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EUTROPHICATION IN LAKES

Updating water quality targets for shallow Lake Balaton (Hungary), recovering from eutrophication

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Abstract The paper presents an overview about recovery of shallow Lake Balaton from eutrophication by assessing quantitative and qualitative changes in phytoplankton, zooplankton, and chironomids as a function of load reduction. The aim was to update the present water quality targets. The proposed targeting scheme supplements the existing one with a range of lake-specific ecological criteria. We conclude that simple targets (desired phytoplankton biomass and permissible load) are the best choice during the initial stage of eutrophication management, but more complex schemes including ecological criteria are needed to trace recovery when re-organization of the ecosystem takes place.

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Eutrophication of shallow lakes with special reference to Lake Taihu, China

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Introduction

Water quality targets serve as a measuring scale during eutrophication management that orients managers and the public in questions like “How much can the status of the lake be improved?”, “What has been achieved by the implemented measures?”, “Which are the main tasks of future management?” Target conditions are usually defined when intolerable economic and ecological consequences of eutrophication call for immediate action that is when the ecosystem had spectacularly transgressed steady state. In the best case, targets are set up using information about past steady state and changes observed during nutrient enrichment. The implicit initial assumption is that apart from some delay, the path of recovery will mirror the trajectory of eutrophication with a resulting steady state identical to the pre-eutrophication status of the lake. Full-scale recovery, however, has not been convincingly confirmed yet, and comprehensive studies of lake recovery suggest hysteretic effects due to both internal P load and biological resilience.

Sas (1989) and coworkers have recognized four stages during lake recovery. No decrease in algal biomass can be expected until external and internal P loads decrease below the level at which P regains its role in determining carrying capacity of the lake (Stage 1). In Stage 2 transparency shows a disproportionately large increase relative to the decrease in area-specific algal biomass due to dispersion of buoyant cyanobacteria and/or motile algae to greater depths with high nutrient concentration. This behavioural response to load reduction can be observed in deep lakes with stable stratification but not in shallow lakes. Once P has become growth limiting, the decrease of phytoplankton biomass is proportionate to the decrease in P supply (Stage 3). During Stage 4, a further reduction in algal biomass is coupled to floristic changes, most conspicuously to the retreat of cyanobacteria. Due to random fluctuations in the biomass and composition of phytoplankton, the various stages have no fixed boundaries. Although not considered in the study of Sas (1989), full recovery of a lake certainly requires one more step (Stage 5): regeneration of former macrovegetation and food web structure (Scheffer et al., 1993). Considering this general pattern of recovery as well as deviations of individual lakes from the "rule", occasional updating of water quality targets is inevitable to maintain the orienting function of these targets over the whole recovery process.

In the present study we overview the recovery of large (596 km^2), shallow (z_{mean} = 3.2 m) Lake Balaton in context of the above general model. Individual processes including the relationship between P load and phytoplankton biomass (Istvánovics & Somlyódy, 2001; Istvánovics et al., 2002), floristic changes of phytoplankton in response to restoration measures (Padisák & Reynolds, 1998), long-term changes in the biomass, composition and production of zooplankton (G.-Tóth, 2000), and the relationship between phytoplankton production and long-term dynamics of the chironomid fauna (Specziár & Vörös, 2001) have been published separately. The present overview was motivated by the practical aim of updating water quality targets considering both the observed response of the ecosystem to eutrophication management and realistic perspectives of further reduction in nutrient loads.

Data and methods

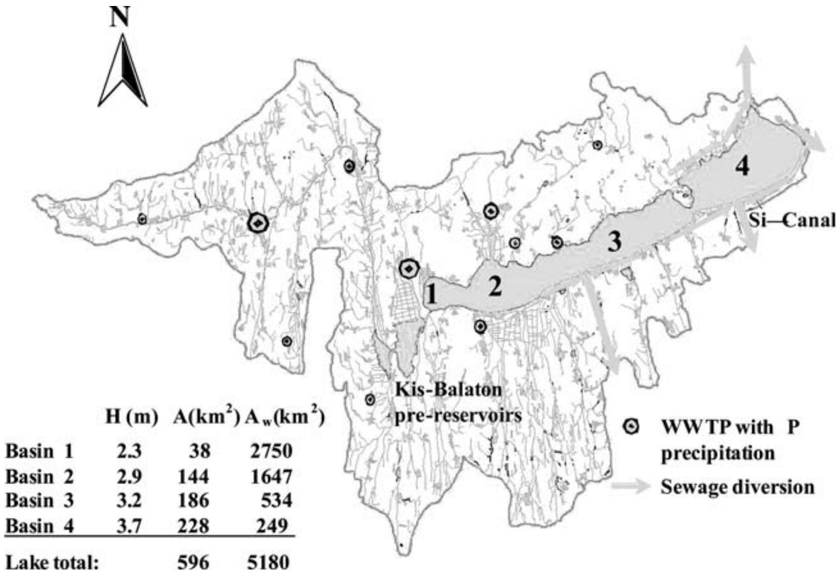
In order to detect trends and clarify patterns, we attempted to use as long data sets as possible. This could be achieved by supplementing the authors own data with critically evaluated historical records from both published sources and various unpublished archives. Whenever quantification was feasible, simple empirical relationships were applied.

Recovery of the lake was traced using one or both of the two independent approaches: (i) analysis of the trajectory of oligotrophication within the same lake area and (ii) comparison of the areas that had previously attained hypertrophy and mesotrophy/slight eutrophy. The advantage of the latter approach is that the influence of random variability caused by physical forcing is partly cancelled (Herodek, 1984). At the same time, poorly explored natural gradients within the lake may impose limitations to the application of this method.

Lake Balaton, its eutrophication and restoration

On the basis of large-scale circulation patterns (Shanahan et al., 1986), Lake Balaton can be divided into four basins (Fig. 1). Mean depth and surface area of the basins increase from west to east with a concomitant decrease in the area of the corresponding sub-watershed (Fig. 1). The largest tributary, the Zala River enters Basin 1, whereas the only outflow is the Sió Canal at the opposite end of the lake. The lake is extremely calcareous with high magnesian calcite making up 50–60% of sediments (Csermák & Máté, 2004). Turbidity is high particularly during summer due to both wind induced sediment resuspension and slow sedimentation of the precipitating carbonates (Istvánovics et al., 2004). Strong wave action restricts macrophyte growth to a narrow littoral zone (during the past 150 years, maximum 10% of lake area; Virág, 1998).

Specific morphometry gives rise to environmental gradients along the longitudinal axis from increasing water retention time to shifts in major ion composition and to decreasing area-specific nutrient loads. One can expect associated



	H (m)	A(km^2)	A$_w$(km^2)
Basin 1	2.3	38	2750
Basin 2	2.9	144	1647
Basin 3	3.2	186	534
Basin 4	3.7	228	249
Lake total:		596	5180

Fig. 1 Lake Balaton and its catchment. (H—average depth, A—surface area, Aw—area of sub-catchment, WWTP—waste water treatment plant)

differentiation in the biotic components under pristine conditions, but the available evidence is faint.

Eutrophication dramatically magnified many of the environmental gradients and particularly the biotic ones. By the end of the 1970s, area-specific P load in Basin 1 exceeded that in Basin 4 by a factor of 8 (Table 1, Somlyódy & Jolánkai, 1986). Primary production has increased 6–8-fold in Basin 1 and doubled in Basin 4 (Herodek, 1986). Blooms of N_2-fixing cyanobacteria developed regularly during summers in the western basins. Four years after its first appearance in 1978 (Padisák, 1997), invasive *Cylindrospermopsis raciborskii* (Woloszynska) Seenayya et Subba Raju became the dominant summer species in each basin (Herodek, 1986; Padisák & Reynolds, 1998; Istvánovics et al., 2002). Significant differences between the western and eastern areas have been reported in composition and production of

Table 1 Multi-annual mean loads (±sd) of total P in tons P year^{-1}

Source	Lake Balaton	Basin 1	Basin 2	Basin 3	Basin 4
Before eutrophication management (*1980–1988*)					
Geochemical background	18 ± 4	10 ± 2	6 ± 1	2	1
Sewage	61 ± 15	31 ± 7	10 ± 1	5 ± 4	16 ± 10
Urban runoff	56 ± 9	16 ± 4	19 ± 4	10 ± 1	10 ± 2
Agricultural (rural) runoff	137 ± 30	39 ± 17	42 ± 14	30 ± 5	26 ± 4
Atmosphearic	17 ± 2	1	4 ± 1	5 ± 1	7 ± 1
Total	289 ± 42	97 ± 23	80 ± 19	53 ± 4	60 ± 10
Through-flow			20 ± 4	21 ± 7	12 ± 6
After eutrophication management (*1989–2002*)					
Geochemical background	17 ± 6	9 ± 3	5 ± 2	2 ± 1	1
Sewage[a]	11 ± 3	7 ± 2	3 ± 1	1	0
Urban runoff	34 ± 11	9 ± 3	12 ± 4	6 ± 2	7 ± 3
Agricultural (rural) runoff	74 ± 27	20 ± 11	23 ± 8	14 ± 5	17 ± 6
Atmospheric	10 ± 5	1	2 ± 1	3 ± 2	4 ± 2
Total	145 ± 47	45 ± 19	46 ± 14	26 ± 8	29 ± 9
Through-flow			18 ± 9	21 ± 12	15 ± 11

a—value in 2002

the chironomid fauna (Dévai & Moldován, 1983; Ponyi et al., 1983) as well as in those of fish (Bíró & Vörös, 1990; Bíró, 1997).

In addition to its ecological values, Balaton is an internationally recognized recreational lake. Eutrophication caused significant economic loss in times when two thirds of Hungary's tourist income originated from the Balaton region. An inclusive eutrophication management plan was approved by the government in 1983. The aim was preventing further deterioration in water quality by 1990 and achieving the trophic conditions experienced during the early 1960s in the period 2005–2100 (Láng, 1986). Water quality targets and permissible phosphorus loads were also expressed in quantitative terms.

The most important management measures included (i) sewage diversion from shoreline settlements around Basin 4 and along the southern shore of Basin 3, (ii) introduction of chemical P precipitation at the large waste water treatment plants in the watershed, and (iii) pre-reservoir construction on the Zala River (Fig. 1). In the meantime, Hungarian agriculture has collapsed following the political changes in 1989. From the perspective of eutrophication, the most important consequence was a 90% drop in fertilizer application. Two periods of drought during 1990–1993 and 2000–2003 further decreased diffuse load of nutrients (Fig. 2). Multi-annual mean load of P decreased by about 50% in each basin (Clement et al., in press; Table 1). Considering, however, the additional load originating from through-flow, the reduction was only 30–40% with the exception of Basin 1 (Table 1).

Results and discussion

The relationship between P load and algal biomass

In the western basins, annual mean biomass of algae decreased significantly after a delay of about one decade (Fig. 3). Although the last few summers were unusually warm and dry, the former adverse conditions have not returned. In the eastern basins, particularly in Basin 3 the improvement was marginal if any. In the period 1995–2003, annual average biomass of phytoplankton exceeded the prescribed range in 5 years (Fig. 3). The mean deviation was the lowest (25%) in Basin 1 and the highest (55%) in

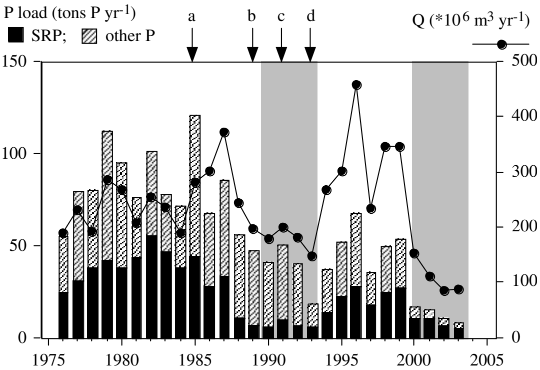


Fig. 2 Loads of soluble reactive P (SRP) and total P (sum of SRP and other P forms), and flow (Q) at the mouth section of the Zala River. (Arrows indicate timing of load reduction (Fig. 1) as follows: a—inundation of the Upper Kis-Balaton reservoir; b—sudden drop in fertilizer application due to the collapse of agriculture; c—introduction of chemical P removal at the sewage treatment plant of Zalaegerszeg, d—inundation of the Lower Kis-Balaton reservoir. Shaded areas indicate two periods of draught)

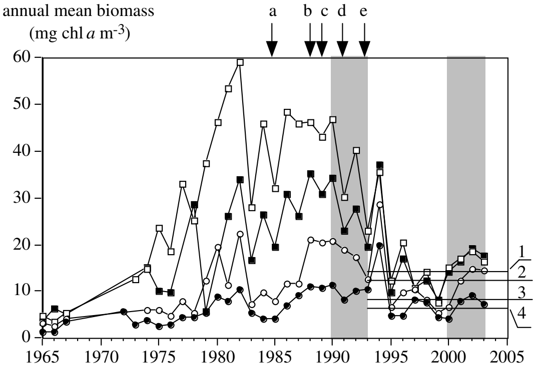


Fig. 3 Annual mean biomass of phytoplankton in the four basins of Lake Balaton. (Basin 1—open square, Basin 2—closed square, Basin 3—open circle, Basin 4—closed circle. Arrows indicate timing of load reduction (Fig. 1) as follows: a—inundation of the Upper Kis-Balaton reservoir; b—sewage diversion from the eastern basins; c—sudden drop in fertilizer application due to the collapse of agriculture; d—introduction of chemical P removal at the sewage treatment plant of Zalaegerszeg, e—inundation of the Lower Kis-Balaton reservoir. Shaded areas indicate two periods of draught. Numbered lines show water quality objectives to be achieved by 2010 in the four basins (cf. Table 2))

Basin 3. The conclusion is that in spite of the conspicuous improvement in the western areas, the lake is still far away from a full recovery.

Due to the large number of small, dynamic inflows (over 50, Fig. 1), annual load estimates are uncertain with the exception of Basin 1 where daily loads through the Zala River have been available from 1976 (Clement, 2000). Consequently, the relationship between algal biomass and phosphorus load can be traced at a higher resolution in this westernmost area than in other basins. The question is then how much the results are transferable to the other basins.

Pre-reservoir construction caused spectacular changes in the loading conditions of Basin 1 in addition to the reduction in absolute terms (Fig. 2). The most important ones were (i) an 80% decrease in the load of suspended solids, (ii) the increase in the calcium load related to total P and suspended solids loads by a factor of 2.5 and 4, respectively, and (iii) the decrease in the mean loading ratio of nitrate-N to phosphate-P below 2 (by weight) during summer. Changes in (i) and (ii) altered the apparent settling velocity of P and improved the composition of the freshly formed sediments (Istvánovics & Somlyódy, 2001), whereas (iii) may favor selection of N_2-fixing cyanobacteria.

Internal load of P is one of the basic processes that determine the path of recovery in shallow lakes (Marsden, 1989; Sas, 1989; Søndergaard et al., 1999). In well-oxygenated Lake Balaton with low concentrations of iron in the sediments (12–20 mg Fe g^{-1} dry mass), resuspension followed by P desoprtion is the predominant mechanism of transport (Boström et al., 1982; Lijklema et al., 1986). At long time scales relevant for recovery, transport can be considered constant provided that no significant changes occur in macrophyte coverage. In such lakes, long-term changes in the internal load depend on the concentration of mobile sediment P. These changes are rarely large enough to detect them directly (Søndergaard et al., 1999). Therefore we estimated long-term trends in mobile sediment P using the mass balance model of Lijklema and coworkers (1986) for the upper sediment layer. Since loads of Ca and suspended solids were available only for the Zala River, additional sources (about 5–10% of the total load) were neglected.

The model (Fig. 4) assumes that the "active" layer is fully mixed to a constant depth of h (m) due primarily to the burrowing activity of

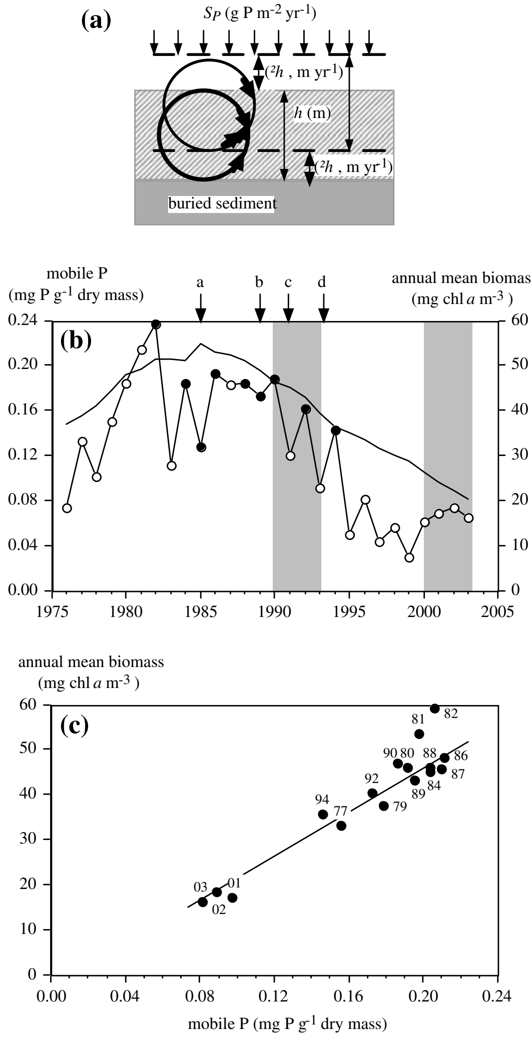


Fig. 4 Mass balance model of Lijklema et al. (1986) (**a**) and the estimated mobile P content of sediments (solid line) compared with annual mean biomass of phytoplankton (circle; **b**) in Basin 1. The relationship between mobile P content and phytoplankton biomass in years when internal P load approached its potential maximum (**c**). (In A closed circles denote years with large blooms of *Cylindrospermopsis raciborskii* (Fig. 6). Arrows indicate timing of load reduction (Fig. 1) as follows: a—inundation of the Upper Kis-Balaton reservoir; b—sudden drop in fertilizer application due to the collapse of agriculture; c—introduction of chemical P removal at the sewage treatment plant of Zalaegerszeg, d—inundation of the Lower Kis-Balaton reservoir. Shaded areas indicate two periods of draught. In B parameters of the regression line are $y = 234.94x$, $r^2 = 0.88$, $n = 16$)

zoobenthos. The rate of sedimentation (Δh, m year^{-1}) can be estimated from the mass balance of suspended solids and calcium. To keep the mixed depth constant, burial must equal the rate of sedimentation. Rate of P sedimentation (S_P, g P m^{-2} year^{-1}) can be derived from the mass balance of total P. The change in the P concentration of the active layer (P, g P m^{-3}) is

$$P = \left(P_0 - \frac{S_P}{\Delta h + kh}\right) \cdot \exp\left(-\frac{\Delta h + kh}{h} t\right) + \frac{S_P}{\Delta h + kh}$$

where P_0 is the initial concentration at $t = 0$ and k (year^{-1}) is the first-order “nutrient decay” rate. For conservative nutrients (like total P) $k = 0$, whereas for non-conservative ones (like total N or mobile P) $k > 0$. Diagenetic processes including re-crystallization of carbonates and exchange of Ca for Mg may result in gradual immobilization of mobile P in the sediments of Lake Balaton. The new equilibrium concentration is $P_{\mathrm{eq}} = S_P/(\Delta h + kh)$. The rate of approaching to equilibrium is controlled by the exponent, $\tau = h/(\Delta h + kh)$.

Using the parameters calibrated in 1999 (P_0 = 67 g P m^{-3} = 0.15 mg P g^{-1} dry mass, h = 0.15 m, and k = 0.1 m year^{-1}; Istvánovics & Somlyódy, 2001), the rapid decrease in the mobile P pool is still in progress (Fig. 4). Considering that the concentration of mobile P sets an upper limit to the internal P load, the output of the model can be interpreted as the trend in the potential maximum of internal P load. This maximum has been fully exploited (i) during the initial period after the reduction in the external load, (ii) during the years of large blooms of *C. raciborskii* (there is a partial overlap between (i) and (ii)), and (iii) during the recent dry years when the weir of the Zala River was closed from about June to November. In other years physical constraints, biotic interactions or both might keep the biomass of phytoplankton below the highest achievable level.

In years when internal P load approached to the potential maximum, strong correlation (r^2 = 0.88, n = 16, Fig. 4) was observed between annual mean biomass of phytoplankton and concentration of mobile P. This suggests that the ultimate reason for the fast improvement in the western areas is the favorable long-term behavior of the highly calcareous sediments. The relationship can be used to predict future biomass of phytoplankton in Basin 1 under various loading scenarios.

The OECD approach (Vollenweider & Kerekes, 1982; Fig. 5) was used to explore the response of algal biomass to the reduction in external load of P in the various basins. Mean in-lake concentration of P (P, g P m^{-3}) was calculated from the area-specific P and hydraulic loads (l_{IN}, g P m^{-2} year^{-1} and q, m year^{-1}, respectively) taking into account both inflow and through-flow:

$$P_\lambda = \frac{l_{\mathrm{IN}}}{q} \cdot \frac{1}{1 + 2\sqrt{\tau}}$$

where τ (year) is the theoretical water retention time.

Deteriorating water balance of the lake cancelled the effect of reduction in nutrient loads on P in the eastern basins (Fig. 5; Istvánovics et al., 2002). Pre- and post-management status of the various basins seemed to follow a common linear curve suggesting that the behavior of the different areas can be considered fairly similar for managerial purposes.

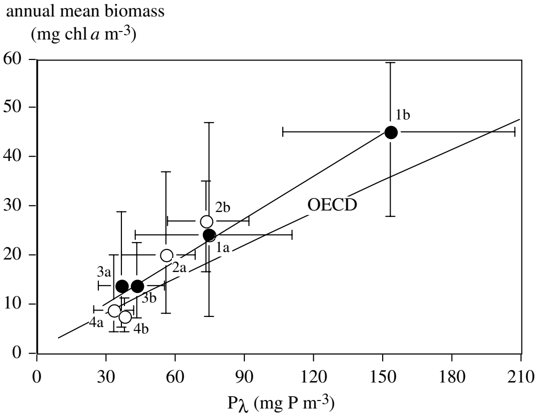


Fig. 5 Multi-annual mean biomass of phytoplankton as a function of the in-lake concentration of P (P_λ). (X and Y bars denote the minimum and maximum values during the period of averaging. Numbers refer to the four basins, “b” refers to the period before management (1980–1988), “a” refers to the period after management (1989–2003). The OECD model line and the linear trend for the four basins of Lake Balaton are given)

Changes in the composition of phytoplankton

In the eastern basins, phytoplankton composition and dynamics showed little difference during the 1970–1980s from those during the 1950s. *Cyclotella ocellata* Pantocsek and *C. compta* (Ehrenberg) Kützing dominated the spring bloom of diatoms. Main species of the summer maximum were *Ceratium hirundinella* (O.F. Müller) Schrank, meroplanktonic diatoms (*Aulacoseira granulata* (Ehrenberg) Simonsen, *A. granulata* var. *angustissima* (O. Müller) Simonsen and *Snowella lacustris* (Chodat) Komárek et Hindák. *Aphanizomenon klebahnii* Elenkin was also present but never dominant. In contrast to this, monomodal succession with a single large summer bloom characterized the phytoplankton in Basin 1 during the 1970s. Blooms of N_2-fixing cyanobacteria (mainly *Anabaena spiroides* Klebahn and *A. flos-aquae* (Lyngbye) Brébisson) might develop from as early as June and collapse as late as October. New cyanobacterial species were recorded almost each year. Floristic data collected during more than 70 years are clearly indicative of fast structural changes preceding the rapid increase in algal biomass.

Structural resilience could clearly be observed following the reduction in external loads. *Cylindrospermopsis raciborskii* continued to form large blooms in the western areas until 1994 (Fig. 6). A sudden shift occurred in 1995: the share of cyanobacteria has not exceeded 30% from this year and duration of the summer bloom shortened significantly (Fig. 6). Although the seasonal succession was monomodal in many years, the ratio of the summer to spring bloom tended to decrease.

In the eastern areas, the slight increase in annual mean biomass during the early post-management period (Fig. 3) coincided with the increasing share of *C. raciborskii* (Fig. 6). In the period 1988–1996, cyanobacteria contributed to the summer maximum by more than 20% in 7 years, whereas previously this occurred only in 1982 (Padisák, 1994; Padisák & Reynolds, 1998). The possible explanation is that blooms in the

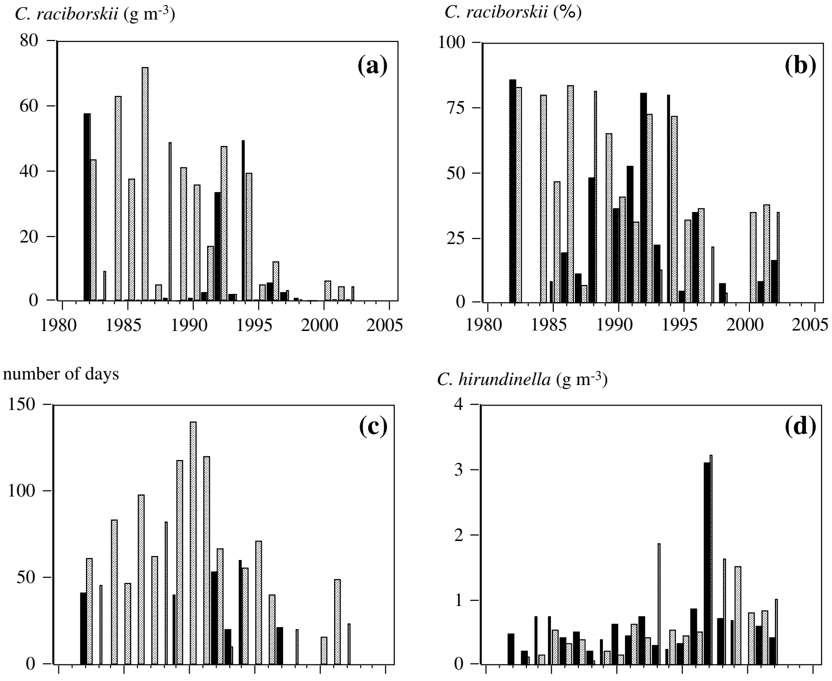


Fig. 6 Maximum biomass (**a**) and contribution to the total biomass during the cyanobacteria bloom (**b**) of *Cylindrospermopsis raciborskii*, number of days when total biomass exceeded 10 g m^{-3} (**c**), and maximum biomass of *Ceratium hirundinella* (**d**) in Basin 1 (dotted bar) and Basin 4 (closed bar). (No data were available in Basin 4 in 2000)

western areas may disperse in 3–4 weeks to inoculate the eastern ones (Istvánovics et al., 2002; Koncsos, 2003).

Apart from *C. raciborskii*, there are two species in Lake Balaton that deserve special attention due to their potential toxicity and/or ability to form excessive blooms. The one is *Aphanizomenon flos-aquae* (Lyngbye) Ralfs, which may form surface scum at the relatively low whole-column biomass of about 4 g m^{-3}. The other one is *Planktothrix agardhii* (Gomont) Anagnostidis et Komárek that bloomed first in the lake in 1989. Its share reached 20% and 50% of the maximum biomass in Basins 1 and 4, respectively. The first bloom might represent one of the unexpected consequences of pre-reservoir construction. By 1988, the upper reservoir (Fig. 1) developed into a hypertrophic pond with yearly mean chlorophyll concentration of 150 mg m^{-3} at the outflow. *P. agardhii* bloomed almost all around the year (Mátyás, 1996), and Basin 1 received a substantial *P. agardhii* "load" between 1988 and 1992. Although the reed-covered lower reservoir has fully eliminated this inoculation from 1993, the cyanobacterium established itself in the western areas of Lake Balaton and made up 30% of the maximum biomass in some years.

Ceratium hirundinella is one of the most successful canopy species of phytoplankton that was historically the prominent summer alga of Lake Balaton (Sebestyén, 1953). Its biomass remained relatively low and stable in Basin 4 (Fig. 6). In contrast to this, biomass of *C. hirundinella* was negatively correlated with that of *Cylindrospermopsis raciborskii* (r = –0.59) in Basin 1. The recent increase of *Ceratium hirundinella* (Fig. 6) is indicative of the beginning of structural recovery in the western areas.

In order to exploit the predictive power provided by compositional changes of phytoplankton, Padisák and coworkers (2003) elaborated a novel indicator, the *Q* value. Phytoplankton guilds/associations were identified in accordance with the phyto-sociological classification of Reynolds and coworkers (2002). Considering growth rates, life history, toxicity and distribution of species in 27 guilds, a water quality factor ranging from 2 to 10 was assigned to each functional group. A higher score indicates better quality. Five water quality classes were set up using the biomass-based relative frequency of the guilds weighted by the water quality factor. Presence of potentially toxic or bloom-forming species decreases the value of *Q* even if the total biomass is low. Vice versa, *Q* may attain a high value when the biomass is relatively high but bloom formation is unlikely in the absence of appropriate species.

The *Q* index was tested on phytoplankton time series from Lake Balaton at various temporal scales (Fig. 7). Annual mean biomass of phytoplankton was low in Basin 4 during the period 1983–1988. The *Q* index, however, qualified the area only as moderate or good. Indeed, annual mean biomass increased in the subsequent years

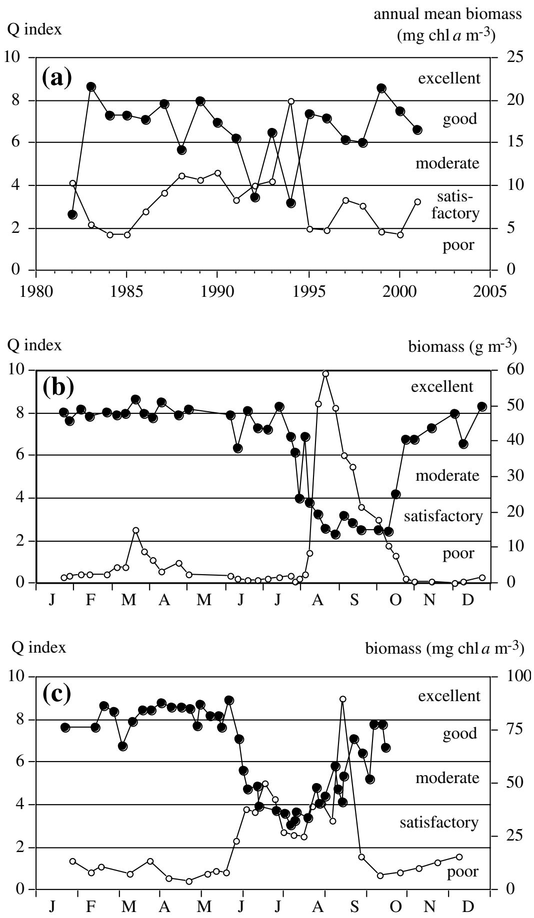


Fig. 7 The value of the *Q* index (closed circle) and the biomass of phytoplankton (open circle) in various years in Basin 4 (**a**), in 1994 in Basin 4 (**b**), and in 2002 in Basin 1 (**c**). (The *Q*-based classification of water quality is shown)

in spite of the reduction in nutrient loads (Fig. 3). At a finer scale, the predictive value of Q was even more striking. Low values of Q warned before the onset of the summer bloom of cyanobacteria that water quality problems were likely to occur.

Changes in the biomass and composition of crustacean zooplankton

Frequent sediment resuspension and intense carbonate precipitation makes Lake Balaton a highly stressful habitat for filter feeding zooplankton (G.-Tóth, 1992). Before eutrophication, *Eudiaptomus gracilis* (Kiefer, 1932) made up more than 80% of crustacean zooplankton. As an average, 8–10% of primary production was channeled to zooplankton. This is low in comparison to most other temperate lakes (G.-Tóth, 1991). The two important consequences are the prevalence of bottom–up regulation of zooplankton and low efficiency of energy utilization by fish (around 0.01%; Bíró & Vörös, 1982).

The moderate increase in the biomass of crustacean zooplankton by a factor of 1.5–3 was disproportionately low compared to the nearly order of magnitude increase in primary production in the western areas. The reasons of uncoupling might be manifold. (i) The share of inedible algae increased within the phytoplankton. The typically low community clearance rates decreased to negligible during blooms of filamentous cyanobacteria. (ii) Quality of ingestible algae might decrease due to frequent N deficiency. (iii) Elevated primary production further enhanced gross carbonate precipitation and thus counterbalanced the increase in the ratio of food to inorganic particles. In general, the most selective feeders (*Cyclops vicinus* (Uljanin, 1875), *Acanthocyclops vernalis* (Fischer, 1853), *A. vernalis* v. *robustus* (Sars, 1863)) benefited the most from eutrophication. Structural re-organization could be observed after 1996. Number of *E. gracilis* nauplii increased by a factor of 2–3 indicating the improved physiological condition of this species that took back its former leading position in the crustacean zooplankton.

Conclusively, crustacean zooplankton is hardly a suitable group for characterizing trophic status of Lake Balaton due to extreme physical constraints acting upon its composition and biomass.

Changes in the biomass and composition of the chironomid fauna

During the last 150–200 years, *Procladius choreus* (Meigen, 1804), *Tanypus punctipennis* (Meigen, 1818) and *Chironomus balatonicus* (Dévai, Wülker et Scholl, 1983) made up about 90% of the biomass and production of chironomid fauna in the profundal of Lake Balaton (Dévai & Moldován, 1983; Specziár & Vörös, 2001). Of them, carnivorous *P. choreus* was usually present in the highest densities.

Substantial structural and quantitative differences developed along the longitudinal axis of Lake Balaton during the period of eutrophication (Fig. 8; Dévai, 1990; Specziár & Vörös, 2001). The biomass remained low in Basin 4 (0.03–0.6 g dry mass m^{-2} depending on the season and the year), and *P. choreus* preserved its leading position. In the western areas, the biomass increased by a factor of 2–8 (0.2–4.25 g m^{-2})

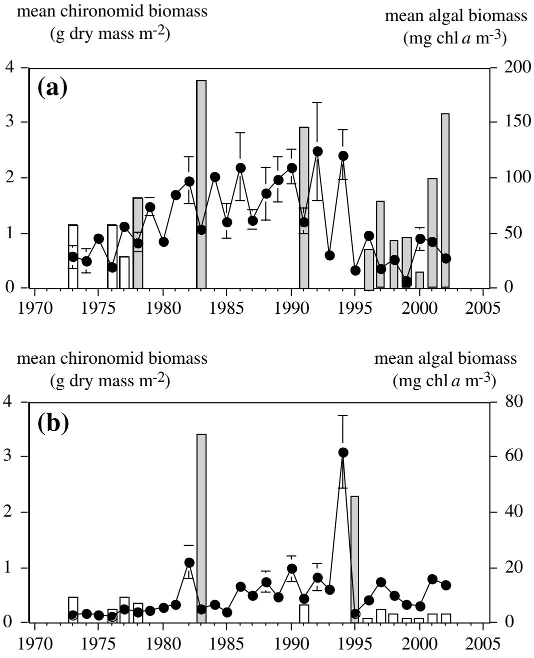


Fig. 8 Annual mean biomass of chironomid larvae (bars) and mean biomass (± sd) of phytoplankton (circle) during summer (July–September) in Basins 1 (**a**) and 4 (**b**)

indicating the relatively strong coupling between algal and chironomid production. The quantitative change was associated with a shift from *P. choreus* dominance to the dominance of phyto- and detritivorous *Ch. balatonicus*.

From the early post-management period, data are available only in 1991. Delayed response of phytoplankton seemed to postpone re-structuring of chironomid assemblages in Basin 1 (Fig. 8). From 1996, mean chironomid biomass tended to decrease but remained higher (0.3–3.1 g m^{-2}) than in the other basins (0.07–0.3 g m^{-2}). Considering the similar biomass of phytoplankton in Basins 1 and 2 (Fig. 3), the difference in the chironomid biomass between the two areas was surprising. The explanation lays in the divergent structure of chironomid assemblages. While *Ch. balatonicus* defended its dominance in Basin 1, *P. choreus* became once again the most abundant species in the other basins. This observation, along with structural changes during eutrophication suggests that high chironomid biomass (>0.5 g m^{-2}) is invariably associated with the dominance of *Ch. balatonicus*. Additional evidence also supports this contention. In Basin 4, two large chironomid peaks were observed following the bloom of *C. raciborskii* in 1982 and 1994. In both cases, *Ch. balatonicus* dominated the chironomid community (*cf.* Figs. 6 and 8).

The above pattern indicates that *Ch. balatonicus* may show a non-linear response to resource availability. To further explore this point, spring biomass of *Ch. balatonicus* was compared to phytoplankton biomass during the previous summer (Fig. 9). Above a threshold chlorophyll concentration of 20–30 mg m^{-3}, abundance of *Ch. balatonicus* exceeded 700–800 ind m^{-2} and the species dominated the chironomid fauna. Below this threshold, its abundance was close to the limit of detectability (≤1 ind m^{-2}). In the latter case, the importance of top–down control by fish increased (Tátrai, 1980; Bíró, 1997).

The shift from *P. choreus* to *Ch. balatonicus* dominance was shown to double (from 0.3–0.6% to 0.7–1.3%) the efficiency of energy transfer from primary producers to chironomids (Specziár & Vörös, 2001; Specziár, 2004). This, in turn, basically influenced population dynamics of several fish species including those that forage primarily for chironomids (*Abramis brama* (Linnaeus, 1758), *Gymnocephalus cernuus* (Linnaeus, 1758)) and those that temporarily supplement their diet with these organisms (*Rutilus rutilus* (Linnaeus, 1758), *Pelecus cultratus*

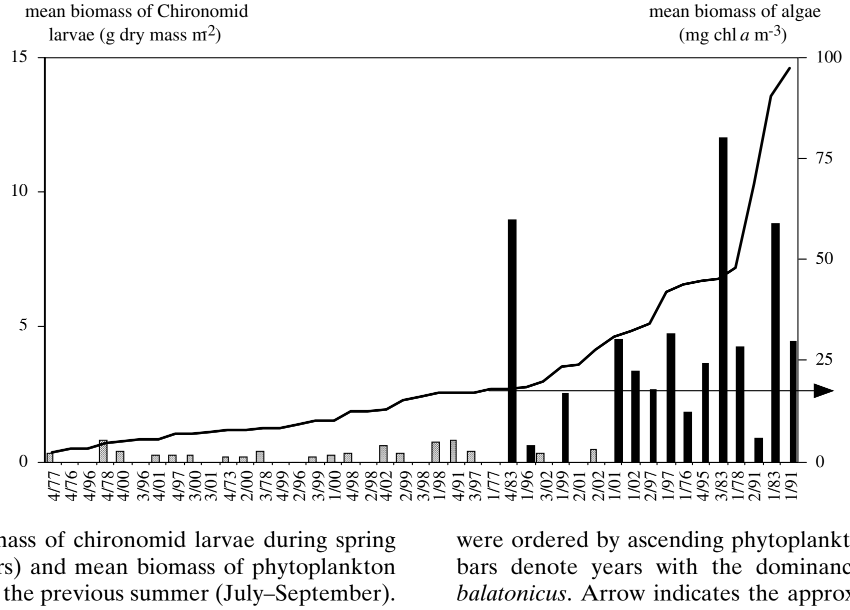


Fig. 9 Mean biomass of chironomid larvae during spring (March–April; bars) and mean biomass of phytoplankton (solid line) during the previous summer (July–September). (In the Z/XY label format of the abscissa Z refers to the basin and XY to the year of the chironomid data. Data were ordered by ascending phytoplankton biomass. Black bars denote years with the dominance of *Chironomus balatonicus*. Arrow indicates the approximate chlorophyll threshold above which *Ch. balatonicus* is likely to be present)

(Linnaeus, 1758), *Alburnus alburnus* (Linnaeus, 1758), *Cyprinus carpio* (Linnaeus, 1758), etc.). For example, growth of *Abramis brama* had been fast during the 1970s with a particularly strong population in Basin 1. In the 1990s, growth slowed down and the bream population decreased dramatically all along the lake (Bíró, 1997).

Updating water quality targets

As a first step, we examine whether the present water quality targets and permissible loads of phosphorus (Table 2) are realistic in the case of Basin 1. Maximum biomass of algae has been predicted for 2010 and 2025 (Table 3) from the relationship between the mobile P pool of the sediments (that is the potential maximum of internal P load) and yearly mean biomass of phytoplankton (Fig. 4). Three distinct load scenarios were defined using the observed loads during the past decade (Table 3). Scenario 1 represented the mean load over the whole period after the latest large management measure taken in 1992. Scenario 2 corresponded to the mean load under average hydro-meteorological conditions. Finally Scenario 3 was the mean load experienced during the draught in 2000–2003.

Table 2 Present water quality targets prescribed for 2010

Basin	Annual mean biomass (mg chl *a* m^{-3})	BAP load (tons P year^{-1})
1	7–14	21
2	6–12	15
3	4–8	8
4	3–6	12

BAP—biologically available P

In order to achieve the present water quality targets until 2010, mean external load of P should be decreased immediately to the level of Scenario 3 (Table 3). We estimate that at the mouth section of the Zala River, 6–8 tons P year^{-1} originate from sewage emission, whereas the diffuse load of biologically available P (BAP) ranges between 10 and 15 tons P year^{-1} in average hydro-meteorological conditions. During the draught, pre-reservoirs released 1–5 tons P year^{-1} of BAP due to increased internal P load. Thus, diffuse load dropped to 0–2 tons P year^{-1}, a value below the natural background load in years with average runoff (Table 1). Consequently, no combination of management measures can reduce the external load to the level of Scenario 3. Therefore the present water quality target is unlikely to be attained by 2010.

Taking into consideration planned development in sewerage, introducing an effluent standard of 0.5 g P m^{-3} in the whole watershed will reduce BAP load from point sources by no more than 30–40%. Such reduction would decrease the external load to the level of Scenario 1. The desired water quality could then be achieved by 2025 (Table 3). However, neither Scenario 1 seems to be realistic at the present time because it contains the strong effect of dry years on mean load (Table 3). In this way improved waste water treatment alone does not guarantee the expected improvement in water quality. In order to accomplish our objectives, diffuse load of BAP experienced under mean hydro-meteorological conditions (Scenario 2, Table 3) must be reduced by 3–4 tons P year^{-1} in addition to improved waste water treatment. This reduction is to be

Table 3 Maximal expected value of annual mean algal biomass in Basin 1 assuming three constant load scenarios derived from past observations

Scenario	Period of past observation	TP load (tons P year^{-1})	BAP load (tons P year^{-1})	2010		2025	
				P_{mobil} (mg P g^{-1} dry mass)	Biomass (mg chl *a* m^{-3})	P_{mobil} (Mg P g^{-1} dry mass)	Biomass (mg chl *a* m^{-3})
1	1993–2003	37 ± 22	21 ± 12	0.066	16	0.056	13
2	1993–1999	50 ± 17	27 ± 10	0.077	18	0.074	17
3	2000–2003	15 ± 4	10 ± 2	0.047	11	0.022	5

TP—total phosphorus, BAP—biologically available phosphorus. P_{mobil}—mobil P content of the sediments. Estimates are based on the relationship in Fig. 4b. Loads of Ca and suspended solids were also assumed to be constant realized in a socio-economic environment where the unsustainable low level of fertilizer application is expected to rise, and abandoned agricultural areas are increasingly built in along the shoreline.

Considering the fairly similar behavior of various basins on the OECD plot (Fig. 5), permitted BAP loads can be expected to result in the desired water quality in each area (Table 2) by 2025.

As a second step, present water quality targets were supplemented with ecological criteria (Table 4). Replacing the simple scheme of Table 2 with a more complex classification system (Table 4) demands additional effort from the responsible authorities, which may reluctantly accept the new criteria. Nevertheless, there are at least three good reasons justifying our proposal. (i) The foregoing analysis indicated that recovery of Lake Balaton reached Stages 4 and 5. While mean biomass of phytoplankton may allow properly assessing progress of eutrophication management during Stages 1 and 3, the value of this indicator diminishes when structural changes take place during subsequent stages. (ii) Structural changes precede quantitative changes. Therefore properly elaborated structural indices may have considerable predictive power during Stages 4 and 5. (iii) The European Water Framework Directive urges classification of aquatic ecosystems using ecological indices.

As discussed above, the *Q* index of Padisák and coworkers (2003) is a suitable indicator of structural changes of phytoplankton that has been successfully tested in Lake Balaton at various time scales (Fig. 7). Thus, the index was incorporated as one of the ecological criteria (Table 4).

The long-term relationship between biomass of phytoplankton and that of chironomids (Fig. 8; Specziár & Vörös, 2001) was used to derive appropriate criteria concerning chironomid biomass (Table 4). A structural index, similar to the *Q* index has not been derived for the chironomid fauna yet. At the same time, presence or absence of *Chironomus balatonicus* was shown to basically influence overall efficiency of energy transfer from phytoplankton to fish. In this way abundance of this key species reflects important characteristics of ecosystem functioning. Moreover, among the 14 chironomid species inhabiting Lake Balaton and known to be capable of bloom formation, *Ch. balatonicus* was the single one that produced the actual blooms up to now. Its mass emergence attains an ‘‘intolerable’’ level when the biomass of chironomids exceeds 3–4 g m^{-2}. The threshold behavior of *Ch. balatonicus* (Fig. 9) allowed establishing ecological criteria in relation to the abundance of this species (Table 4).

Although we aimed at setting up the ecological targets in a consistent way, one cannot exclude that various indicators occasionally result in divergent classification. The question arises how an aggregated index of ecological status can be determined. One may accept the worst indication. Alternatively, the aggregated index may be composed from individual indices by appropriate weighting. The final choice must be made after evaluating the

Table 4 The proposed ecological target system to be realized by 2025

Basin	Ecological targets						Permissible TP load[a] (tons P year^{-1})	Load reduction[g] (%)
	Phytoplankton			Chironomidae				
	Annual MB[b]	*Q* index[c]		Maximum MB[e]	*Ch. balatonicus*[f]			
	mg chl *a* m^{-3}	Value	Classification[d]	g dry mass m^{-2}	ind m^{-2}	Classification[d]		
1	14	6–8	Good	2.0	800	Good	37	17
2	12	6–8	Good	1.5	500	Good	21	54
3	8	8–10	Excellent	0.5	<1	Excellent	17	34
4	6	8–10	Excellent	0.5	<1	Excellent	22	24

MB—mean biomass. a—refers to the average of 5 years, yearly variability can reach ±50–60%. b—using the present monitoring frequency. c—in the period July–September. d—scaling: ‘‘excellent’’, ‘‘good’’, ‘‘moderate’’, ‘‘satisfactory’’, ‘‘poor’’. e—in the period March to June. f—maximum density of *Chironomus balatonicus* larvae. g—relative to the present load experienced in years of average runoff experience of experimental introduction of the new targets during the next few years.

Unlike the present scheme that expresses permissible load of P in terms of BAP, we propose to relay on total P loads (Table 4). Several conceptual and methodological problems introduce an additional uncertainty into the estimates of BAP load compared to that of total P load. Undertaking this higher uncertainty has been rational during the initial phase of eutrophication management, when the main effort had to be concentrated on the reduction of loads from point sources (Somlyódy & Jolánkai, 1986). Future management, however, must focus on the reduction of diffuse loads and therefore BAP can be replaced with total P. The proposed total P loads (Table 4) are equivalent to the BAP prescriptions of the present scheme (Table 2). The proposed values refer to loads during the years of average runoff. Depending on the hydrometeorological conditions, the deviation may reach ±50–60%.

Concluding remarks

During the initial stage of eutrophication management, the response of various shallow lakes converges to a single pattern: ones the sum of external and internal P loads reached capacity limiting levels, the phytoplankton biomass decreases in proportion to P supply. Obviously, long-term behavior of sediments determining the magnitude of internal load cause large variability with respect to timing and degree of recovery (Marsden, 1989; Sas, 1989). Nevertheless, a simple targeting scheme that includes the clear objectives (water quality to be restored) and the prerequisites of accomplishing these objectives (permissible loads) is satisfactory to orient both managers and the public. At this early stage, complicated targeting schemes should be avoided since they likely to make confusion.

The situation changes when recovery comes to the stage of re-organization of the aquatic ecosystem. During this stage, the simple targeting system should be replaced with one that integrates specific ecological features of the individual lakes. For example, biomass and composition of zooplankton can be good indicators of recovery in the majority of shallow lakes. In the specific geomorphological environment of Lake Balaton, however, this group fails to properly indicate trophic changes.

As demonstrated in our study, long-term observations must be available about biota in order to choose the most appropriate lake-specific indices and to define a consistent set of ecological criteria. Therefore we recommend including each major group that may a priori be considered to fulfill the role of an indicator in the given lake into the monitoring program from the very beginning of eutrophication management.

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