Rönicke, H., **Frassl, M.A.**, **Rinke, K.**, **Tittel, J.**, **Beyer, M.**, Kormann, B., Gohr, F., **Schultze, M.** (2021):

Suppression of bloom-forming colonial cyanobacteria by phosphate precipitation: A 30 years case study in Lake Barleber (Germany)

Ecol. Eng. **162** , art. 106171

Suppression of bloom-forming colonial cyanobacteria by phosphate precipitation: a 30 years case study in Lake Barleber (Germany)

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Abstract

Although the treatment of eutrophied lakes with aluminium (Al) compounds has been established for more than 40 years, publications reporting on long-term consequences for phytoplankton are rare. Here we present observations from Lake Barleber for the period 1985-2016. The lake was treated in autumn 1986 because of high phosphorus (P) concentrations and cyanobacteria blooms, which limited the lake's recreational use. Within six weeks 480 t of Al sulphate solution (37 t of Al) were applied to the lake. This was equivalent to a dose of 36 g Al^{3+} m^{-2} or 5.7 mg Al^{3+} L^{-1}. Already after having applied half of the Al sulphate, the concentration of soluble reactive phosphorus (SRP) reached its analytical limit of quantification (3 µg L^{-1}). Removal rates calculated after completion of the treatment were 98% for SRP and 90% for total phosphorus (TP). In the following 13 years from 1987 to 1999, cyanobacteria were almost absent. In the years 2000 to 2003 as well as in 2005 and 2014 they appeared in low abundances. In the period 1987-2014, almost complete absence of cyanobacteria and high transparency provided good conditions for recreational use of Lake Barleber.
Compared to pre-treatment conditions, phytoplankton biomass increased temporarily from 1987-2016. This increase in biomass did not interfere with the use for bathing and swimming, because phytoplankton community composition changed towards a dominance of chlorophytes and dinophytes. In 2016, however, P concentration and cyanobacterial biomass rose again to the level of the last pre-treatment years (TP 134 µg L^{-1}, cyanobacterial biomass 1 mg L^{-1}; averages for the period May-October). We conclude that Al treatment is effective and can last for decades. For recreational lake

use, the effects of the alum treatment on phytoplankton community composition showed to be more important than its effects on total phytoplankton biomass.

Dedication statement

This paper is dedicated to Prof. Dr. Helmut Klapper (June 2, 1932 to June 19, 2019). Until his retirement in 1997, he was the leader of the team that designed the treatment of Lake Barleber in 1986 and that did the accompanying monitoring.

Key words: eutrophication, diazotrophic cyanobacteria, phytoplankton composition, lake management, aluminium application, cyanoHAB

1 Introduction

Growing incidences of cyanobacterial blooms in lakes worldwide place a serious burden on water resources (Pearl and Scott, 2010; Ibelings et al., 2016a). Cyanobacterial blooms cause a range of ecological, economic and health effects (Huisman et al., 2005; Sharma et al., 2013), which can lead to serious problems, especially in water bodies used primarily for drinking water supply or recreational purposes. The negative side effects of such blooms can include the formation of algal mats, taste and odour compounds as well as the production of cyanotoxins, which can have far-reaching effects on aquatic and terrestrial organisms (Chorus and Bartram, 1999; Backer et al., 2015). Moreover, if water bodies are used primarily for bathing, direct exposure to algal toxins poses a health risk to the public (Chorus et al., 1999). Anthropogenic eutrophication is considered to be the main reason for the increase in cyanobacterial blooms (Carpenter, 2008; Watson et al., 2008).

The addition of aluminium (Al), which permanently binds P and removes it by sedimentation, is recognized as one of the most effective strategies for controlling internal P loads (Cooke et al. 1982, 2005; Klapper, 1991; Wagner 2017). In P-rich lakes, the addition of Al can reduce the internal P loading and thereby limit primary production and suppress toxic cyanobacterial blooms (Cooke et al. 2005). The addition of Al sulphate ($Al_2(SO4)_3$; referred to as alum) to the water body rapidly leads to hydrolysis and the formation of amorphous flocs of $Al(OH)_3$, which exhibit a high adsorption affinity for P (Huang et al., 2002). They react with PO_4^{3-} to form insoluble $AlPO_4$ (Nogaro et al., 2013), whereby a high binding capacity for inorganic P is reached at a pH of 6 to 8. In addition to reducing the P concentration, the application of Al sulphate also reduces cyanobacterial blooms by flocculation (Cooke et al., 2005; Brattebo et al., 2017).

Since the early 1970s, hundreds of lakes have been treated with Al (Landner, 1970; Cooke et al. 2005; Jensen et al., 2015; Huser et al., 2016). Most of those treatments have been carried out in the USA. In addition to P precipitation in the pelagic zone, the formed Al hydroxide can also increase P-retention in the sediment once the flocs have settled to the sediment (Lewandowski et al., 2003; Huser et al., 2011). According to Cook et al. (2005) an increase of the P retention capacity of the sediment even is the main goal of P inactivation when using alum. Unless overcritical external inputs or burial of new sediments interfere with the sustained adsorption of P at the sediment-water interface, alum treatment thereby can lead to a long-lasting improvement of water quality (Lewandowski et al., 2003; Huser et al., 2016). Recently, Huser et al. (2016) reviewed cases from 114 lakes of different morphology to determine key parameters for a long-term success of alum treatment. They analysed treatment results with respect to the long-term effects on total P concentration (TP), water clarity (Secchi depth) and phytoplankton biomass (chlorophyll *a*). Decisive criteria for long-term success were: the applied Al dose, the watershed-to-lake area ratio, and the morphology of the lakes. The duration of success varied widely, ranging from a few months to more than 30 years (Welch and Cooke, 1999; Huser et al., 2016).

Lake Barleber (Germany) was treated with alum (autumn 1986; Rönicke and Bahr, 1989; Klapper and Geller, 2001) in order to re-establish conditions allowing for safe recreational use after years of increasing dominance of N_2-fixing cyanobacteria (*Anabaena*, *Aphanizomenon*; Fig. 1). The reduction of the P concentration lasted until 2014 and for almost 30 years the occurrence of cyanobacteria could be limited to a level acceptable for recreational use. Although reviews, individual reports and even three special issues of journals dealing with alum treatment of lakes have been published in recent years (Zamparas and Zacharias, 2014; Lürling et al., 2016; Huser et al., 2016; Ibelings et al., 2016a,b; Wagner, 2017), long-term studies including the response of the plankton have rarely been reported. By presenting results from the long-term monitoring (1985-2016) of Lake Barleber (Germany), we intend to contribute to fill this gap. We further discuss the criteria for evaluating the success of P precipitation with particular emphasis on the role of phytoplankton community composition.

2 Methods

2.1 Study Site

Lake Barleber is located in the north of the city of Magdeburg, Germany (Table 1). It is a gravel pit lake, created mainly between 1929 and 1931 (Bauch, 1953). After excavation, it filled rapidly with groundwater. The lake does not have any surface inflow nor outflow, but groundwater is flowing through the lake (ca. 640,000 m^3a^{-1} inflow, ca. 530,000 m^3a^{-1} outflow; Hannappel and Strom, 2020). The surrounding area is used for agriculture, i.e., mainly the cultivation of grains and root vegetables. The most important morphometric parameters are presented in Table 1. Fig. S1 (see Supplementary Material) provides a bathymetric map of Lake Barleber.

Around Magdeburg, Lake Barleber is the most heavily used lake for recreation. It is fully developed for tourism and mainly used for bathing, swimming, diving and angling. Since the 1970s, a large camping site, an anglers' allotment and a main beach for bathing have been established.

Lake Barleber is monomictic, with stable thermal stratification during summer (Fig. S2 in Supplementary Material). Only exceptionally, in case of complete ice cover, the lake is stratified inversely in winter. The shallow depth of Lake Barleber results in a very small hypolimnion, with a waterbody volume of only 117,000 m^3 below 7 m depth, i.e. 1.5% of the total volume.

Soluble reactive phosphorus (SRP) has been measured in Lake Barleber since the 1950s (Fig. 1; Bauch 1953). Until 1962, SRP concentrations remained below the analytical detection limit of 3 µg L^{-1}. The lake was in clear-water stage with extensive underwater grasslands (*Chara* and *Elodea* populations; Bauch 1953). From the mid-1960s to the mid-1970s, SRP concentrations steadily increased up to 50 µg L^{-1}. This increase in SRP concentrations was linked to the onset of turbidity as a result of the increased presence of planktonic algae (greens and diatoms). In the following decade (1975 to 1985), a rapid increase in the SRP concentration was observed which, in the mid-80s, led to a concentration above 150 µg L^{-1}. In the same period, planktonic cyanobacteria appeared during the summer months, culminating in cyanobacteria mass developments and a high turbidity of the water body (Rönicke and Bahr, 1989).

To reduce the internal P loading sufficiently enough to inhibit summer cyanobacterial blooms, alum was applied to Lake Barleber in autumn 1986 (for details see Table 2). Within a period of six weeks, the alum treatment was accomplished by surface spraying (Rönicke and Bahr, 1989; Klapper and Geller, 2001). Since there were no surface inflows and no known inflows of waste water, a reduction of the external phosphorus load was not possible.

2.2 Sampling and analytical methods

Sampling of Lake Barleber was done at the deepest point of the lake at intervals of 2-3 weeks during the vegetation period (May to October) from 1986 to 2000. Since 2001, sampling was done at monthly intervals. Until 1992, water samples were taken with a Ruttner sampler; from 1993 onwards, a Limnos sampler (LIMNOS, Finland) was used. In general, water samples were taken at depths of 0, 1, 2.5, 5 and 7.5 m and the analytical data were averaged over the water column. To examine the water-sediment contact zone (9 m), a specially developed sediment corer (Rönicke and Bahr 1990) that enabled the collection of water samples right above the sediment was used. The chemical parameters Al, sulphate, soluble reactive phosphorus (SRP), total phosphorus (TP) and dissolved inorganic nitrogen (DIN; calculated by summing up of nitrogen analysed as nitrate, nitrite and ammonia) were analysed as described in Herzsprung et al. (2005, 2006). We calculated DIN:TP ratios according to Dolman et al. (2016), who identified this ratio as most informative with respect to nutrient limitation of primary production. They defined a critical value of 1.6, with a DIN:TP ratio of < 1.6

corresponding to N limitation and a DIN:TP ratio of > 1.6 to P limitation. Chlorophyll-*a* concentration was analysed photometrically according to the German standard DIN 39412. Qualitative and quantitative determination of the phytoplankton and ciliate abundance was done by microscopic analysis following Utermöhl (1958), after fixation with Lugol's solution. Cell sizes were measured and their biovolume calculated based on suitable geometric shapes. The biomass of the individual species was determined under the assumption of a specific density of 1 g cm^{-3} (Premazzi, 1980; Benndorf et al., 1983). Zooplankton samples were taken by vertical hauls from 7.5 m depth to the surface using a conical plankton net (opening diameter 25 cm, length 0.5 m, mesh size 55 µm, corrected for filtering efficiency). These samples were also fixed with Lugol's solution and species were identified and quantified under the microscope. Rotifer volume was calculated according to Ruttner-Kolisko (1977). Crustacean biomass was calculated according to published length-weight relationships assuming that dry weight equals 15% of wet weight.

Macrophytes were investigated four times during the study period (July 23, 2007; August 26, 2010; June 29, 2014; July 13, 2016) following the requirements of the EU Water Framework Directive (WFD, Guideline 2000/60/EG) by contractors of Landesbetrieb für Hochwasserschutz und Wasserwirtschaft Sachsen-Anhalt. A semi-quantitative methodology was appied following the German assessment standard that is based on Schaumburg et al. (2004) and Stelzer et al. (2005).

First, a survey of the occurrence of macrophytes, macrophyte species and their estimated abundance was done (according to a five degree scale: 1 - very rare; 2 - rare;

3 - common; 4 - frequent; 5 - abundant, predominant). Occurrence was mapped along transects (four in 2007 and 5 during the other years) and separated by depth zones 0-1 m, 1-2 m, 2-4 m and >4 m according to Melzer (1999). Abundances were converted into plant quantities using the function $y = x^3$ (Melzer 1999). From these, the reference index and the class of ecological potential were determined following Schaumburg et al. (2004) (the five status classes are: 1 - very good, 2 - good; 3 - moderate; 4 - poor and 5 – bad).

In order to visualize changes in the abundance of the dominating species (abundance degrees 5-3), the calculated plant quantities of all transects were averaged per depth zone and sampling date.

3 Results

3.1 Effects of alum addition on P, Al, and sulphate concentrations

At the start of the alum treatment at 3rd of October 1986, the SRP concentration in the pelagic zone was 180 µg L^{-1} and the TP concentration was 190 µg L^{-1} indicating that 95% of the P was present in soluble, mineralized form. By October 22nd after approximately half of the total precipitant quantity had been applied, the detection limit for SRP (3 µg L^{-1}) was reached in the pelagic zone (Fig. 2). The remaining 50% of the Al sulphate solution was applied as excess precipitant to condition the sediment with $Al(OH)_3$ flocs in order to enhance the P binding capacity. After having completed the precipitation, 98% of SRP had been removed and TP had been reduced by over

90%. The mean SRP concentrations in the subsequent years until 2014 fluctuated between 6 µg L^{-1} (1989) and 15 µg L^{-1} (1995), and the mean TP concentration ranged between 24 µg L^{-1} (1997) and 54 µg L^{-1} (1998) (Fig. 3). In 2016, however, a substantial increase in P was observed (SRP 86 µg L^{-1}, TP 134 µg L^{-1}). Concentrations of Al and sulphate increased during the treatment, but reached similar levels to the ones observed before the treatment by the end of the year 1986 (Table 3).

3.2 Phosphorus accumulation in the bottom layer

In the summer of 1986, before the alum was applied, a massive release of dissolved P from the lake sediment was detected (Fig. 4). Already in Arpril, a rapid decline in oxygen concentrations occurred near the bottom of the lake, and led to anaerobic conditions during the summer months. This oxygen decline was accompanied by a marked increase in SRP: in mid-April (16 Apr) 1986 the SRP concentration was 49 µg L^{-1} and at the end of September (30 Sept) a concentration of almost 1 mg L^{-1} (980 µg L^{-1}) was recorded. These SRP concentrations at the lake bottom were several times higher than those measured in the surface water (0.18 mg L^{-1}) indicating a substantial release of P from the phosphate-rich sediment during the summer stratification period.

In the years after the alum treatment, even during periods of low oxygen levels, the release of P was almost entirely inhibited and SRP concentration remained low (Fig. 4). From 1987 to 2014, no or only very low P release rates from the sediments were recorded (Fig. 4). In late September 2016, however, 1.15 mg L^{-1} SRP was measured above the sediment indicating significant P release from the sediment.

3.3 N:P ratios

During the summer period of 1986 (June-August), before the phosphate precipitation took place, the mean mass-related DIN:TP ratios remained below the critical value of 1.6, indicating N-limitation of the phytoplankton (Fig. 5). After the application of alum, DIN:TP ratios increased as a result of the reduction in the pelagic P concentration. In the following years 1987 to 2014, DIN:TP ratios were mostly above the critical value of 1.6, indicating P-limitation. In 1987, the value even rose to 5.3. In summer 2016, the DIN:TP ratio again dropped below the critical value of 1.6.

3.4 Secchi disc transparency and chlorophyll-*a* concentration

Bi-weekly to monthly Secchi depths measurements were averaged over the six-month vegetation period. The mean Secchi depths in 1985 and 1986 prior to application of the precipitant, were 1.3 m and 2 m, respectively (Fig. 6). During the summer months and the mass development of diazotrophic cyanobacteria, the values dropped to well below 1 m. In the years after the application of the precipitant, Secchi depths increased. The highest mean values were calculated for the years 1998, 1999, 2002 and 2013 at approximately 6 m. Even individual values ≥8 m were observed: 8.6 m (15 June 1994), 8 m (19 June 2000), 8.5 m (20 June 2013) and 9.0 m (9 June 2016). In addition to the general increase in Secchi depths, also the duration of periods of high transparency (> 5 m) increased. It continually lasted over several weeks in the posttreatment years. In 1994, for instance, the period of high Secchi depths lasted over two months from 15 June until 24 August 1994. In August 2016, however, Secchi depths decreased to 0.5 m again, caused by an algal mass development.

In the six-month vegetation period of the years 1985 to 2016, mean chlorophyll-*a* concentrations overall remained at a low level (Fig. S1). Significant fluctuations occurred during the study period as a whole. The lowest recorded chlorophyll-*a* concentration was 2.0 $\mu g\ L^{-1}$ in 1998. The highest annual average (10.1 $\mu g\ L^{-1}$) was measured in 2001. The highest single concentration peak during the years 2000 and 2001 corresponded with the mass occurrence of *Ceratium hirundinella* (dinophytes) (see below).

3.5 Development of phytoplankton

In the years 1985 and 1986, the summer phytoplankton community was largely characterized by diazotrophic cyanobacteria (Fig. 6, Fig. S2). The dominant species were *Anabaena flos-aquae, A. lemmermannii* and *A. circinalis*. The genus *Aphanizomenon* was represented by the species *Aphanizomenon flos-aquae* and *A. gracile*. The average percentage contribution of cyanobacteria to the total phytoplankton biomass was 45% in 1985 and 58% in 1986. In the summer months June to August, cyanobacteria dominated the phytoplankton community with maximum percentages of 96% (11 July 1985, mean 85%) and 98% (09 July 1986, mean 88%). These mass developments led to the formation of extensive floating algal mats. In the subsequent 13 years after P precipitation (autumn 1986), N_2-fixing cyanobacteria were observed rarely. The low abundances of cyanobacteria detected in autumn 1987 and spring 1988 were composed by the species *Limnothrix redekei* and *Microcystis aeruginosa*. Diazotrophic cyanobacteria (*Anabaena* and *Aphanizomenon* species) were first detected again in the years 2000, 2002, 2003 and 2005. However, their percentage contribution to the total phytoplankton was significantly lower than measured in 1985 and 1986. Dunig the six-month vegetation period they accounted for 6% in 2000, 13% in 2002, 15% in 2003 and 3% in 2005. Besides cyanobacteria, over the entire study period dominant representatives of chrysophytes (*Kephyrion spec., Ochromonas spec., Chrysochromulina parva, Dinobryon divergens*), greens (*Chlamydomonas spec., Ankistrodesmus angustus, A. falcatus, Scenedesmus longispina*, *S. quadricauda, Kirchneriella obesa, Crucigenia rectangularis*), diatoms (*Stephanodiscus astraea, S. hantzschii, Nitzschia acicularis, Tabellaria fenestrata, Diatoma elongatum, D. vulgare, Fragilaria crotonensis, Synedra acus*), dinophytes (*Ceratium hirundinella, Gymnodinium spec., Peridinium umbonatum*) and cryptophytes (*Cryptomonas erosa, C. ovata, Rhodomonas pusillum*) were detected. During the years 2000 to 2005, dinophytes were increasingly present. *Ceratium hirundinella*, a species with very high cell volume, dominated the phytoplankton community. Its mean percentage share during the six-month vegetation period ranged between 66% and 99%. During the *Ceratium*-dominated years, the total phytoplankton biovolume was high compared to the years before or afterwards (Fig. 6). Starting in 2014, the contribution of cyanobacteria to the total phytoplankton biomass increased again (2014: 6%, 2016: 20%; Fig. 6), parallel to the increase of P concentrations (Fig. 3). However, on average dinophytes still dominated during the years 2014 and 2016.

3.6 Dynamics of zooplankton

Zooplankton total biomass was 1.3 mm^3 L^{-1} in spring and summer 1986, but decreased to 0.5 mm^3 L^{-1} during the alum treatment (Fig. 7). During treatment, cladocerans disappeared and both rotifers and ciliates declined in number. Copepods were less affected. From 1987 onwards, cladocerans re-established and contributed significantly to the algivorous biomass (*Daphnia hyalina*, *D. galeata*, *Diaphanosoma brachyurum*, *Bosmina longirostris*). With the exception of 1990, copepods (*Eudiaptomus gracilis*, *Cyclops spp*.) remained dominant in terms of biovolume. The average annual zooplankton biomass was somewhat lower (0.6 – 1.2 mm^3 L^{-1}) in the years after the treatment compared to 1986t (pre-treatment; Fig. 7).

3.7 Macrophytes

The macrophyte status of Lake Barleber evaluated according to the WFD was moderate (2007), good (2010 and 2014) and moderate (2016) (Tab. 4).

The species reaching the abundance degree of 5 in at least one transect and at least once were *Phragmites australis*, *Typha angustifolia*, *Potamogeton pectinatus*, *Ranunculus circinatus*, *Elodea canadensis*, and *Chara contraria*. The abundance degree of 4 was reached in at least one transect and at least once by *Fontinalis antipyretica*, *Potamogeton lucens*, *Potamogeton perfoliatus*, *Ceratophyllum demersum*, *Lemna trisulca*, *Chara globularis* and *Nitellopsis obtuse*. Detailed results of the macropyhte assessment are provided as supplementary material (Roenicke_etal_Macrophytes_LakeBarleber_2007-2016.xls). There was no consistent trend in the lake-wide averages of plant quantities separated by depth zones (Fig. S5). At all investigation dates, the entire lake bottom was also colonised by the filamentous green alga *Vaucheria spec.* down to the deepest parts of the lake basin.

4 Discussion

4.1 Suitability of aluminium sulphate treatment for Lake Barleber

From the available approaches for suppressing cyanobacterial blooms alum treatment was chosen for Lake Barleber. Under the conditions in Magdeburg (former GDR, i.e. East Germany) in 1986, alum treatment was seen as the most effective and efficient approach with the highest longevity. However, the use of alum is often seen critically and subject of discussion in the public due to the well-known toxicity of Al. Al and low pH values can have direct toxic effects on plankton (Havens, 1990, Wauer et al., 2004), as demonstrated for *Daphnia* at Al concentrations of 1 mg L^{-1} (Havas and Likens, 1985). Also indirect negative effects on zooplankton can occur. The non-selective filter feeding daphnids are particularly vulnerable to decreases in food quality such as increasing amounts of abiotic seston (e.g. Al flocs) (Kirk, 1992). Smaller zooplankton organisms (ciliates, rotifers) may also be removed from the water column directly by flocculation. In Lake Barleber, only a slight and short increase in Al concentrations was observed (Table 3). The the sufficiently high alkalinity in Lake Barleber (≥2 meq L^{-1}) kept pH at a natural level where Al solubility is very low. The occurrence of a diverse and abundant zooplankton community in the years following the treatment implies that there was no sustainable toxic effect. Also no damage of fish was reported by anglers.

High concentrations of sulphate can promote the formation of iron sulphide under anoxic conditions and, thus, reduce the P binding capacity. In Lake Barleber, the alum treatment did not change the sulphate concentration considerably (Table 3). Thus, selection of Al sulphate for phosphate precipitation in Lake Barleber was adequate. Due to the improvement of Secchi depths and the suppression of cyanobacteria mass developments of cyanobacteria allowing for undisturbed recreational use for almost 30 years, the treatment of Lake Barleber was a success.

4.2 Comparison of the treatment results with other lakes

In the 114 case studies examined by Huser et al. (2016) for the long-term impact and effectiveness of alum treatment, the applied quantities of alum ranged between 5 and 122 g Al m^{-2}. In six Danish lakes treated with polyaluminium chloride during the years 2001 to 2009, the applied Al doses ranged between 10 and 54 g Al m^{-2} (Jensen et al., 2015). Thus, the Al dose of 36 g Al m^{-2} applied to Lake Barleber is in the middle range of the published data.

The reduction of the phosphate concentration reached immediately after the treatment was very high (98% for SRP, 90% for TP). Although the average reductions for the years 1987-2014 were only 89% for SRP and 68% for TP, this was still enough to cause considerable changes in the phytoplankton composition (Fig. 7) and was in the range reached in other lakes (Huser et al., 2016). The reduction lasted at least until 2014, i.e. 28 years. This is much longer than Huser et al. (2016) found as average for unstratified (6 years) and stratified lakes (21 years). Obviously, in Lake Barleber, the intended surplus of binding capacity of the sediment, which resulted from the second half of the applied alum amount worked well until it was exhausted in 2015 or 2016. Nevertheless, our results indicate that the water quality improvements after the alum treatment were limited in time – a statement that was also clearly made by Huser et al. (2016).

The absence of surface inflows and the small catchment area of Lake Barleber probably has not provided enough sediment to bury the Al hydroxide rapidly as reported for Lake Süßer See (Lewandowski et al., 2003). Bioturbation during times of sufficient supply of oxygen at the sediment surface (Fig. 4) may have contributed to the long lasting reduction of P concentrations in Lake Barleber. Firstly, the consequent mixing of the sediment can have prevented a too deep burial of the Al hydroxide as observed in Lake Süßer See (Lewandowski et al., 2003). Secondly, bioturbation can allow for deeper oxidation of the sediment surface (Lewandowski and Hupfer, 2005). Future sediment investigations and a detailed P balance are needed to reliably identify the reasons for the end of the success phase of the alum treatment in Lake Barleber.

4.3 Evaluation of changes in phytoplankton

Comparing chlorophyll-*a* concentrations and phytoplankton biomass before and after the treatment may not lead to the conclusion that the measure was a success. The biomass and also the concentration of chlorophyll-*a* remained in the same range before and after the treatment, in particular when considering the interannual fluctuations. However, there were remarkable changes in the community composition of the phytoplankton. Green algae and increasingly dinophytes were dominant after the treatment. The mass occurrence of dinophytes in the years 2000 to 2005 mainly consisted of the species *Ceratium hirundinella. Ceratium* has the ability to migrate from surface waters during the day to the metalimnion during the night in order to take up P at depth (Whittington et al., 2000). This may have given the species a competitive advantage under P limitation. The change in phytoplankton community composition was accompanied by increases in Secchi depths and cell volumes of the dominant algal species.

Relevant phytoplankton consumers are in the group of crustaceans rather than rotifers or ciliates. For the onset of the clear-water phase, Lampert (1988) quoted a critical biomass of crustaceans of 1.5 g m^{-2} dry weight, which corresponds to a value of 1.3 mg L^{-1} fresh weight for our study lake. This biomass was reached or exceeded on 16 sampling days, i.e. in 15% of the samples. Within the group of crustaceans, large *Daphnia* species are the most efficient phytoplankton grazers. Earlier studies observed significant changes in the abundance of available prey at a *Daphnia* biomass of 0.7 mg L^{-1} fresh weight (Jürgens 1994, Tittel et al. 1998). This value was exceeded in 12% of the sampling days. Smaller cladocerans (*Bosmina*) were included in our calculations, which are less efficient phytoplankton consumers than *Daphnia* species. High biomasses occurred only in summer with maximum values in July 1987 (2.1 mg L^{-1}) as well as in June and August 1988 (1.9 mg L^{-1}). The well available and likely the most affected prey can be expected to be in the size of up to ca. 30 µm length. These were green algae, chrysophytes and partly diatoms, but not dinophytes (*Ceratium*) or filamentous cyanobacteria. We conclude that in about 30% to 40% of all summer days the zooplankton contributed significantly to high transparencies.

The phagotrophy of *Ceratium* was subjected to a debate among phycologists. Earlier reports of Hofeneder (1930) and of Dodge und Crawford (1970) have provided evidence of phagotrophy. Later on phagotrophy of *Ceratium* has been questioned (Chapman et al., 1981, Hansen und Calado, 1999, Stoecker 1999). However, there is clear evidence of phagotrophy by another species of the genera (*C. furcoides*, Bockstahler and Coats 1993). Further work is needed to address this issue (Moestrup and Calado 2018). If *C. hirundinella* was capable of ingesting bacteria this phagotrophy might enlarge the accessible P pool for the phytoplankton and contribute to the relatively high phytoplankton biovolume as was observed after the alum treatment. No decline in phytoplankton biomass after P reductions also was reported for other lakes, including the progressing dominance of phagotrophic algae (mixotrophs) (Jeppesen et al., 2005; Weyenmeyer and Broberg, 2014; Wentzky et al., 2018). Therefore, the total biovolume of phytoplankton often is a poor indicator of successful eutrophication control, while community composition and transparency are usually more powerful indicators.

The predominant cyanobacteria in Lake Barleber were *Anabaena* and *Aphanizomenon* in Lake Barleber. Both genera have the potential to fix nitrogen, to form surface scums and to produce toxins. Due to the formation of filaments and large colonies, both genera can hinder efficient grazing by filter feeders. Diazotrophic cyanobacteria are likely favoured by low nitrogen concentrations, but require high P levels for mass development (Reynolds et al., 2002).

To understand their dominance, the N:P ratio plays an important role. In a series of studies, the DIN:TP ratio in the water column was shown to be the best indicator for limiting conditions (e.g. Morris and Lewis, 1988; Bergström, 2010; Dolman et al., 2016). According to this ratio, the conditions in Lake Barleber changed from N-limitation to P-limitation after the phosphate precipitation. This counteracts the dominance of diaozotrophic cyanobacteria and can be regarded as the major reason for the suppression of cyanobacterial blooms after Al treatment (Brattebo et al. 2017).

Unfortunately, no quantitative data exist for the occurrence of macrophytes in Lake Barleber before the restauration in autumn 1986 and in the years immediately after the treatment. However, property owners at Lake Barleber reported that there was a considerable increase of the abundance of submerged macrophytes in 1987 and the following years. This is consistent with the memories of the author H. Rönicke.

Macrophytes are well known to stabilize clear water conditions in shallow lakes (Scheffer and van Nes 2007). According to Hilt (2015) this stabilizing effect may be found in deep lakes, too. Therefore, stabilizing effects can also be expected for Lake Barleber. The status class “good” found for macrophytes in 2010 and 2014 suggests that the macrophytes were abundant enough to contribute to the high water clarity. The decline in abundance for a number of species in 2016 already may have been a consequence of the re-occurrence of considerable P release from the sediment and, thus, an indicator for a shift back to a turbid state.

5 Conclusion

The precipitation of P using Al sulphate, as carried out in Lake Barleber in autumn 1986, has proven to be an effective restoration method. The P content of the water body was largely reduced and harmful cyanobacterial blooms were suppressed over a period of almost thirty years. This pronounced long-term effect had been achieved by applying an excess Al dose. By conditioning the P-rich sediment with $Al(OH)_3$ flocks, the release of P (internal fertilization) in the following years was considerably reduced. Consequently, only at the end of the study period, a significant increase in P concentrations was observed. In well-buffered eutrophic standing waters with little through-flow and low external input of P, precipitation with Al can lead to a marked long-term improvement in water quality. The missing reduction of the phytoplankton biomass and even its increase above the pre-treatment level did not hinder the recreational use of Lake Barleber since the transparency of the water was high and toxin producing cyanobacteria did not reach relevant abundances. We conclude that the success of the measure was based on a shift in the phytoplankton community composition caused by the strong reduction of P availability.

7 Acknowledgments

We wish to thank Klaus Bahr for the technical preparation of the restoration measure. We are also grateful to Doretta Mätzold and Erika Ruschak for conducting the microscopic analysis of the phytoplankton and zooplankton samples as well as Siegfried Frimel and Claudia Henschel for collecting the samples from the water body. Part of the research (1991-1993) was funded by the German Federal Ministry for Research and Technology (Funding ID 0339450B). We thank the anonymous reviewers for their constructive comments.

8 References

Backer, L.C., Manassaram-Baptiste, D., LePrell R., Bolton, B., 2015. Cyanobacteria and Algae Blooms: Review of Health and Environmental Data from the Harmful Algal Bloom-Related Illness Surveillance System (HABISS) 2007–2011. Toxins 7, 1048-1064.

Bauch, G., 1953. Der Barleber See. Mitteilungen für Naturkunde und Vorgeschichte. Aus dem Museum für Kulturgeschichte in Magdeburg 3, 173-183.

Benndorf, J., Kneschke, H., Kossatz, K., Penz, E., 1983. Manipulation der pelagischen Nahrungskette durch Raubfischbesatz in einem Experimentalgewässer. Veröff. Mus. Westlausitz 7, 41-70.

Bergström, A.-K., 2010. The use of TN:TP and DIN:TP ratios as indicators for phytoplankton nutrient limitation in oligotrophic lakes affected by N deposition. Aquat. Sci. 72, 277-281.

Bockstahler, K.R., Coats, D.W., 1993. Spatial and temporal aspects of mixotrophy In Chesapeake Bay *Dinoflagellates*. J Eukaryotic Microbio. 40, 49-60.

Brattebo, S.K., Welch, E.B., Gibbons, H.L., Burghdoff, M.K., Williams, G.N., Oden, J.L., 2017. Effectiveness of alum in a hypereutrophic lake with substantial external loading. Lake Reserv. Manag. 33, 108-118.

Carpenter, S.R., 2008. Phosphorus control is critical to mitigating eutrophication. PNAS 105, 11039-11040.

Chapman, D.V., Livingstone, D., Dodge, J.D., 1981. An electron microscope study of the excystment and early development of the dinoflagellate *Ceratium hirundinella*. Br. phycol. J. 16, 183-194.

Chorus, I., Bartram, J., 1999. Toxic cyanobacteria in water: a guide to their public health consequences, monitoring and management. World Health Organisation, E&FN Spon, London.

Chorus, I., Falconer, I.R., Salas, H.J., Bartram, J., 1999. Health risks caused by freshwater cyanobacteria in recreational waters. J Toxicol. Env. Heal. B 3, 323-347.

Cooke, G.D., Heath, R.D., Kennedy, R.H., McComas, M.R., 1982. Change in lake trophic state and internal phosphorus release after aluminum sulfate application. Wat. Resour. Bull. 18, 699-705.

Cooke, G.D., Welch, E.B., Peterson, S.A., Nichols, S.A., 2005. Restoration and management of lakes and reservoirs. 3rd edition. CRC Press, Boca Raton, FL.

DIN 38412, 1986. Bestimmung des Chlorophyll-a-Gehaltes von Oberflächenwasser. Deutsche Einheitsverfahren der Wasser-, Abwasser- und Schlammuntersuchung, 16. Lieferung, Teil L16, Verlag Chemie, Weilheim.

Dodge, J., Crawford, R., 1970. The morphology and fine structure of *Ceratium hirundinella* (Dinophyceae). J. Phycology 6, 137-149.

Dolman, A.M., Mischke, U., Wiedner, C., 2016. Lake-type specific seasonal patterns of nutrient limitation in north German lakes and target nitrogen and phosphorus concentrations for a good ecological status. Freshwat. Biol. 61, 444-456.

Hannappel, S., Strom, A., 2020. Methode zur Ermittlung des Phosphoreintrags über das Grundwasser in den Barleber See bei Magdeburg. Korrespondenz Wasserwirtschaft 13, 24-30 + Correction. Korrespondenz Wasserwirtschaft 13, 110.

Hansen, P.J., Calado, A.J., 1999. Phagotrophic mechanisms and prey selection in free-living *Dinoflagellates*. J. Eukaryotic Microbio. 46, 382-389.

Havas M., Likens G.E., 1985. Toxicity of aluminium and hydrogen ions to *Daphnia catawba*, *Holopedium gibberum*, *Chaoborus punctipennis* and *Chironomus anthrocinus* from Mirror Lake, New Hampshire. Can. J. Zool. 63, 1114-1119.

Havens, K.E., 1990. Aluminium binding to ion exchange sites in acid-sensitive versus acid-tolerant *cladocerans*. Env. Pollut. 64, 133-141.

Herzsprung, P., Bozau, E., Büttner, O., Duffek, A., Friese, K., Koschorreck, M., Schultze, M., von Tümpling, W., Wendt-Potthoff, K., 2006. Routine analysis of sediment pore water of high ionic strength. Acta hydrochim. hydrobiol. 34, 593-607.

Herzsprung, P., Duffek, A., Friese, K., de Rechter, M., Schultze, M., von Tümpling, W., 2005. Modification of a continuous flow method for analysis of trace amounts of nitrate in iron-rich sediment pore-waters of mine pit lakes. Wat. Res. 39, 1887-1895.

Hilt, S., 2015. Regime shifts between macrophytes and phytoplankton – concepts beyond shallow lakes, unravelling stabilizing mechanisms and practical consequences. Limnetica 34, 467-480.

Hofeneder, H., 1930. Über die animalische Ernährung von *Ceratium hirundinella* O.F. Müller und über die Rolle des Kernes bei dieser Zellfunktion. Arch. Protistenkunde 71, 1-32.

Huang, P.M., Wang, M.K., Kampe, N., 2002. Aluminum hydroxides. In: Dixon JB, Schulze DG (eds) Soil Mineralogy with Environmental Applications. SSSA Book Series, no. 7. Soil Science Society of America, Madison, WI, pp 261-289.

Huisman, J., Matthijs, H.C.P., Visser, P.M. (Eds.), 2005. Harmful Cyanobacteria. Springer, Dordrecht.

Huser, B.J., Brezonik, P., Newman, R., 2011. Effects of alum treatment on water quality and sediment in the Minneapolis Chain of Lakes, Minnesota, USA. Lake Reserv. Manag. 27, 220-228.

Huser, B.J., Egemose, S., Harper, H., Hupfer, M., Jensen, H., Pilgrim, K.M., Reitzel, K., Ryding, E., Futter, M., 2016. Longevity and effectiveness of aluminum addition to reduce sediment phosphorus release and restore lake water quality. Wat. Res. 97, 122-132.

Ibelings, B.W., Fastner, J., Bormans, M., Visser, P.M., 2016a. Cyanobacterial blooms: Ecology, prevention, mitigation and control: Editorial to a CYANOCOST Special Issue. Aquat. Ecol. 50, 327-331.

Ibelings, B.W., Bormans, M., Fastner, J., Visser, P.M., 2016b. CYANOCOST special issue on cyanobacterial blooms: synopsis – a critical review of the management options for their prevention, control and mitigation. Aquat. Ecol. 50, 595-605.

Jensen, H.S., Reitzel, K., Egemose, S., 2015. Evaluation of aluminum treatment efficiency on water quality and internal phosphorus cycling in six Danish lakes. Hydrobiologia 751, 189-199.

Jeppesen, E., Søndergaard, M., Jensen, J.P., Havens, K.E., Anneville, O., Carvalho, L., Coveney, M.F., Deneke, R., Dokulil, M.T., Foy, B., Gerdeaux, D., Hampton, S.E., Hilt, S., Kangur, K., Köhler, J., Lammens, E.H.H.R., Lauridsen, T.L., Manca, M., Miracele, M.R., Moss, B., Nõges, P., Persson, G., Phillips, G., Portielje, R., Romo, S., Straile, D., Tatrai, I., Willén, E., Winder, M., 2005. Lake responses to reduced nutrient loading - an analysis of contemporary long-term data from 35 case studies. Freshwat. Biol. 50, 1747–1771

Jürgens, K., 1994. Impact of Daphnia on planktonic microbial food webs–A review. Mar. Microb. Food Webs 8, 295-324

Kirk, K.L., 1992. Effects of suspended clay on *Daphnia* body growth and fitness. Freshwat. Biol. 28, 103-109.

Klapper, H., 1991. Control of Eutrophication in Inland Waters. Ellis Horwood Limited, Chichester.

Klapper, H., Geller W., 2001. Water quality management of mining lakes – a new field of applied hydrobiology. Acta hydrochim. hydrobiol. 29, 363-374.

Lampert, W., 1988. The relationship between zooplankton biomass and grazing: A review. Limnologica 19, 11-20

Landner, L., 1970. Lake Restoration: Trials with direct precipitation of P in polluted lakes. Swedish Water Air Pollution Res. Lab. Stockholm.

Lewandowski, J., Hupfer, M., 2005. Effect of macrozoobenthos on two-dimensional small-scale heterogeneity of pore water phosphorus concentrations in lake sediments: A laboratory study. Limnol. Oceanogr. 50, 1106-1118.

Lewandowski, J., Schauser, I., Hupfer, M., 2003. Long term effects of phosphorus precipitations with alum in hypereutrophic Lake Süsser See (Germany). Wat. Res. 37, 3194-3204.

Lürling, M., Mackay, E., Reitzel, K., Spears, B.M., 2016. Editorial – A critical perspective on geo-engineering for eutrophication management in lakes. Wat. Res. 97, 1-10.

Melzer, A., 1999. Aquatic macrophytes as tools for lake management. Hydrobiologia 396, 181–190.

Morris, D.P., Lewis, W.M., 1988. Phytoplankton nutrient limitation in Colorado mountain lakes. Freswat. Biol. 20, 315-327.

Moestrup, Ø., Calado, A., 2018. Freshwater flora of central Europe, Vol. 6, Dinophyceae. Springer, Berlin.

Paerl, H.W., Scott, J.T., 2010. Throwing fuel on the fire: synergistic effects of excessive nitrogen inputs and global warming on harmful algal blooms. Env. Sci. Technol. 44, 7756-7758.

Premazzi, G., 1980. Phytoplankton productivity in Lake Lugano before water treatment. Schweiz. Z. Hydrol. 12, 155-170.

Reynolds, C.S., Huszar, V., Kruk, C., Naselli-Flores, L., Melo, S., 2002. Towards a functional classification of the freshwater phytoplankton. J. Plankton Res. 24, 417-428.

Rönicke, H., Bahr, K., 1989. Nährstoffausfällung an einem Magdeburger Badesee zur Verbesserung der Wasserbeschaffenheit und Erhöhung des Erholungswertes. Wasserwirt.-Wassertech. 39, 187-188.

Rönicke, H., Bahr, K., 1990. Entwicklung eines Schlammstechers zur Entnahme ungestörter Sedimentproben aus Oberflächengewässern. Wasserwirt.-Wassertech. 40, 24.

Ruttner-Kolisko, A., 1977. Suggestions for biomass calculation of plankton rotifers. Arch. Hydrobiol. Beih. Ergebn. Limnol. 8, 71-76.

Schaumburg, J., Schranz, C., Hofmann, G., Stelzer, D., Schneider, S., Schmedtje, U.,2004. Macrophytes and phytobenthos as indicators of ecological status in German lakes – a contribution to the implementation of the Water Framework Directive. Limnologica 34, 302–314.

Scheffer, M., van Nes, E.H., 2007. Shallow lakes theory revisited: various alternative regimes driven by climate, nutrients, depth and lake size. Hydrobiologica 584, 455-466.

Sharma, N.K., Rai, A.K., Stal, L.J. (Eds.), 2013. Cyanobacteria: an economic perspective. John Wiley & Sons, New York.

Stelzer, D., Schneider, S., Melzer, A., 2005. Macrophyte based assessment of lakes – acontribution to the implementation of the European Water Framework Directivein Germany. Int. Rev. Hydrobiol. 90, 223–237.

Stoecker, D.K., 1999. Mixotrophy among Dinoflagellates. J. Eukaryotic Microbio. 46, 397-401.

Stroom, J.M., Kardinaal, W.E.A., 2016. How to combat cyanobacterial blooms: strategy toward preventive lake restoration and reactive control measures. Aquat. Ecol. 50, 541–576.

Tittel, J., Zippel, B., Geller, W., Seeger, J., 1998. Relationships between plankton community structure and phytoplankton size distribution in lakes of northern Germany. Limnol. Oceanogr. 43, 1119-1132

Utermöhl, H., 1958. Zur Vervollkommnung der quantitativen Phytoplankton-Methodik. Mitt. Int. Ver. Limnol. 9, 1-38.

Wagner, K.J., 2017. Preface: Advances in phosphorus inactivation. Lake Reserv. Manag. 33, 103-107.

Watson, S.B., Ridal, J., Boyer, G.L., 2008. Taste and odour and cyanobacterial toxins: impairment, prediction, and management in the Great lakes. Can. J. Fisher. Aquat. Sci. 65, 1779-1796.

Wauer, G., Heckemann, H.-J., Koschel, R., 2004. Analysis of toxic aluminium species in natural waters. Microchim. Acta 146, 149-154.

Welch, E.B., Cooke, G.D., 1999. Effectiveness and longevity of phosphorus inactivation with alum. Lake Reserv. Manag. 15, 5-27.

Wentzky, V.C., Tittel, J., Jäger, C.G., Rinke, K., 2018. Mechanisms preventing a decrease in phytoplankton biomass after phosphorus reductions in a German drinking water reservoir – results from more than 50 years of observation. Freshwat. Biol. 63, 1063-1076.

Weyhenmeyer, G.A., Broberg, N., 2014. Increasing algal biomass in Lake Vänern despite decreasing phosphorus concentrations: A lake-specific phenomenon? Aquat. Ecosys. Health & Manag. 17, 341–348.

Whittington, J., Sherman, B., Green, D., Roderick, L.O., 2000. Growth of *Ceratium hirundinella* in a subtropical Australian reservoir: the role of vertical migration. J Plankton Res 22, 1025-1045.

Zamparas, M., Zacharias, I., 2014. Restoration of eutrophic freshwater by managing internal nutrient loads. A review. Sci. Total. Env. 496, 551-562.

Table 1 Morphometric parameters for Lake Barleber

Lake parameter	Value
Location	52° 13′ 15′′ N, 11° 39′ 0′′ O
Area	103 ha
Volume	6.9×10^6 m^3
Mean depth	6.7 m
Maximum depth	10.0 m
Inflow	groundwater

Table 2 Application of aluminium sulphate to the surface of Lake Barleber

Period:	3 October to 13 November 1986
Precipitating agent:	aluminium sulphate
Quantity added:	480 t of solution containing 37 t Al^{3+}
Al dose per volume:	5.7 mg Al^{3+} L^{-1}
Al dose per area:	36 g Al^{3+} m^{-2}
Distribution technique:	floating vessel with tank for the aluminium sulphate solution and a perforated tube (with 10 mm holes) pushed by a motor boat

Table 3 Aluminium and sulphate concentrations as well as pH values (mean values 0-7.5 m) before, during and after the application of aluminium sulphate (n.d. - not analysed)

Date	Al^{3+} (mg L^{-1})	SO_4^{3-}(mg L^{-1})	pH
11/09/1986	n.d.	505	8.3
29/09/1986	0.04	495	8.1
08/10/1986	0.12	510	7.8
15/10/1986	0.17	513	7.6
23/10/1986	0.25	525	7.5
29/10/1986	0.5	534	7.4
12/11/1986	0.4	651	7.6
19/11/1986	0.1	611	7.7
10/12/1986	0.05	454	7.8

Table 4 Results of macrophyte investigations summarized per transect (for all depth zones) and evaluation in terms of status classes

Date	Transect	Location of transect	Lower limit of macrophytes (m)	Species number of transect	Total species in Lake Barleber	Status class of transect	Status class of Lake Barleber
23.07.2007	1	West	5.4	5	16	moderate	moderate
23.07.2007	2	East	6.5	12	16	good	moderate
23.07.2007	3	North-west	6.0	11	16	moderate	moderate
23.07.2007	4	South	6.5	8	16	good	moderate
26.08.2010	1	West	7.0	7	15	good	good
26.08.2010	2	East	7.7	7	15	moderate	good
26.08.2010	3	North-west	7.0	5	15	good	good
26.08.2010	4	South	7.0	8	15	good	good
26.08.2010	5	South-east	7.2	6	15	very good	good
29.06.2014	1	West	4.8	6	19	moderate	good
29.06.2014	2	East	6.1	8	19	good	good
29.06.2014	3	North-west	5.8	10	191	good	good
29.06.2014	4	South	4.8	9	19	good	good
29.06.2014	5	South-east	5.8	10	19	very good	good
13.07.2016	1	West	8.3	5	18	moderate	moderate
13.07.2016	2	East	8.2	13	18	good	moderate
13.07.2016	3	North-west	6.5	11	18	good	moderate
13.07.2016	4	South	6.5	8	18	good	moderate
13.07.2016	5	South-east	7.2	7	18	good	moderate

Figure captions

Figure 1 Concentrations of phosphorus (SRP, spring overturn values, 0-7.5 m), 1950 to 2017 (based on Rönicke et al. 1995, extended)

Figure 2 SRP concentration (0-7.5 m) and oxygen in the deep water layer (9 m), 1986 and 1987 (IC – ice cover, AP – aluminium precipitation)

Figure 3 Alum application and the effects on phosphorus concentration (mean values 0-7.5 m), 1986 to 2016

Figure 4 Dynamics of SRP- and oxygen concentration in the deep water layer (9 m) in Lake Barleber, 1986 to 1992, 1999, 2010, 2013, 2014, and 2016

Figure 5 Mean values of N:P ratio (DIN:TP) in the summer months (June to August) in Lake Barleber, 1986 to 2016

Figure 6 Mean values of phytoplankton biomass and composition (0-7.5 m) and Secchi depths during the vegetation period (May to October) in Lake Barleber, 1985 to 2016

Figure 7 Zooplankton biovolume and community composition. Upper part: total biovolume, means from 9-19 samplings per year; lower part: composition derived from proportions at individual sampling days. * before precipitation (February to October 2, 14 samples), # during precipitation (October 15 to December, 6 samples)

Electronic supplementary material for

Suppression of bloom-forming colonial cyanobacteria by phosphate precipitation: a 30 years case study in Lake Barleber See (Germany)

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The supplementary material consists of Figures S1-S4, provided below in this document and a MS Excel file named SupplementaryMaterial_Roenicke_etal.xlsx. The latter provides the data used in the paper.

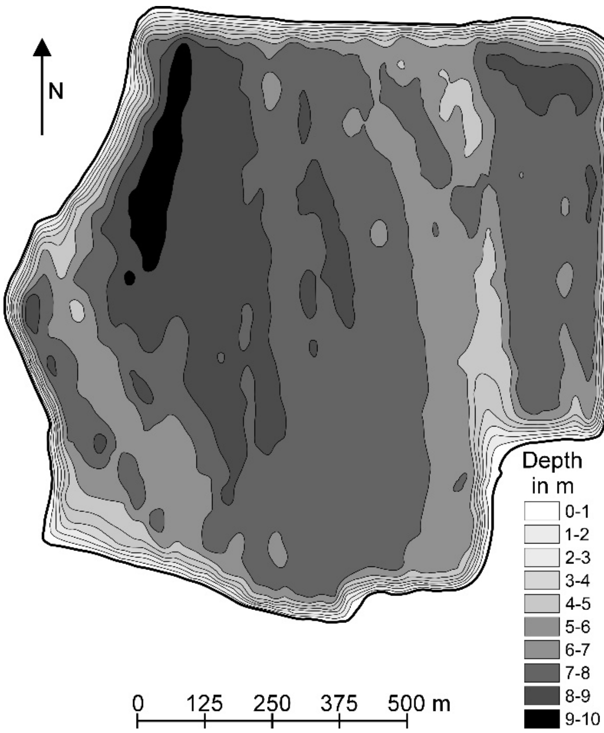

Figure S1 Bathymetric map of Lake Barleber (redrawn based on the map prepared by Fisch und Umwelt e.V., Rostock, Germany by order and for account of Landesbetrieb für

Hochwasserschutz und Wasserwirtschaft Sachsen-Anhalt, Gewässerkundlicher Landesdienst, Magdeburg, Germany in 2003).

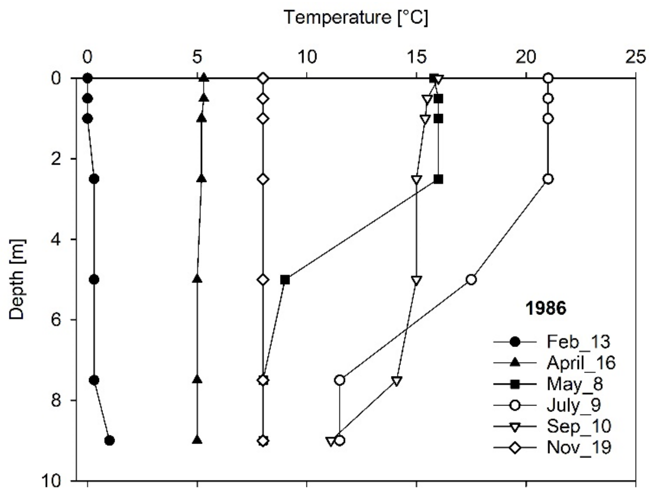

Figure S2 Results of selected temperature measurements from surface to bottom in 1986 at the deepest site of Lake Barleber.

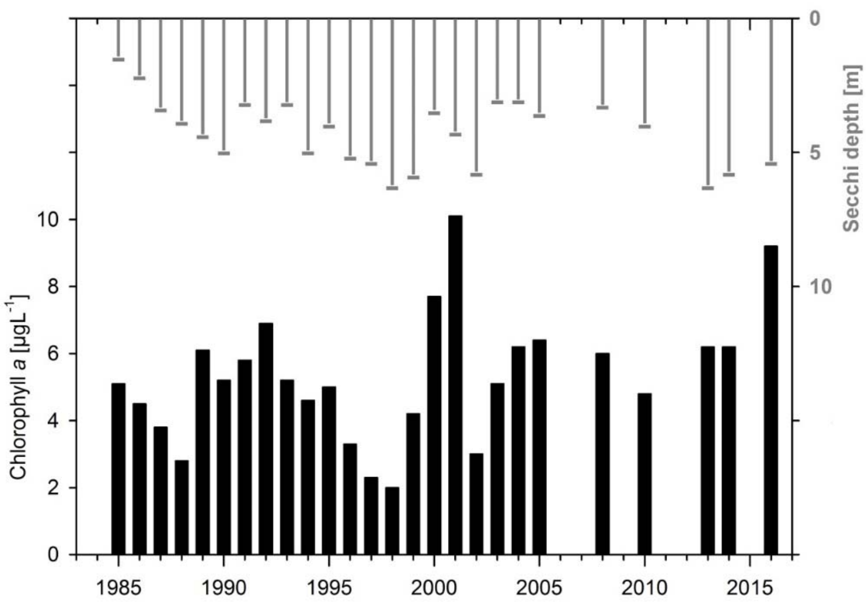

Figure S3 Mean values of chlorophyll-*a* concentration (0-7.5 m) and Secchi depths during the vegetation period (May to October) in Lake Barleber, 1985 to 2016.

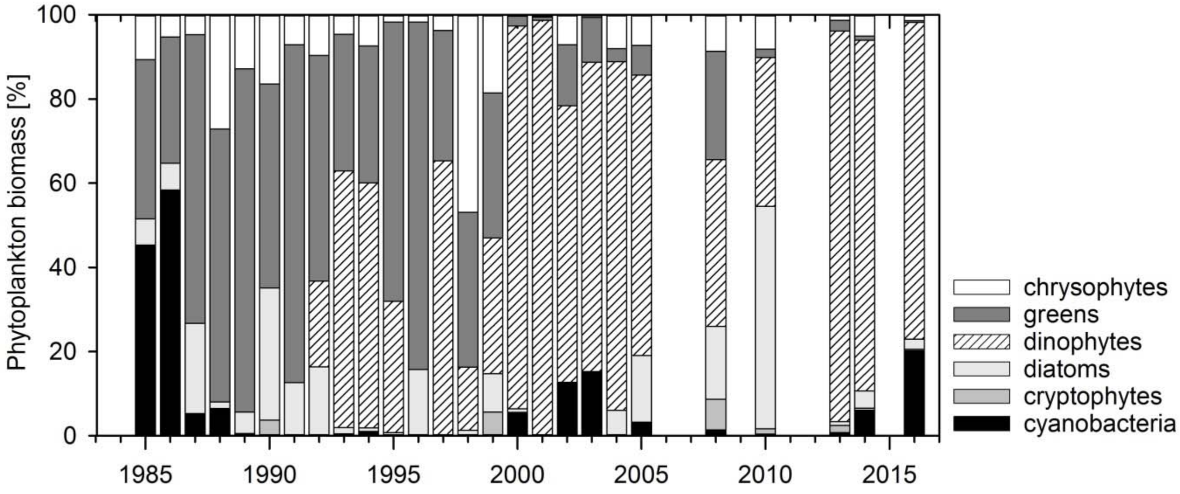

Figure S4 Percentage of phytoplankton biomass classified into the major taxonomic groups (annual mean values, 0-7.5 m) during the vegetation period (May to October) in Lake Barleber, 1985 to 2016.