Combating cyanobacterial proliferation by avoiding or treating inflows with high P load – experiences from 8 case studies

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Sampling and sample analyses of case study lakes

Sampling of **Lago Maggiore** was performed at the deepest point of the lake (370 m). Phytoplankton samples, integrated in the 0-20 m water layers, were gathered fortnightly from March until October, monthly in January, November and December; chemistry samples (0, 5, 10, 20 and 30 meters depth) were collected monthly. Phytoplankton determinations were carried out following Lund et al. (1958). Phytoplankton biovolumes were calculated using the equations provided by Sun & Liu (2003) and Hillebrand et al. (1999). Chemical parameters were determined following the methods reported in Mosello & Ruggiu (1985).

From 1962 to the present, water from Lake Washington has been collected at biweekly intervals using van Dorn samplers at a designated deep water (65 m) station in mid-lake. Grab samples were generally taken at 5 m intervals from 0.5 m to 60 m during periods of stratification and at 10 m intervals during periods of mixing. Temperatures were collected with a bathythermograph from 1962 to approximately 1988 when a Whitney thermistor came into use, followed by YSI and Hydrolab field probes as sampling technologies advanced. Nutrient data has continued to be evaluated using standard laboratory methods throughout the entire period in order to maximize comparability over time. In some years weekly samples were taken at more closely spaced depths through the epilimnion during periods of intense phytoplankton growth and reproduction.

Lake Tegel and Schlachtensee were sampled at their deepest point and samples were usually taken at discrete depth intervals of 0.5 - 2 m and the data depth-integrated to represent the epiliminion; in few cases with homogeneously mixed epilimnion, samples from 1 or 2 m depth were taken to represent the epilimnion. As sampling intervals over time varied between years and seasons (ranging from weekly and monthly), seasonal means were calculated from monthly means. Chl.-a was analysed spectrophotometrically after extraction with boiling ethanol and total phosphorus was analysed after digestion under acidic conditions measuring the molybdenum complex as described in detail previously (Chorus and Schlag 1993).

In **Lake Constance**, the phytoplankton community was sampled within the upper 20 m of the water column using an integrating Schröder sampler (Jochimsen et al. 2013). Average chlorophyll a and total phosprphous concentrations for this depth range were calculated from measurement at distinct depths based on standard protocols (Häse et al. 1988, Rossknecht et al. 1998).

Onondaga Lake was sampled either weekly or biweekly during Jun-Sep at a deep-water site (Matthews et al. 2015). Epilimnetic averages of TP and Chl-a were determined from samples collected at depths of 0 and 3 m. Standard Methods (APHA 1992) were used for laboratory measurements of TP (4500-P E.) and Chl-a (10200 H.2).

Samples from **Lake Balaton** were taken with a tubular water column sampler, i.e. data refer to the whole water column. This applies to all data except the TP between 1980 and 2003, which were estimated values from Istvánovics et al. (2007). The analysis of total phosphorus and phytoplankton are in detail described in Pálffy et al. (2013).

Sampling of the polymictic Barton Broad is based on a surface dip (Philipps et al. 2005).

In-detail results of case studies

Deep, stratified lakes

The deep, stratified, pre-alpine lakes Lake Constance (Germany) and Lago Maggiore (Italy), both oligotrophic by nature, experienced eutrophication with increasing phosphorus (P) concentrations since the 1950s (Table 1 in the main text, Mosello and Ruggiu 1985; Müller 2002). International cooperation for an integrated management of these transboundary lakes founded in 1959 for Lake Constance and in 1973 for Lago Maggiore identified sewage as main nutrient source and launched the

implementation of respective measures (Müller 2002). Beside the ban of phosphorus in detergents at that time, the main measures taken were connecting households to sewerage and upgrading of WWTPs with a tertiary treatment to remove phosphorus.

For Lake Constance, first sewage plants in the catchment were built in 1969, and 10 years later coverage of residential area with sewage plants exceeded 75%. As a consequence, input of dissolved P into the lake declined from a maximum of 1300 tonnes in the 1970s to roughly 200 tonnes in 1996/1997 (Bührer et al. 2000); loads since then remained rather constant. The decrease of in-lake TP concentrations followed this load reduction after some delay. Despite early load reductions in the 1970s winter mixis TP concentrations increased further until a maximum of 87 µg L⁻¹ was reached in 1979. Since then concentrations declined to < 10 µg L⁻¹ in 2001 and further down to 7.6 µg L⁻¹ in 2007 (Jochimsen et al. 2013). The response pattern for seasonal mean epilimnetic TP is similar, showing after an initial lag phase a steady decline down to 11-12 µg L⁻¹ by 1996/1997 (Fig. 1, Fig. 2 in the main text). However, although the TP load was not substantially reduced after 1997, in-lake TP did decline slightly more: since 2000 seasonal means have ranged between 8 and 10 µg L⁻¹ reflecting internal processes leading to P-export from the lake (to the sediment and downstream of the lake).

Phytoplankton biomass (seasonal means for Chl *a*) in Lake Constance, remained unchanged in the range of 5-6 μ g L⁻¹ from 1980 to 1990, even though means for TP declined from 44 to 17 μ g L⁻¹ (Fig. 1, main Text), and the highest seasonal mean cyanobacterial biovolume of 1.7 mm³ L⁻¹ was observed in 1990. As TP declined further, i.e. below 17 μ g L⁻¹ and down to 7-11 μ g L⁻¹, seasonal mean Chl *a* -concentrations followed – plotted against TP for this time period indeed quite linearly and tightly (Fig. 2 in the main text; R² = 0.64 when using depth integrals down to 20 m and an R² of 0.76 for surface data from 1987-2013, Table 1). Cyanobacteria were never dominant in Lake Constance (Sommer et al. 1993) – the only taxa of quantitative relevance were *Aphanizomenon flos-aquae*, *Anabaena spp.*, and *Microcystis spp.*, and at maximum cyanobacteria contributed 15% to the total phytoplankton biovolume during the study period. Seasonal mean cyanobacterial biovolumes greater than 0.7 mm³ L⁻¹ were no longer observed after mean TP-concentrations had declined to less than 15 μ g L⁻¹. Interestingly, since then, i.e. since 1994, yield (i.e. biomass in terms of Chl *a* per unit TP) increased to the range of 0.3-0.45 and sometimes higher, and patterns over time closely follow those of chlorophyll-a, reflecting that a higher share of the lake's available phosphorus was incorporated in phytoplankton and that TP strongly limited the lake's carrying capacity for phytoplankton biomass (Jochimsen et al. 2013). This is also reflected in the concentrations of dissolved inorganic phosphorus (DIP), which also declined down to 3 μ g L⁻¹ when TP dropped below 17 μ g L⁻¹ (IGKB 2009; data not shown here). Overall, the data from Lake Constance confirm that, while high cyanobacterial biomass is not likely at this low end of the trophic range encompassed by the lakes presented here, their biomass and contribution to overall phytoplankton biomass is likely to be higher in the TP-range above 20-45 μ g L⁻¹ and a further reduction of TP can further reduce their biomass.

Lago Maggiore reached a state close to eutrophy in the late 1970s, when the phosphorus loads amounted to 600 t y^{-1} . The eutrophication trend began to be reversed in the early 1980s through upgrading of sewage treatment in the catchment to remove phosphorus (Mosello and Ruggiu 1985). Since that time, the P loads have been gradually reduced, reaching values around 250 t y^{-1} at the end of the 1980s. This load remained stable until 2004, when further measures reduced phosphorus loads below 200 t y^{-1} .

Winter mixing TP data have been used to characterize the response of Lago Maggiore's in-lake TP concentrations (Fig. 1, Fig. 2 in the main text). These have decreased from 30-40 μ g L⁻¹ in the late 1970s (data before 1981 not shown) to <10 μ g L⁻¹ between 1994 and 1999. Between 2000 and 2008 TP concentrations at winter mixing were steadily at 11 μ g L⁻¹, and since then they have shown a slight increase up to 13 μ g L⁻¹. DIP followed the TP-decline quite closely, dropping from 20 to < 10 μ g L⁻¹ (since the mid 1990s) to 7-11 μ g L⁻¹ in the late 1990s (data not shown).

Phytoplankton responded to this decline only after a time lag of several years: while TP at winter mixing was still at or above 15 μ g L⁻¹ seasonal means of Chl a remained in the range of 4.6 – 5.5 μ g L⁻¹, with cyanobacteria constituting 12-65 per cent of total phytoplankton biovolume. Only after TP decreased well below 15 μ g L⁻¹, i.e. to 11 μ g L⁻¹ did phytoplankton show a rather sudden response with: seasonal means now ranged between 3 and 4 μ g L⁻¹ with cyanobacteria contributing only 10-36 per cent of total phytoplankton biovolume. Moreover, as in Lake Constance as TP increasingly limited

carrying capacity for phytoplankton biomass, yield increased sharply, with mean Chl *a*/TP often ranging between 0.4 and 0.75 (higher Chl *a*/TP as compared to Lake Constance is due to TP being winter mixing data rather than seasonal means). While winter mixing TP remained very constant at $11-12 \ \mu g \ L^{-1}$ throughout the first decade of the millennium, phytoplankton biomass increased again starting in 2003, chiefly in response to increasing epilimnetic temperature (Morabito and Manca 2014). Interestingly, in spite of this overall increase the cyanobacteria populations remained minor, contributing 6 - 26 per cent of biovolume.

Cyanobacteria dominated the phytoplankton assemblage of Lago Maggiore only in few summers. During the eutrophication phase and till the end of the 1980's, *Planktothrix rubescens* was the most important species. In the 1990'S, the chroococcales became more important, in particular with *Aphanothece* sp. and *Aphanocapsa* sp. The years after 2005 were characterised by occasional blooms of *Dolichospermum lemmermannii*.

Thus, the trophic state of both lakes returned to near pristine conditions within 15-20 years after restoration began. Their recovery time seems quite long, particularly as pre-restoration TP concentrations were not very high. However, this can be attributed to their large volume of water and relatively low water exchange rate: while for both lakes it has been given as only once in about 4 years (Tab. 1), for Lago Maggiore Ambrosetti et al. (2012) point out that the real renewal time of the whole water mass would be significantly longer that 4 years – while the epilimnetic waters can be exchanged faster, exchange of the hypolimnion with its high concentrations of phosphorus is much slower.

Lake Washington (USA) is one of the earliest examples of lake restoration by diversion of sewage (Table 1, Edmondson 1970). Phosphorus concentrations in the lake increased (Fig. 1 in the main text) after the input of the effluent of a secondary biological treatment plant during the 1950s with sewage effluent contributing 56% of the lake's phosphorus in 1956 (Edmondson 1970) and up to 72.4 per cent in 1962 (Edmondson and Lehmann 1981). This was followed by increases in phytoplankton biomass and the proportion of cyanobacteria: Edmondson and Anderson (1956) report that *Planktothrix rubescens* occurred "in great quantity … during the spring and summer of 1955, probably for the first time" with biovolumes in the range of 2.8 mm³ ml⁻¹. In contrast, previously

cyanobacterial populations were minimal (in 1950 biomass corresponded to 0.14 mm³ L⁻¹ biovolume) and involved *Dolichospermum lemmermannii*, *Planktothrix rubescens* and *Phormidium spp*.

Increased public awareness and concern about the lake water quality led to the stepwise diversion of the effluent from 1963 to 1968, by which year 99 per cent of the sewage was diverted (Edmondson 1970). In spite of a moderate retention time of 2.4 years (see Table 1 in the main text), Lake Washington reacted quickly and sensitively to the reduced nutrient supply, with seasonal means for TP declining from 64 μ g L⁻¹ in 1963 to 15 μ g L⁻¹ by 1973 and to 8-12 μ g L⁻¹ by the mid 1980's. In face of the fact that sewage diversion reduced the total phosphorus load by only about one half this reaction of in-lake concentrations is surprisingly strong. Edmondson (1970) explains this with the high concentration of phosphorus in sewage -200-fold above that of the influent streams which diffuse their load across a large volume of lake water. Another important factor is that little phosphorus is recycled internally from sediments in the lake, as oxygenation of the deep water remains strong even during thermal stratification and does not promote release (Edmondson and Lehman 1981). The lesson from this strong decline is that concentration patterns from inflows and the proportion of external vs internal inputs matter for the availability of phosphorus that drives the proliferation of algae and cyanobacteria. Nearby Lake Sammamish in the same watershed, with less water volume and a hypolimnion that becomes anoxic every summer, was slower to respond to sewage diversion (Welch et al 1986)

Phytoplankton biomass followed the decline of TP very closely (Fig. 1, Fig. 2 in the main text) with a regression coefficient between seasonal means of Chl *a* and TP of 0.91, and by 1973 at a mean TP-concentration of $15 \ \mu g \ L^{-1}$ the biovolume of cyanobacteria had dropped below 0.3 mm³ L⁻¹ (Table 1, Fig. 1 in the main text). Interestingly, while the biomass both of total phytoplankton and of cyanobacteria remained low, the relative contribution of cyanobacteria increased again in some years, i.e. up to a seasonal mean of 29% in 1988 and 19% in 1995. However, this was no longer due to dominance of *Planktothrix rubescens*; rather for the period between 1995 and 2000 Arhonditsis et al. (2003) list 5 genera comprising over 95% of the Cyanobacteria – i.e. chiefly *Anabaena* and *Aphanizomenon but also Microcystis, Anacystis, Chroococcus* and *Planktothrix (c.f. limnetica). Woronchinia* and *Aphanothece* were also frequently observed (Edmondson et al. 2003).

In spite of the very low biovolumes since 1973 microcystins have been detected at a concentration of up to 417 μ g L⁻¹ at a downwind shoreline where wave action accumulates *Microcystis* and *Anabaena* as well as minor amounts of *Aphanizomenon* (Bartley Abella, unpublished data). Due to the large size of the lake, these buoyant colonies recruit themselves from a very large water volume and thus can transiently attain sizable scums, even with microcystin concentrations considered unsafe for recreational use, in particular where small children and dogs play in such shallow shoreline areas.

The overall lesson learned from the recovery of Lake Washington is that cyanobacteria could be strongly and sustainably reduced to very low levels of biomass (Fig. 1, Fig. 2 in the main text) once TP-concentrations declined below a threshold of 20-25 μ g L⁻¹. This also led to a switch in species composition away from the previously dominant *Planktothrix rubescens* and towards other cyanobacterial taxa. While this does not totally rule out the occurrence of toxic scums, these now appear to be restricted to limited downwind and protected areas along the shoreline.

Smaller stratified lakes

The water quality of **Onondaga Lake**, New York (USA) was severely deteriorated due to a WWTP discharge amounting to 25% of total inflow to the lake and > 60% of the TP load. Summer average concentrations of TP exceeded 1000 μ g L⁻¹ in the early 1970s and continued to exceed 100 μ g L⁻¹ through the late 1980s causing dense summer blooms of cyanobacteria, oxygen depletion, and high turbidity with Secchi depths often below 1.2 m (Effler et al. 2013). A ban on high phosphorus content in detergents in 1971, repeated upgrading of phosphorus treatment (secondary in 1979, tertiary with calcium in 1981, tertiary with ferrous sulfate in 1986), and refinements of the treatment process led to a reduction of the in-lake TP concentration down to levels of around 40 μ g L⁻¹ TP (summer mean) by 2000 (Fig.1 in the main text, Effler et al. 2013). Summer means of Chl *a* concentrations decreased from 40-60 μ g L⁻¹ in the 1980s to 20 μ g L⁻¹ by the year 2000 (Fig. 1 in the main text; see also Matthews et al. 2015). Since 2005 phosphorus in the WWTP effluent was further decreased below 100 μ g L⁻¹ by an advanced flocculation/settling technique. Maintenance of high nitrate concentrations (>1.0 mg N L⁻¹) in the hypolimnion has nearly eliminated the flux of phosphorus and methylmercury

from profundal sediments (Matthews et al. 2013). As a result of these measures in-lake summer average concentrations of total phosphorus decreased to 15-25 μ g L⁻¹ over the 2007-2014 interval. Interannual variations in total phosphorus concentrations since 2007 have been attributed to year-to-year variability in food web effects (e.g., *Daphnia* grazing) and tributary loading (Matthews et al. 2015).

During 1967-1971, when the lake was highly eutrophic, nuisance forms of cyanobacteria (e.g., *Microcystis, Aphanizomenon, Anabaena*) dominated the phytoplankton community in summer (Sze and Kingsbury 1972). Cyanobacteria were largely replaced by chlorococealean green algae during the 1972–1986, possibly in response to a ban on detergents with high phosphorus content and an increase in the N/P ratio (Sze 1975, 1980; Effler 1996). Nuisance filamentous cyanobacteria returned as a dominant component of the summer phytoplankton community in 1987, driven by the selective feeding of *Daphnia* (Siegfried et al. 1996). As concentrations of TP first declined below 60 μ g L⁻¹ in 1997, the yield of phytoplankton biomass (as Chl *a*/TP) increased, showing that carrying capacity started to be limited by phosphorus. A shift to phosphorus limitation of phytoplankton growth is also reflected in the relationship of the seasonal means for Chl *a* and TP, which shows considerable scatter prior to 1996 but a tight coupling (R² = 0.71) after 1996 (Fig. 2 in the main text).

Data for cyanobacterial biomass, available since 1998, show that cyanobacteria remained abundant during summer through 2004. Mean growing season concentrations of cyanobacteria decreased rapidly after seasonal mean TP had dropped from 59 μ g L⁻¹ in 2004 to 34 μ g L⁻¹ in 2005. Interestingly, at summer mean TP-concentrations below 40 μ g L⁻¹ cyanobacteria became negligible in Onondaga Lake, though *Aphanizomenon, Aphanocapsa, Pseudanabaena*, and *Cuspidothrix* remain present at low concentrations in late summer and early fall. Trophic state conditions in Onondaga Lake are now consistent with upper mesotrophy, with seasonal mean concentrations of chlorophyll-a in the range 6 – 7 μ g L⁻¹ (Fig. 1 in the main text).

Schlachtensee and Lake Tegel (Germany) were both highly eutrophic with TP concentrations up to $600-800 \ \mu g \ L^{-1}$ before phosphorus-stripping in the main inflow started. Both lakes are largely fed by one tributary, and phosphorus-stripping from it reduced concentration in the inflows from 300-900

 μ g/l to around 20 μ g L⁻¹ at Lake Tegel and from 400-600 μ g L⁻¹ to 8-10 μ g L⁻¹ at Schlachtensee (Schauser and Chorus 2007) while maintaining the water exchange rates of the lakes. This drastic and sudden 30-60-fold load reduction was decisive for the further trophic development of both lakes.

In-lake TP concentrations declined exponentially in Schlachtensee reaching summer means below 40 μ g L⁻¹ in 1985, four years after restoration had begun, and since 2000 they have ranged between 12 and 18 μ g L⁻¹ (Fig. 1, Fig. 2 in the main text).

In Lake Tegel TP means took longer to decline below 40 μ g L⁻¹, i.e. until 1994, 8 years after the phosphorus-stripping plant went into operation. This slower recovery was largely due to the influence of untreated water from the Havel River, the extent of which depended on the discharge of the phosphorus stripping plant. From 1997 to 2001 this discharge was reduced to economise operational costs, and this increased the proportion of untreated river water in the lake, causing TP concentrations to increase again up to seasonal means of 64 μ g L⁻¹. As in consequence cyanobacteria increased again, the operation regime of the stripping plant was optimized to guarantee sufficient effluent during the summer, and thus seasonal mean TP concentrations decreased to around 30 μ g L⁻¹ by 2007 (Fig. 1, Fig. 2 in the main text).

Both lakes had anoxic hypolimnion, and measures were taken to address P-release from the sediments – hypolimnetic aeration in Lake Tegel throughout the summers of 1979-1989 and withdrawal of the hypolimnion of Schlachtensee in late summer (1981-1996). However, retrospective analysis by Schauser and Chorus (2007) revealed the impact of these measures on the phosphorus budget of these lakes to be minor: in Lake Tegel mineralization (at elevated temperatures caused by partial mixing through aeration) proved the main driver of P-release from the sediment, and this decreased as sedimentation of organic matter declined in consequence of reduced phytoplankton biomass. At Schlachtensee, though release rates from the sediment were high, the volume of the hypolimnion is small in relation to the total lake volume and thus withdrawal removed only a minor fraction. Both lakes have rather high water exchange rates of 2-5 times per year, and within 10 to 15 years after restoration begun, their sediments became sinks rather than sources of phosphorus (Schauser and Chorus 2009).

Before the onset of the phosphorus elimination both lakes experienced recurrent heavy cyanobacterial blooms with dominance of *Limnothrix*, *Planktothrix agardhii* and *Aphanizomenon* in Schlachtensee and *Microcystis* (at times with *Aphanizomenon*) in Lake Tegel. Phytoplankton in Schlachtensee responded to the reduced TP-concentration already after 4 years: below a threshold TP concentration of 30-40 μ g L⁻¹ cyanobacteria the formerly often dominant *Planktothrix agardhii* largely disappeared and if cyanobacteria occurred, these were scum-forming species such as *Microcystis* and *Anabaena*, though only in minor amounts (Chorus and Schlag 1993; Chorus and Niesel 2011). These decreased yet further after 2003: with summer mean TP concentrations ranging below 15 μ g L⁻¹ maximal cyanobacterial biovolumes have not exceeded 0.7 mm³ L⁻¹. Although these were so low, cyanobacteria contributed up to a mean of 20-30 per cent of total biomass in a number of summers since TP had declined.

In the deeper and more turbulently mixed Lake Tegel cyanobacteria did not dominate phytoplankton with almost 100 percent of biovolume as in Schlachtensee (they usually co-occurred with diatoms). Blooms also disappeared once summer TP concentrations dropped below 30-40 μ g L⁻¹, first in 1994-1996 and again after 2002. Between these phases the lake experienced several years of "re-eutrophication" in consequence of an increase of the TP-concentrations (back to means of 42-64 μ g L⁻¹) due to reduced stripping plant operation, and during these years both the absolute biomass of cyanobacteria as well as their per cent contribution to phytoplankton biomass increased again. Interestingly, they did not reach the same levels as before 1994 at similar TP-means (previously at >40 μ g L⁻¹ mean TP their seasonal mean biovolumes were at 3-4 mm³ L⁻¹; after 1996 they only reached 0.1 – 0.6 mm³ L⁻¹). This experience highlights the importance of keeping TP below a threshold in order to prevent blooms. Since 2003 cyanobacterial biovolumes have been below < 0.2 mm³ L⁻¹ with *Microcystis* and *Anabaena* as prevailing genera. In both lakes, microcystins can still be found together with these cyanobacteria, but concentrations are usually well below 1 μ g L⁻¹.

Shallow, polymictic lakes

Lake Balaton (Hungary) is a large, shallow lake, and often very turbid due to pronounced resuspension of calcareous sediment (Istvánovics et al. 2007). There is an inherent spatial heterogeneity in the lake, with decreasing trophic conditions from west to east (Padisák and Reynolds 1998). This trophic gradient is caused by the river Zala, the major inflow and an important source of nutrients entering the lake at its westernmost part. Due to the restoration efforts to eliminate excessive nutrient loading, including the rehabilitation of a reservoir (Kis-Balaton) at the major inflow in 1985, the adoption of P-removal in wastewater treatment in 1991, sewage diversion (Somlyódy and van Straten 1986), the connection of further households to the sewer system and a reduction of fertilizer application in agriculture after 1989, total external P loads reaching the whole lake were reduced by about half, i.e. from 289 ± 42 tons of P per year (i.e. $1.8 \text{ tons m}^2 a^{-1}$) in the 1980's to 145 ± 47 tons per year (i.e. $0.9 \text{ tons m}^2 a^{-1}$) in the time span 1989-2002. During 1990-1993 and 2000-2003 periods of drought further reduced diffuse loading from the catchment (Istvànovics et al. 2007).

Data on TP in Lake Balaton between 1980 and 2003 are estimated values (Istvánovics et al. 2007), data after 2003 are measured values from the MTA Centre for Ecological Research, Balaton Limnological Institute. All data on chlorophyll-a and cyanobacterial abundance are from measurements by the co-authors.

Lake Balaton responded with a decline in mean seasonal TP-concentrations from levels around $100 \ \mu g \ L^{-1}$ to 30-50 $\ \mu g \ L^{-1}$ (Fig. 1 in the main text). This decline followed the load decline depicted in Istvánovics et al. (2007) quite closely. In-lake concentrations increased only slightly in response to peak loads re-occurring particularly in 1996, 1998 and 1999.

From the 1970s to the mid-1990s large summer blooms of nitrogen-fixing cyanobacteria (Vörös and Nagy Göde 1993), mainly *Aphanizomenon flos-aquae*, *Cylindrospermopsis raciborskii* and *Anabaena* spp. were typical for Lake Balaton. Non-nitrogen-fixing *Planktothrix agardhii* also reached high density. Seasonal means of chlorophyll-a substantially responded to the reduction in TP only after 1994, and this decline was not closely coupled to the decline in the seasonal means of the TP-concentrations (see Fig. 2 in the main text) in the western basin; R² for Chl.-a against TP was only 0.31, Table 1). Mean seasonal biovolumes of cyanobacteria were no longer above 5 mm³ L⁻¹ since mean seasonal TP concentrations were no longer above the range of 50 µg L⁻¹. Furthermore,

Istvánovics et al. (2007) showed that after 1994 *Cylindrospermopsis raciborskii*, the dominant cyanobacterium species during the summer blooms, no longer contributed more than 30% of total phytoplankton biomass and the number of days that it occurred with high biomass (i.e. biovolumes > $10 \text{ mm}^3 \text{ L}^{-1}$) markedly declined. Since the mass blooms of nitrogen-fixing cyanobacteria were highly favoured by N limitation as a result of the higher P loads experienced in previous decades (Présing et al., 2008), the reduction in the latter caused an apparently decreasing trend in their summer biomass, as seen in the boxplot of Fig 1.

Reduction of the phosphorus load to **Barton Broad** (United Kingdom) began in the 1977 when phosphorus control was introduced in sewage treatment plants discharging to the River Ant which feeds the Broad. By 1980 further measures had been introduced for other effluents, either P removal or effluent diversion, and by the year 2000, the TP-load was reduced by more than half, i.e. from over 10 g per m² and year to less than 5 g per m² and year (Phillips et al. 2005).

A pronounced decline of the TP concentrations in Barton Broad followed the load reduction quite closely until 1987, but concentrations showed some increase during the late 1980's and early 1990's (Lau and Lane 2002). Sediment dredging was undertaken between 1997 and 2000 to reduce internal loading, and this coincided with further improvement of P removal in sewage treatment. Since 2000, the lake's TP means for May-October ranged between 50 and 70 μ g L⁻¹.

Although Barton Broad has remained turbid and macrophytes have not yet repopulated this shallow lake, phytoplankton biomass declined considerably: Seasonal means of chlorophyll-a closely followed the TP-decline (Fig 1; R² over the time span from 1974 to 2011 = 0.61, Table 1). However, though mean concentrations of chlorophyll-a no longer ranged above $100 \ \mu g \ L^{-1}$, means of 20-40 $\ \mu g \ L^{-1}$ since the year 2000 are still high, and with Secchi depths between 0.4 and 0.7 m the lake remains turbid. Also, in contrast to the stratified lakes discussed above which showed much stronger reductions of TP and phytoplankton biomass, in Barton Broad yield (i.e. Chl *a*/TP) has not increased, reflecting that the concentrations of TP are still too high to impose very pronounced nutrient limitation. Hunter et al. (2008) report a marked decrease in cyanobacterial abundance, particularly filamentous and N-fixing taxa (*Anabaena* and *Aphanizomenon*; see also Phillips et al. 2005), while in 2005 and 2006 a

bloom of *Microcystis aeruginosa* occurred. This decline is also reflected in a decline of the per cent contribution of cyanobacteria to overall phytoplankton cell counts by the year 2000 (Fig. 2). The shift in cyanobacterial species composition may be explained by surplus N-availability as TP and biomass declined due to the onsetting P-limitation. It also shows that TP-levels between 50 and 70 μ g L⁻¹ are still too high to induce a marked switch in species composition disfavoring cyanobacteria.

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Tables and Figures

Table 1: Linear regressions of seasonal mean (May-Oct) concentrations of Chlorophyll-a against those of total phosphorus (TP) (epilimnion or near-surface depth) excluding data at high TP-concentrations where curves in Fig. 2 in the main text level off

Lake	TP ranges excluded	Slope	Regression Coefficient
Lake Constance	$> 22 \ \mu g/L$	0.19	0.64
Lago Maggiore		0.13	0.3739
Lake Washington		0.59	0.9096
Onondaga Lake	$> 70 \ \mu g/L$	0.36	0.5750 and 0.71 for 1996-2011
Lake Tegel	$> 100 \ \mu g/L$	0.53	0.5689
Schlachtensee	$>75 \ \mu g/L$	1.22	0.8835
Barton Broad		0.52	0.6084
Balaton Basin 1		0.26	0.3069



Fig. 1. Boxplot of the summer (July-September) biovolume of N_2 -fixing cyanobacteria in the western basin of Lake Balaton during the periods 1983-1990 and 1999-2013 (data for 2007 is not available).



Fig. 2: Percent contribution of cyanobacteria to the total phytoplankton cell counts in Barton Broad.