritzrau & Graf (1992) showed that resuspension does not only passively increase bacterial abundance but also stimulates bacterial growth.

Aquatic macrophytes

Wind-exposed northern and eastern coasts of the lake are bordered with a mostly continuous belt of *Phragmites australis* (Cav.) Trin. ex Steud., followed by submerged *Potamogeton perfoliatus* L. at a depth of about 2 m. The reed-beds become narrow or fragmentary only in places exposed to the strongest wave action, often separated from the shore by a shallow open water area. The western coast is richer in vegetation. The wide zone of *P. australis* and *Schoenoplectus lacustris* (L.) Palla, starting right on the shore, has dense vegetation. The zone of floating-leaved plants (*Nuphar luteum* (L.) Smith) is distinct in places, while the zone of submerged vegetation is well formed everywhere. The southern corner of the lake is fully overgrown with macrophytes. All types of aquatic macrophytes are abundantly represented, and often become merged. This description by Mäemets (1973) rendered briefly here, is still valid, though the area and density of macrophyte stands has increased.

Depending on fluctuations in the WL, the abundance and distribution boundaries of different species vary greatly. Besides eutrophication, also lasting low-level periods in dry years contribute to the broadening of reed-bed areas. The formation of macrophyte stands along the shoreline northward of the mouth of the Tänassilma River has been associated with the extremely low WL in 1939–40, which enabled emergent macrophytes (*P. australis*, *S. lacustris*) and submerged vegetation to fix and extend to the previously sandy areas (Pihu, 1959). In order to keep the open water area free of vegetation, main attention has to be paid on submerged macrophytes. In this context, the narrow southern part of the lake can be regarded as the "hot spot" of the lake. In recent years of low WL, *Myriophyllum spicatum* L. has become a nuisance on large areas south from Tondisaar, tending to overgrow the whole open water area. It complicates motor boat traffic and clogs the fishing nets. It colonizes the bottom in a depth range from 1.5 to 3.2 m at mean WL. Together with *P. lucens*, it has the largest colonization depth among aquatic macrophytes in the lake. *M. spicatum* prefers muddy bottom and does not tolerate strong wave action. Probably the latter limits its spreading to the north.

Light conditions at the bottom limit the colonization depth of submerged macrophytes. The 1% surface PhAR criterion is commonly used to assess the lower boundary of the euphotic zone (Tilzer, 1987). In Võrtsjärv the proportion of the bottom area left outside the euphotic zone (oligophotic area), varies from

30 to 90%, depending on the mean depth and water transparency (Fig. 6). The "realistic range" marked by the broken lines delimits the most probable combinations of mean depth and water transparency. All the curves have a maximum at the mean depth of 3.4 m (34.5 m absolute level). At higher WL, large plain areas are flooded and the total area of the lake increases faster than the oligophotic area. Thus, the raising of the WL could effectively limit the distribution area of submerged macrophytes and prevent the overgrowing. In order to maximize the areal proportion of the lake unsuitable for plant growth, a 0.8 m increase of the WL (a 0.6 m increase of the mean depth) would be recommended.

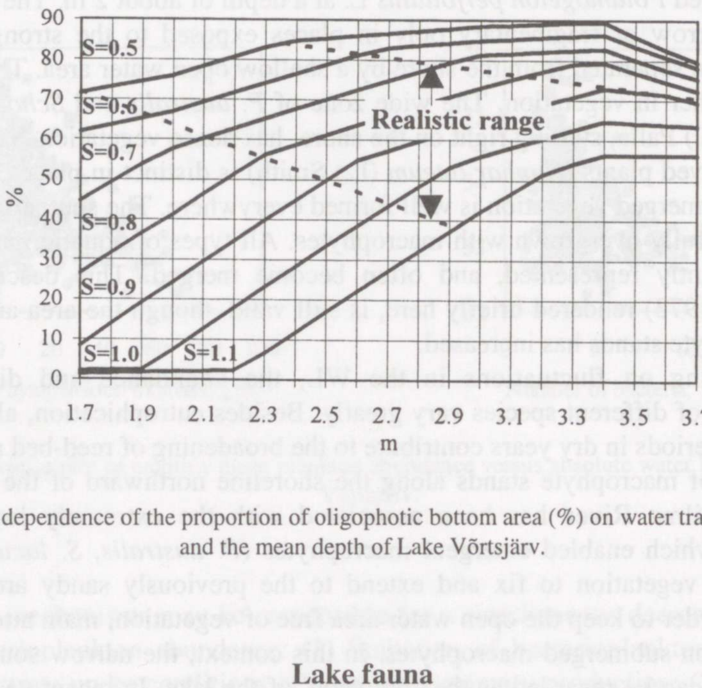

Fig. 6. The dependence of the proportion of oligophotic bottom area (%) on water transparency (S) and the mean depth of Lake Võrtsjärv.

Lake fauna

The zooplankton of Võrtsjärv is dominated by small cladocerans, mainly *Chydorus sphaericus* (Müller), and rotifers specialized in feeding in patches between filamentous algae. Recent studies made in 1995 and 1996 revealed an important role of protozoa in the lake, among which ciliates have an almost equal biomass (~1 g m^{-3}) with the crustacean and rotifer metazooplankton (Haberman, 1998; Zingel, submitted). The high detritus content of water, resulting from continuous mixing and wave action, has been considered the main factor determining the small biomass and mean body size of filtering zooplankters, as it clogs their filtering apparatus (Haberman, 1984). Revival of the ecosystem through the increasing WL at the beginning of the 1980s was reflected in a decrease of rotifer biomass (Haberman, 1983). Although several changes have occurred in the species composition, no significant trends were observed in the biomass of crustaceans and rotifers in a 30-year time series (Nõges et al., 1997).

Among the bottom fauna of Võrtsjärv Chironomidae dominate in all zones of the lake. In 1973–78 big larvae of *Chironomus plumosus* L. formed on the average 60.8% of the macrozoobenthos (MZB) biomass (Kangur, 1982). During the rising phase of the WL in 1970–91, a general tendency towards increasing abundance and biomass of Chironomidae and Oligochaeta, as well as decreasing abundance of Mollusca, was noticeable (Kangur et al., 1998). As annual differences in the MZB amount are strongly influenced by changes in Chironomidae, whose abundance depends on meteorological conditions during the reproduction period and on fish pressure (Kangur, 1982), no significant correlations occurred between MZB and WL. As a rule, the littoral area has been the richest in species diversity, as well as in the abundance and biomass. Only *Chironomus plumosus* achieved its highest biomass in the muddy profundal area (Kangur et al., 1998). In dry years the shoreline recedes in places up to a kilometre in the second half of summer, exposing large littoral areas, where the benthic fauna cannot survive. A more stable water regime would retain these areas of high bottom fauna production available for the feeding of the basically benthivorous fish community.

Bream (*Abramis brama* (L.)), eel (*Anguilla anguilla* (L.)), pikeperch (*Stizostedion lucioperca* (L.)) and pike (*Esox lucius* L.) are the main commercial fishes in Võrtsjärv, while perch (*Perca fluviatilis* L.), roach (*Rutilus rutilus* (L.)), ruffe (*Acerina cernua* (L.)) and burbot (*Lota lota* (L.)) are of second-rate economic importance (Pihu, 1998). All the eel production of Võrtsjärv is based on stocking with wild caught elvers (38 million during 1956–96) (Kangur, 1998).

Several winter fish kills (in 1939, 1948, 1967, 1969, 1978, 1987) have been documented during this century (Kirsipuu et al., 1987). The authors showed the fish kill in 1987 to have been caused by anthropogenic factors. An analysis of the rest of the cases made by us revealed a significantly lower WL in the years of fish kills, especially in winter months (Table 2). The winter of 1995/96 is another example of the disastrous consequences of a low WL. After the highly productive summer of 1995, the lake was frozen at an extremely low level. The winter was cold and the lake was covered with thick ice (~0.6 m) and snow (~0.3 m). There were no thaws from December till March and the lowest oxygen concentration during the studied 30 years was registered. At the beginning of March 1996, there was still some oxygen left (2.3 mg l^{-1} just below the ice, 0.4 mg l^{-1} in the bottom layer). Anoxia extended to the whole water column in most parts of the lake to the middle of March. Probably, most of the pelagic fishes could find some refuge near the river mouths where oxygen conditions were better, but the oxygen depletion caused a large kill of eel (10–20 t according to different estimates), which digs itself into mud for overwintering.

Table 2

Differences in water level from the mean value in the years of winter fish kills in Lake Võrtsjärv (1939, 1948, 1967, 1969, 1978, 1996)

Period	Difference from the mean, cm	Significance
Winter minimum	–44	0.06
Spring maximum	–39	0.04
January	–50	0.04
February	–52	0.03
March	–50	0.04
April	–42	0.06
May	–36	0.06
June	–34	0.06
Annual	–36	0.05

The flooding of lakeside meadows starts at the WL of 34.47 m (Pihu, 1959). In dry springs the flood plain remains dry, and the spawning area of pike is restricted. The spawning of pike in Võrtsjärv lasts 2–3 weeks (Haberman et al., 1973) and, in order to guarantee successful spawning, the WL has to exceed the above-mentioned height at least during this time. That requires a height of the flood peak about 34.6 m. Since 1922 the spring WL has reached this height only in 27% of the years. The average of the flood peak in different years is 34.24 m (minimum 33.00, maximum 35.24). The raising of the WL by 1 m in dry springs would guarantee favourable spawning conditions for pike in 88% of years. Strong winds coinciding with low water in May endanger the spawn of pikeperch, which can be buried under sediments (A. Kangur, pers. comm.).

CONCLUSIONS

Lasting low-water periods in Võrtsjärv are accompanied by a number of adverse biological phenomena expressed as destabilization of the ecosystem. An increase in phytoplankton and bacterioplankton biomass deteriorates the transparency and gas regime of the lake. Low-level periods accelerate the overgrowing of shallow areas with macrophytes and deteriorate the spawning conditions for pike, restricting its spawning places, and for pikeperch, whose spawn can be buried under sediments. Regulation of the water regime, by raising the minimum and maintaining the optimum level, would improve conditions in the lake. Greatest effect would be achieved due to reduced sediment resuspension and stronger light limitation on phytoplankton, which would control primary production and the amount of particulate matter in the water.

REFERENCES

Behrendt, H. & Stellmacher, R. 1987. Long-term changes in water quality parameters of a shallow eutrophic lake and their relations to meteorologic and hydrologic elements. In *The Influence of Climate Change and Climatic Variability on the Hydrologic Regime and Water Resources (Proceedings of the Vancouver Symposium, August 1987). IAHS Publ.*, **168**, 535–544.

Eipre, T. 1971. Eesti mageveevarud ja veeolude reguleerimine I. *Eesti Loodus,* 9, 549–555.

Fondén, R. 1969. Heterotrophic bacteria in Lake Mälaren and Lake Hjälmaren. *Oikos,* **20**, 344–372.

Galicka, W. 1992. Total nitrogen and phosphorus budgets in the Lowland Sulejów Reservoir for the hydrological years 1985–1988. *Arch. Hydrobiol. / Suppl.*, **90**, 2, 159–169.

Gons, H. J., Veeningen, R. & van Keulen, R. 1986. Effects of wind on a shallow lake ecosystem: Resuspension of particles in the Loosdrecht Lakes. *Hydrobiol. Bull.*, **20**, 1/2, 109–120.

Haberman, H., Kangur, M., Kirsipuu, A., Luts, A., Mikelsaar, N., Pihu, Evi, Pihu, Ervin & Tell, H. 1973. Kalad ja kalandus. In *Võrtsjärv* (Timm, T., ed.). Valgus, Tallinn, 144–194.

Haberman, J. 1983. Comparative analysis of plankton rotifer biomass in large Estonian lakes. *Hydrobiologia,* **104**, 293–296.

Haberman, J. 1984. Võrtsjärve zooplankton aastail 1964–1982. In *Võrtsjärve ökosüsteemi seisund* (Haberman, J., ed.). ENSV Teaduste Akadeemia, Tartu, 65–78.

Haberman, J. 1998. Zooplankton of Lake Võrtsjärv. *Limnologica,* **28**, 1, 49–65.

Jaani, A. 1973. Hüdroloogia. In *Võrtsjärv* (Timm, T., ed.). Valgus, Tallinn, 37–60.

Jaani, A. 1990. Võrtsjärve veerežiim ja -bilanss. *Eesti Loodus,* 11, 743–747.

Jalasto, H. 1978. *Elementaarstatistika käsiraamat.* Valgus, Tallinn.

Jensen, J. P., Kristensen, P. & Jeppesen, E. 1990. Relationships between nitrogen loading and in-lake nitrogen concentrations in shallow Danish lakes. *Verh. Internat. Verein. Limnol.*, **24**, 1, 201–204.

Järvet, A. 1995. Kui Emajõgi voolab tagurpidi. *Eesti Loodus,* 5/6, 144–146.

Järvet, A. 1997. Long-term changes of pollution load of Lake Võrtsjärv. In *Problems of Contemporary Ecology. Temporal Changes in Estonian Nature and Environment* (Frey, T., ed.). Teadusühistu IM SAARE, Tartu, 60–65.

Kaljumäe, H. & Koskor, I. 1980. Võrtsjärve veerežiimi reguleerimise võimalustest. In *Eesti NSV pinnavete kasutamine ja kaitse* (Mäemets, A., ed.). Valgus, Tallinn, 104–114.

Kangur, A. 1998. European eel (*Anguilla anguilla* (L.)) fishery in Lake Võrtsjärv: Current status and stock enhancement measures. *Limnologica,* **28**, 1, 95–101.

Kangur, K. 1982. On the quantitative distribution of zoobenthos, dynamics of the population density and biomass in Lake Võrtsjärv. In *Long-time Observations of Zoobenthos of the Large Lakes of Estonia. Hydrobiological Researches* (Timm, V., ed.), **11**, 110–146.

Kangur, K., Timm, H., Timm, T. & Timm, V. 1998. Long-term changes in the macrozoobenthos of Lake Võrtsjärv. *Limnologica,* **28**, 1, 75–83.

Kirsipuu, A., Tiidor, R. & Järvet, A. 1987. Kõige eest tuleb maksta. *Eesti Loodus,* 12, 806–812.

Kivi, R. & Russak, V. 1990. Precipitation. In *Climate of Tartu and Its Changes During the Recent Decades* (Kivi, R., ed.). Eesti TA AAI, Tartu, 137–156.

Kongo, A. 1973. Võrtsjärve ümbruse füüsilisgeograafiline iseloomustus. In *Võrtsjärv* (Timm, T., ed.). Valgus, Tallinn, 17–25.

Mäemets, Aare. 1972. Veel mageveevarudest ja nende kasutamisest I. *Eesti Loodus,* 8, 449–455.

Mäemets, Aime. 1973. Kõrgem taimestik. In *Võrtsjärv* (Timm, T., ed.). Valgus, Tallinn, 77–82.

Malinina, T. I., Filatova, I. V. & Filatov, N. N. 1985. Long-term changes in the elements of water budget of Lake Ladoga. In *Problemy issledovaniya krupnykh ozer*. Nauka, Leningrad, 79–81 (in Russian).

Masanova, M. D. & Filatova, I. V. 1985. Probability structure of interannual water level changes in North-Western lakes. In *Problemy issledovaniya krupnykh ozer*. Nauka, Leningrad, 81–84 (in Russian).

Mühlen, M. von zur. 1919. Zur Geologie und Hydrologie des Wirtzjärv-Sees. *Abhandlungen der Königlichen Preussischen Geologischen Landesanstalt. Neue Folge,* **13**.

Nicklisch, A., Conrad, B. & Kohl, J.-G. 1981. Growth kinetics of *Oscillatoria redekei* van Goor as a basis to understand its mass development in eutrophic lakes. *Verh. Internat. Verein. Limnol.*, **21**, 3, 1427–1431.

Nõges, P. 1995. Long-term decrease in phyto- and bacterioplankton abundance in shallow, strongly eutrophic Lake Võrtsjärv (Estonia): Seeming or real improvement of the trophic state? In *Proc. 6th Intern. Conf. Conservation & Management Lakes – Kasumigaura'95*, **2**, 818–821.

Nõges, P., Nõges, T., Laugaste, R., Haberman, J. & Kisand, V. 1997. Tendencies and relationships in the pelagic environment and plankton community of Lake Võrtsjärv in 1964–93. *Proc. Estonian Acad. Sci. Biol. Ecol.*, **46,** 1/2, 40–57.

Orviku, K. 1973. Võrtsjärve geoloogilisest arengust. In *Võrtsjärv* (Timm, T., ed.). Valgus, Tallinn, 26–32.

Pihu, E. 1959. Kalade kudemisest Võrtsjärves 1956. ja 1957. aasta kevadel. In *Eesti NSV Teaduste Akadeemia juures asuva Loodusuurijate Seltsi aastaraamat*, **51**, 225–247.

Pihu, E. 1998. Fishes and fisheries management in Lake Võrtsjärv. *Limnologica,* **28**, 1, 91–94.

Riley, G. A. 1957. Phytoplankton of the North Central Sargasso Sea. *Limnol. Oceanogr.*, **2**, 252–270.

Ritzrau, W. & Graf, G. 1992. Increase of microbial biomass in the benthic turbidity zone of Kiel Bight after resuspension by a storm event. *Limnol. Oceanogr.*, **37**, 5, 1081–1086.

Rumyantsev, V. B., Druzhinin, G. V. & Tshekhin, L. P. 1985. Present variability of optical properties of water in large lakes. In *Problemy issledovaniya krupnykh ozer*. Nauka, Leningrad, 103–107 (in Russian).

Sakshaug, E. 1980. Problems in the methodology of studying phytoplankton. In *The Physiological Ecology of Phytoplankton. Studies on Ecology*, 7 (Morris, J., ed.). Blackwell Scientific Publications, Oxford, 57–91.

Sievers, G. von. 1854. Flussfahrt auf dem Embach, von dessen Ausfluss aus dem Wirzjärw, bis Dorpat. *Archiv für Naturkunde Liv-, Ehst- und Kurland,* **1**, 1, 353–366.

Tilzer, M. M. 1987. The productivity of phytoplankton and its control by resource availability. A review. In *Phycotalk* (Kumar, H. D., ed.). Bavaras Hindi University, Varanasi, 3–38.

Veber, K. 1973. Põhjasetete geoloogiast ja levikust. In *Võrtsjärv* (Timm, T., ed.). Valgus, Tallinn, 33–36.

Wainwright, S. C. 1987. Stimulation of heterotrophic microplankton production by resuspended marine sediment. *Science,* **238**, 1710–1711.

Williams, P. J. le B. & Jenkinson, N. W. 1980. A simple and inexpensive 4-π light collector and two designs for a light meter for light attenuation studies. *Freshwat. Biol.*, **10**, 6, 491–496.

Zingel, P. Pelagic ciliated protozoa in a shallow eutrophic lake: Influence of water level on community composition and abundance. *Archiv für Hüdrobiol.* (submitted).

MUUTLIKU VEETASEME MÕJU VÕRTSJÄRVE ÖKOSÜSTEEMILE

Peeter NÕGES ja Tiina NÕGES

Madal häguse veega eutroofne Võrtsjärv (270 km^2) on Baltimaade suuruselt teine järv. Väikese keskmise sügavuse (2,7 m) kõrval on Võrtsjärvele iseloomulik veetaseme suur muutlikkus nii sesoonselt kui ka aastati. Aasta keskmine ja suurim veetaseme muutumise ulatus on vastavalt 1,38 ja 2,20 m; absoluutvahemik 3,20 m. Veerikaste ja veevaeste aastate keskmiste veeseisude vahe võib erineda rohkem kui meetri võrra. Veetaseme pikaajaliste mõõtmiste tulemused näitavad ligikaudu 30-aastase tsükli olemasolu kõrge- ja madalaveeliste aastate vaheldumises. Veevaeseid aastaid iseloomustab sinivetikate massiline areng, järve kinnikasvamine, haugi kudealade vähesus ning tihti ka kalade suremine.

Järve keskmise sügavuse suurenemisel väheneb tunduvalt resuspensioonikiirus. Hoolimata toitainete (N, P) suuremast koormusest veerikastel aastatel, on veetaseme tõusul positiivne mõju järve ökoloogilisele seisundile: kahaneb füto- ja bakterplanktoni biomass ning sestoni üldhulk. Troofsustaseme näiline alanemine seostub eelkõige fütoplanktoni produktsiooni langusega teravnenud valguslimitatsiooni tõttu. Väikese läbipaistvusega Võrtsjärve puhul põhjustab veetaseme tõus nii järve põhja kui ka veekeha keskmise valgustatuse vähenemist.

Artiklis on kirjeldatud Võrtsjärve elustiku reaktsiooni muutuvale veetasemele ja toodud esile mõned ökoloogilised kriteeriumid, mida saaks aluseks võtta veetaseme optimaalse regulatsioonistsenaariumi valimisel.