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Controlling internal phosphorus loading in lakes by physical methods to reduce cyanobacterial blooms: a review

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Abstract The aim of this paper was to summarize the current knowledge on how physical methods can reduce or control internal P release from sediments in lakes and reservoirs. Particular emphasis is given to the role of internal phosphorus load in fueling cyanobacterial blooms which are predicted to increase in frequency and intensity in response to climate change and eutrophication. We present selective case studies (both successful and unsuccessful) to assess the applicability and efficiency of major physical approaches used for decades to reduce internal loading in different systems of various morphology. In particular, we concentrate on where and when (1) hypolimnetic aeration/oxygenation, (2) hypolimnetic withdrawal and (3) sediment dredging are likely to reduce cyanobacterial blooms and whether these methods have an adverse impact on other organisms. We conclude that each method has its strength and weakness depending on the system considered. Sufficient knowledge of all lake nutrient sources and their dynamics together with detailed lake and sediment characteristics is an essential prerequisite for choosing an appropriate control method. We also report that many experiences demonstrated that a combination of restoration methods is often more successful than a single method.

Keywords Internal P loading · Cyanobacterial control · Physical in-lake restoration methods · Adverse impacts on biota

Introduction

Cyanobacterial blooms are becoming more frequent and more intense as well as more widespread (Pearl and Huisman 2008; O'Neil et al. 2012) and create many adverse effects on ecosystems and human health including toxin exposures (Codd et al. 2005). Although the environmental factors (i.e., nutrients, light and temperature) controlling their onset and proliferation have been known for many decades, methods to control their development have been moderately successful. As phosphorus (P) is typically the limiting nutrient for productivity in lakes, mechanisms affecting P bioavailability have been well studied (Vollenweider 1975; Reynolds 1984; Wetzel 2001) although nitrogen plays a role in some shallow lakes (Sagrario et al. 2005; Pearl et al. 2011). While nutrients from catchments are partly in particulate form that settles in the water column and are not

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directly used by phytoplankton until released from particles, internal loads in lakes are predominantly in dissolved form directly available for algal growth. From this point of view, sediments are key components for considering in-lake measures to reduce phosphorus availability for cyanobacterial growth from internal load.

Preventive methods consisting of limiting nutrient inputs from external loading are without any discussion the long-term solution. However, they are often complemented by in-lake methods in order to accelerate the recovery process. Among them, physical, chemical and biological methods to reduce internal loading have been implemented in many waterbodies with relative success. Chemical methods (such as increase sediment sorption capacity by adding iron, aluminum or calcium) and biological methods (such as biomanipulation) will be discussed in separate papers in this special issue. In this paper, we review the applicability and relative success of major physical approaches used for decades to reduce internal loading in different systems of various morphology from representative case studies. While there are several excellent existing reviews dealing with factors influencing internal phosphorus loading in lakes (i.e., Søndergaard et al. 2001; Boström et al. 1988; Nürnberg 2009) or methods to control P release (Cooke et al. 2005), the main aim of this paper is to summarize the current knowledge on where and when physical methods can be used to control cyanobacterial blooms that are likely to increase in frequency and intensity in response to climate change and eutrophication.

We first discuss the direct and indirect links between internal loading and cyanobacterial blooms. After describing the working mechanism of each restoration method, we assess their relative successes and failures in controlling cyanobacterial blooms and try to understand the underlying reason. We then review the current knowledge on the ecological impact of each restoration method. Finally, we give some recommendations for choosing the appropriate physical control method.

Direct and indirect links between sediment phosphorus release and cyanobacterial growth

In shallow lakes, there is a direct link between the phosphorus release at the sediment water interface and the euphotic zone where photosynthesis takes place, as the water column remains vertically well mixed under the influence of wind-induced mixing (Søndergaard et al. 2003; Pettersson et al. 2010). In such systems, internal load often represents the main summer P load (Steinman et al. 2009; Nürnberg et al. 2012), which has a strong influence on algal bioavailability. Nürnberg et al. (2012) quantified internal phosphorus load over 26 years in a shallow mesotrophic lake and estimated that it was larger than external loads during years of high water temperature and low water load. Similarly, Steinman et al. (2009) conducted a study in a small eutrophic lake in Michigan and showed that in summer internal loading contributed 80 % of the total load. The increase in TP load in summer is often an indication of internal load. Interestingly, Istvánovics et al. 2002 reported on a positive feedback between high photosynthesis, elevated pH and P desorption from the resuspended sediments in the shallow Lake Balaton, highlighting that the cyanobacterium *Cylindrospermopsis raciborskii* was capable of generating high internal P release. Cyanobacteria are likely to dominate over other phytoplankton groups in these systems if the temperature is high (Pearl and Huisman 2008; Rigosi et al. 2014) and if the turbidity is also high. Indeed, high nutrient concentrations and high turbidity are often associated with cyanobacterial blooms in shallow systems, referred as one of the alternative stable state (Scheffer et al. 1997; Ibelings et al. 2007). High turbidity in shallow lakes is often associated with wind-driven sediment resuspension of both organic (algal biomass) and inorganic matter (suspended solid particles). Phlips et al. (1997), however, pointed out that if turbidity from inorganic particles becomes too high from wind resuspension, cyanobacteria productivity also becomes light limited.

In deep stratified lakes, a similar direct link may occur in the shallow parts (littoral zone) of the lake, which are well mixed and illuminated (Callieri et al. 2014) and where P release from sediments is in contact with the epilimnion (Søndergaard et al. 2005). Indirect link on the other hand occurs in the deeper parts (pelagic zone) of stratified lakes by mixing across the thermocline. Internal loading transfer to the epilimnion in stratified lakes occurs through turbulent vertical flux across the thermocline by mixing events, which entrain bottom waters through thermocline deepening (Soranno et al. 1997; Pannard et al. 2007; Pettersson et al. 2010), or by internal waves (MacIntyre et al. 1999; Bormans et al. 2004). Reported studies that quantify this entrainment are rare. Soranno et al. (1997) quantified the entrainment from hypolimnetic P to the budget of epilimnetic P in the deep eutrophic lake Mendota, USA, over a 2-year period. They showed that the entrainment potential depended on the mixing potential and the nutrient load potential. While Chl *a* concentration was similar in both years, cyanobacteria dominated the algal biomass with 85 and 96 %, respectively, over the 2 years. P entrained in the epilimnion from internal load represented up to 78 % of yearly total TP load. Entrainment events were related to storms and concentrated in early and late summer when the stratification was weaker in accordance with other studies (Pannard et al. 2011). Bormans et al. (2004) explained the persistence of a cyanobacterial bloom dominated by *Cylindrospermopsis raciborskii* from this entrainment of hypolimnetic high nutrients from wind-induced internal waves. Similarly, Pannard et al. (2011) demonstrated the close relationship between the metalimnetic *Planktothrix agardhii* bloom in a small Canadian lake and the entrainment of the nutrient-rich hypolimnion associated with internal waves.

In deep lakes in autumn and winter, mixing by convection redistributes phosphorus throughout the entire water column. Hence, there is generally a time lag between the high internal load generated predominantly in summer and its transfer into the epilimnion, which predominantly occurs in autumn. Interestingly, Sherman et al. (2000) have reported in a deep Australian reservoir a high correlation between the phosphorus concentration in the hypolimnion and the mean annual biomass of the cyanobacterial, suggesting a strong dependency on internal loading for cyanobacterial growth. Hence, considering internal loading from entrainment may help explain the observed variability in epilimnetic TP dynamics in stratified lakes. As opposed to other phytoplankton groups, cyanobacteria through the regulation of their buoyancy are well adapted to grow in strongly stratified lakes and reservoirs, gas vesicles allowing them to float and proliferate in the well-illuminated surface layers while other phytoplankton groups settle below the thermocline (Huisman et al. 1999; Bormans and Condie 1998). Cyanobacteria still require nutrients, which are more limiting in the surface layer in the summer than in the fall following overturn. Cyanobacteria are therefore the primary phytoplankton group to benefit from internal loading.

Hence, in both shallow and deep systems, the reduction of internal loading is likely to reduce the biomass of cyanobacterial blooms.

Expected impact of climate change on internal load

Climate change is now widely recognized to be one of the most distinct threats to aquatic ecosystems. The impacts of climate change on internal phosphorus loading have only been studied quite recently with particular attention to the effects of year-to-year variability in climate on the internal loads (Pettersson et al. 2010; Nürnberg et al. 2012). Jeppesen et al. (2007) discussed how climate change, in particular warming and drought, might influence the shallow lakes response to nutrient loading reduction. They suggested that the most probable outcome is that shallow European lakes will reach a more turbid state and in the south will become more saline as well. Expected increase in temperature and strengthening of the stratification leading to shallower surface mixed layer depth will likely increase the surface area of the sediments in contact with anoxic waters, hence enhancing phosphorus release from sediments (Schauser and Chorus 2009). Moreover, temperature increase will also lead to greater bacterial mineralization, which also should increase phosphorus release (Søndergaard et al. 2003). It is therefore expected that internal loading will be enhanced in response to climate change, and it is likely that cyanobacteria will be the first group to benefit from these increases as they are adapted to warm, turbid as well as stratified conditions (Rigosi et al. 2014; Mantzouki et al. 2016).

Control methods of P release by physical approaches

P release from sediments is influenced by many factors and processes that depend on the characteristics of the sediments and the water column, including redox potential, concentrations of Fe, Al and Ca, sediment organic matter and its reactivity, pH, temperature, resuspension, diffusion, microbial and bioturbation activities. Although internal loads are difficult to quantify due to the complexity of the many physical, chemical and biological processes involved, restoration methods to control them have been used for several decades. We now present the main physical techniques used to reduce P release from sediments and discuss selective representative case studies, which explicitly report on the restoration methods responses on cyanobacteria biomass.

Hypolimnetic aeration and oxygenation

Working mechanism

Hypolimnetic aeration or oxygenation is a technique used since the seventies to combat water quality problems associated with anoxia. Its primary goal is to raise or maintain dissolved oxygen (DO) at the bottom of lakes or reservoirs to suppress sediment release of reduced species (i.e., Fe, Mn, P and ammonia) by creating oxic conditions at the sediment–water interface. The technique intends to avoid complete destratification or warming the hypolimnion. In the particular case of controlling cyanobacterial blooms, the basic principle of hypolimnetic aeration/oxygenation system is to reduce phosphorus concentrations in the hypolimnion with minimum mixing across the metalimnion to avoid entrainment of nutrient-rich bottom waters into the epilimnion.

Aeration can be achieved by compressors that introduce air in the bottom of the lake through perforated tubes. An overview of various designs for hypolimnetic aerators is available in Cooke et al. (2005). They are grouped into three categories: (1) mechanical agitation, (2) injection of pure oxygen and (3) injection of air through a full lift or partial lift design or through a down-flow injection design. The hypolimnetic aeration may not operate satisfactorily if the water body is too shallow even if stratification exists, as the density gradient may not be sufficient to resist thermocline erosion while the hypolimnion is mixing. Procedures for sizing hypolimnetic aerators are described in detail by Lorenzen and Fast (1977), Ashley (1985) and Little (1995). The costs associated with this method are not very often reported in the literature. In Lake Amisk, capital costs were $30 k and operating cost were about $49 k per year (Prepas and Burke, 1997). Due to the need of the electric power, the operating costs of this method are relatively high. Moreover, aerators are usually put in spring and run during the whole summer season until autumn. Costs of oxygen injection estimated by Hickey and Gibbs (2009) were around $2.5 k/ha/year, while Cooke et al. (2005) reported overall costs for an average of 15 lakes in the USA of $3 k/ha/year.

Aeration systems are also used to completely mix the water column. In that configuration, although increasing O_2 concentration in the hypolimnion is also an objective to reduce internal P loading, the complete mixing has other impacts (on light climate and buoyancy regulation) that affect cyanobacterial growth by other means than just reducing internal P loading. Hence, a separate article by Visser et al. (2016) in this special issue is specifically dedicated to artificial mixing as a method to control cyanobacterial blooms.

Successful and unsuccessful examples

Many examples of hypolimnetic aeration applications in lakes and reservoirs worldwide have been reported in the literature, and extensive reviews include Beutel and Horne (1999), Cooke et al. (2005) and Singleton and Little (2006). Here, we select studies that explicitly report on phytoplankton and cyanobacterial blooms responses to hypolimnetic aeration/oxygenation (Table 1). Moore et al. (2012) reported on a successful application of long-term hypolimnetic aeration on phytoplankton and cyanobacteria dynamics in Newman's Lake (USA), a relatively shallow ($z_{\max} = 9$ m) dimictic eutrophic lake, where aeration was implemented since 1992 to control phosphorus recycling from the anoxic lake sediments. They stated that the system was capable of maintaining adequate oxygen to reduce internal loading but needed to stay in operation year round. They reported a phytoplankton community dominated by diatoms (*Aulacoseira*) with minimum cyanobacteria during operations and found a good correlation between phytoplankton biomass and TP concentrations with significant decrease over time. Cyanobacteria only contributed significantly to the biomass before 2000 when TP was higher than 40 µg/L. However, they also mentioned that external loads were not well monitored, and therefore, it was difficult to assess the efficiency of the hypolimnetic oxygenation. Toffolon et al. (2013) reported on a long-term (10 years) study of side stream pumping in Lake Serraia, Italy ($z_{\max} = 18$ m) that although increased DO was observed during operations, an increase in hypolimnetic temperature resulted from probable entrainment of hypolimnetic waters into the epilimnion. In response,

Table 1 Summary of lake characteristics and effects of physical restoration methods on internal P release and on cyanobacteria biomass in different lakes and reservoirs

Lake	Lake characteristics	Method used	Effect on internal P load	Effect on cyanobacteria	References
Newman, USA	Eutrophic, dimictic $z_{max} = 9$ m area = 5.6 km^2	Hypolimnetic aeration (+microfloc alum injection)	Increase DO, reduced internal P load	Decreased biomass	Moore and Christensen (2009) and Moore et al. (2012)
Serraia, Italy	Eutrophic, dimictic, $z_{max} = 18$ m area = 0.45 km^2	Hypolimnetic oxygenation	Increase DO, first increase temp hypolimnion than decrease P availability	First no change than decrease biomass and shift from cyanobacteria to chlorophytes	Toffolon et al. (2013)
Amisk, Canada	Eutrophic, dimictic $z_{max} = 34$ m area = 5.5 km^2	Hypolimnetic oxygenation	Increase DO and reduced hypolimnion and epilimnion TP	Reduced Chl *a* biomass and shift from cyanobacteria to a mix including diatoms	Prepas and Burke (1997) and Webb et al. (1997)
Sempach, Switzerland	Eutrophic, monomictic $z_{max} = 87$ m area = 14.5 km^2	Hypolimnetic aeration	Increase DO, no change in P release due to high sedimentation rates and sed/water interface remaining anoxic	Increase in cyanobacterial biomass	Gächter and Müller (2003) and Burgi and Stadelman (2002)
Tegel, Germany	Eutrophic, dimictic $z_{max} = 16$ m, area = 3.06 km^2	Hypolimnetic aeration (+external load control)	No effect on P reduction due to high sedimentation rates, no effect on redox-sensitive adsorption of iron	No effect on phytoplankton or cyanobacterial biomass	Lindenschmidt and Chorus (1997) and Chorus and Schauser (2011)
Medical Lake, USA	Eutrophic $z_{max} = 18$ m area = 0.63 km^2	Hypolimnetic aeration (+alum treat)	Reduced TP in hypolimnion but entrainment into epilimnion	No effect on Chl *a*, shift from cyanobacteria to chlorophytes	Soltero et al. (1994)
Wilcox, Canada	Mesotrophic, dimictic $z_{max} = 17$ m area = 0.56 km^2	Hypolimnetic oxygenation/ Aeration (+destratification)	Entrainment of hypolimnion TP into epilimnion during fall and winter destratification	Induced surface bloom of *Planktothrix rubescens*	Nürnberg et al. (2003)
Schlachtensee, Germany	Mesotrophic $z_{max} = 9$ m area = 0.42 km^2	Hypolimnetic withdrawal (+external load control)	Successful P release decrease initially from redox-sensitive P adsorption, very stable stratification with little exchange with epilimnion	Minor effect on phytoplankton and cyanobacterial biomass	Schauser and Chorus (2009)
Ford, USA	Eutrophic, $z_{max} = 11$ m area = 4 km^2	Hypolimnetic withdrawal	Increase DO	Elimination of *Aphanizomenon* bloom, no change in *Microcystis* biomass	Lehman et al. (2009)

Table 1 continued

Lake	Lake characteristics	Method used	Effect on internal P load	Effect on cyanobacteria	References
Several lakes		Hypolimnetic withdrawal	TP much reduced in epilimnion	Reduced cyanobacterial biomass	Nürnberg (2007)
			TP in epilimnion reduced but still >50 µg/L	No change in cyanobacterial biomass	
			No change in epilimnetic TP	Increase cyanobacterial biomass	
Mauensee, Switzerland	Eutrophic, dimictic $z_{max} = 9$ m area $= 0.5$ km^2	Hypolimnetic withdrawal	Reduced metalimnetic P flux	Decline of *Planktothrix rubescens*	Gächter (1976)
Bled, Slovenia	Eutrophic, monomictic $z_{max} = 30$ m area $= 1.4$ km^2	Hypolimnetic withdrawal (+external load control)	Reduced metalimnetic P flux	Reduced Chl *a* biomass, Decline of *Planktothrix rubescens*	Vrhovsek et al. (1985)
Trummen, Sweden	Hypereutrophic initially, polymictic $z_{max} = 2$ m area $= 1$ km^2	Dredging (after external load control)	Much reduced TP concentrations	Increase in phytoplankton diversity, shift from cyanobacteria to other groups, disappearance of *Planktothrix agardhii*	Bjork et al. (2010)
Vajgar, Czech Republic	Hypereutrophic, polymictic $z_{max} = 2$ m area $= 0.4$ km^2	Dredging (no external load control)	Internal load reduced but no external load control	Improvement during first year after dredging than return to initial state	Pokorny and Hauser (2002)
Groβer Muggelsee, Germany	Eutrophic, dimictic $z_{max} = 8$ m area $= 7.4$ km^2	Dredging	Reduction of P release from dredged cores due to progressed diagenesis, P release from P supply by sedimentation	No improvement in cyanobacteria (*Limnothrix, Planktothrix, Aphanizomenon*)	Kleeberg and Kohl (1999)
Taihu, China	Hypereutrophic, polymictic $z_{max} = 2.6$ m area $= 2\,250$ km^2	Dredging (after external load control)	Reduction of P release by 50 % from cores with lower OM content	No explicit response but positive response suggested	Zhong et al. (2008)
Geerplas, the Netherlands	Eutrophic, polymictic $z_{max} = 2$ m area $= 0.28$ km^2	Dredging (+external load control)	Decrease in lake TP concentrations	No strong effect as phytoplankton is nitrogen limited	Van der Does et al. (1992)

phytoplankton biomass decreased and cyanobacteria were replaced by chlorophytes. This was explained by the overall reduction of P availability likely due to the reduction of the overall amount of the nutrients released from the sediments. Hypolimnetic pure oxygen was added year round during 1988–1993 to improve the water quality of Amisk Lake, a moderate-sized deep eutrophic lake in Alberta Canada ($z_{\max} = 34$ m) (Prepas and Burke 1997). Pretreatment hypolimnetic TP concentrations in summer reached 120 µg/L and Chl *a* 36 µg/L with the dominance of the cyanobacteria genera *Aphanizomenon* and *Anabaena*. During operations, DO remained above 5 mg/L in the hypolimnion and summer TP concentrations dropped to less than 50 % of pretreatments concentrations. This was achieved without disturbing the stratification but warming of the hypolimnion reached up to 3.6 °C. Epilimnetic TP decreased to 87 % of pretreatment mean summer values down to 27 µg/L and Chl *a* to 45 % down to 7 µg/L. Total phytoplankton biomass during summer plankton community shifted from cyanobacteria to a mixed assemblage of diatoms and cyanobacteria (Webb et al. 1997). The use of hypolimnion oxygen delayed the stratification onset and established a diatom population to persist longer and removed P from the water column so that less TP in the epilimnion became available for cyanobacterial growth. The low P release in Amisk Lake and similar hardwater lakes was due to low Fe:P ratio resulting from the lack of iron from the precipitation of Fe by sulfides as ferrous sulfides. Indeed, the iron to P molar ratio in the hypolimnion is 0.03, whereas a ratio of 2 is required for the adsorption of P to iron in oxic water (Manning et al. 1994). On the contrary, Burgi and Stadelman (2002) reported on unsuccessful responses with long-term phytoplankton composition changes in Lake Sembach, Germany, before and after reduction of external and internal loads implemented in 1984 when the lake was highly eutrophic. Improved oxic conditions decreased the remobilization of P from sediments and TP in the epilimnion during spring overturn decreased to 42 µg/L. General trends have shown decrease in dinoflagellates and chlorophytes and increase in cyanobacteria and diatoms with increase in carnivorous crustaceans and decline of herbivorous crustaceans. Zooplankton also migrated to deeper layers lowering the grazing pressure on algae. Surprisingly, Gächter and Müller (2003) concluded based on 10 years of data from the same lake that increased DO concentration from aeration did not reduce the P release from sediments during summer. They showed that irrespective of oxic conditions in the hypolimnion the sediment/water interface remained anoxic due to unchanged high sedimentation rates of organic matter, which could explain why oxygenation did not increase the P retention capacity of the sediment. The chemical characteristics of the sediments (very high P, vivianite) also contributed to prevent restoration success. Similarly, Schauser and Chorus (2009) reported no effect on phosphorus reduction and phytoplankton biomass in Lake Tegel, Germany, following several years of hypolimnetic aeration. They concluded that the improvement was entirely due to the reduction of external loading. In the same lake, (Lindenschmidt and Hamblin 1997) had demonstrated in a modeling study that the aerators used were in fact destratifying the water column and therefore enhancing the phosphorus flux to the epilimnion. In a comprehensive report, Chorus and Schauser (2011) highlighted that Lake Tegel P from sediments originated from the mineralization of recently sedimented organic matter acted as a sink annually but seasonally as a source and that redox-sensitive adsorption to Fe played a negligible role. Finally, partial and full lift hypolimnetic aeration were used in Medical Lake, USA ($z_{\max} = 18$ m), to improve water quality after a return to eutrophic conditions (Soltero et al. 1994). While both systems in place for 3 years during the growing season from spring to fall reduced TP concentrations in the hypolimnion to one-third of pretreatment values (with max of 50 µg/L), they also increased hypolimnetic temperature and had no effect on Chl *a* concentrations with averaged values around 5 µg/L, or on phytoplankton biomass. They attributed this lack of response to an undersized system in light of the increase biological oxygen demand (BOD) during aeration.

In summary, hypolimnetic oxygenation and aeration have generally increase DO in the hypolimnion even if it has increased the BOD. However, the reduction of P loading has not always reached expectations partly due to a lack of iron to bind all the P released. When P released was reduced, cyanobacteria biomass was always reduced and more so when hypolimnetic aeration/oxygenation did not entrain hypolimnetic waters into the epilimnion. Unsuccessful cyanobacterial responses reported have come from (1) aerators inadequately sized to account for increased BOD demand or such that increased diffusion of P into the epilimnion resulted in an enhanced cyanobacterial growth, (2) lack of iron to bind released P, (3) lack of external load control and (4) lack of sufficient time of operation. If these limitations are overcome, hypolimnetic aeration should reduce hypolimnetic phosphorus accumulation and internal cycling and ultimately reduce the development of cyanobacterial blooms.

Ecological impacts on other organisms

Adverse biological effects resulting from aeration have also been reported. The supersaturation of hypolimnetic water with N_2 that might lead to a gas bubble disease in fish is a possible problem in some cases (Kortmann et al. 1994). Some biological beneficial effects also result from aeration. In particular, aeration allows zooplankton to swim deeper into the lake where they can hide from predators in the dark bottom during the day (McComas 2002). Additionally, the expanded aerobic environment should enhance growth and expansion of coldwater fish. In Lake St Mary, Canada, ecological benefits of hypolimnetic aeration included expansion of fish habitat from increased oxygen concentrations, increased visibility and enhanced density of zooplankton (Rieberger 1992). In Amisk Lake, crustaceous zooplankton increased in the deep waters, densities and biomass of benthic macroinvertebrates increased and Cisco expanded their distribution to deeper depths than untreated basins (Webb et al. 1997). To our knowledge, no studies have been reported on other biological organisms including bacteria.

Hypolimnetic withdrawal

Working mechanism

This technique is often used in reservoirs discharging water for power generation or which are dammed near the surface and allow bottom release waters. Although hypolimnetic withdrawal is generally used to control the temperature of the outflow from reservoirs through outlets at different depths, it can also be used in lakes as a control strategy for limiting cyanobacterial blooms through reduced phosphorus internal loading. Hypolimnetic withdrawal relies on the density stratification, which enhances the stability of the water column. It involves the removal of phosphorus from the hypolimnion with the water removal or occasionally its treatment before been returned to the lake. The decrease in phosphorus concentration is then thought to limit the growth of cyanobacteria, particularly in lakes or reservoirs where the internal phosphorus load dominates. This method ensures that bottom water residence time is reduced, as well as the availability of nutrients to the epilimnion by limited vertical mixing. However, hypolimnetic withdrawal has a tendency to store heat as it expands the epilimnion heated by solar radiation (Kennedy 1999) and as such decreases the thermal stability of the water column which can promote vertical entrainment of nutrients in the epilimnion (Effler et al. 1986). The amount of water withdrawn is limited to prevent excessive water-level variations when high-quality inflow is not available. Timing of hypolimnetic withdraw is best to coincide with higher summer internal P loads and before the onset of destratification in the fall.

There are several types of systems based on either passive siphoning (relying on the pressure head between the lake and the pipe outlet) or active pumping, with withdrawal depth very close to the bottom in the deepest area. This restoration method is mostly applied to small surface area lakes and to deep lakes (usually at least seasonally stratified) (Nürnberg 2007). The advantage of this method is relatively low capital cost and potentially long-term effectiveness. Costs estimated by Hickey and Gibbs (2009) range around \$400 k and \$10 k per treatment. Similar costs have been reported for systems in US lakes (i.e., \$420 k for Lake Ballinger, \$310 k for Lake Devil and \$282 k for Pine Lake) by Cooke et al. (2005). Relatively low costs of operating and maintenance are an advantage of this method.

Successful and unsuccessful examples

Successful hypolimnetic withdrawal installation has been documented in the USA, Canada, Germany, Finland, Poland and Australia (Nürnberg et al. 1987; Dunalska et al. 2007; Keto et al. 2004; Cooke et al. 2005; Sherman et al. 2001). Maximum hypolimnetic TP concentration as well as epilimnetic TP concentration decreased in almost all the studied lakes. The longer the withdrawal was operating, the greater was the change in epilimnetic TP concentrations. Schauser and Chorus (2007) reviewed the performances of both external and internal control measures in two Berlin lakes. In Schlachtensee, an Olszewski pipe was operated yearly for a few weeks since the eighties to withdrawal the P-enriched hypolimnion water at the end of the stagnation period. The comparison between the P content in the lake, the external P load and the P withdrawn from the hypolimnion showed that the hypolimnion withdrawal had in most years only a minor effect on the total P budget of the lake (Schauser and Chorus 2009) and on the phytoplankton and cyanobacteria biomass (Chorus and Schauser 2011). The redox-sensitive P desorption from the sediments was found to be the main source of P in the hypolimnion. However, the strongly stable summer thermal stratification largely kept the desorbed P in the hypolimnion with minimal transfer to the epilimnion and therefore not directly accessible for phytoplankton growth. They reported a shift from cyanobacteria dominance to insignificant contribution when summer epilimnetic TP < 25 µg/L. Interestingly, they also clearly demonstrated that phytoplankton biomass decreases only below a threshold of TP $= 50$ µg/L when phytoplankton switched from light limitation to phosphorus limitation. Lehman et al. 2009 reported on the use of hypolimnetic withdrawal as a control method for cyanobacteria. They presented the growth response of *Aphanizomenon flos-aquae* and *Microcystis aeruginosa* in Ford Lake, Michigan, USA, following three very short term (1 week), hypolimnetic withdrawals, where they replenished oxygen by forced advection and destabilized the water column. Ford Lake is a eutrophic lake ($z_{\max} = 11$ m) supporting warm water fisheries, and lake manipulation was intended to lengthen the spring diatoms dominance and limit summer cyanobacteria growth. Typically *Aphanizomenon flos-aquae* dominate from July to early August when it is replaced by *Microcystis aeruginosa* through to September. Their results show that withdrawal of bottom water (at 11 m depth) led to higher NO3, an extended diatom bloom and the elimination of the *Aphanizomenon* bloom. It did not, however, decrease *Microcystis* biovolume. Nürnberg (2007) provided a comprehensive review of general patterns of lakes responses to hypolimnetic withdrawal applications (in 40 European and 8 North American lakes). The evaluation in terms of phosphorus concentration compared before and during treatments revealed a decrease in both hypolimnetic and epilimnetic summer TP with reduced differences between the concentrations in both layers. The evaluation of Chl *a* concentration changes was less often reported, and only 6 lakes presented explicit results. Among those, only 3 showed reduced biomass (associated with much reduced TP in the epilimnion), two unchanged (with moderate changes in epilimnion TP and still values higher than 50 µg/L) and one increase (associated with no TP change in epilimnion). Gächter (1976) suggested that the interruption of the metalimnetic flux of nutrients under hypolimnetic withdrawal was responsible for the disappearance of *Planktothrix rubescens* in Lake Mauensee, Switzerland. A similar decline of *Planktothrix rubescens* was also observed in a Slovenian lake as a result of reduced metalimnetic P flux from hypolimnetic withdrawal (1980–1982) together with reduction of external loads from sewage diversions (1982–1985) (Vrhovsek et al. 1985). Therefore, all these case studies suggest that the metalimnetic P flux is responsible for the intensity of the *Planktothrix rubescens* blooms observed.

In summary, hypolimnetic withdrawal has generally decreased the internal P loading at the sediment water interface but not always controlled cyanobacterial blooms (Table 1). In each case study, understanding the reasons of failure (and success) is crucial. Unsuccessful cyanobacterial responses presented here have resulted from (1) lack of external control, (2) TP concentrations above threshold for P limitation, (3) decreases in water column stability leading to bottom nutrient-rich waters to be entrained into the epilimnion.

Adverse effects on water quality and other organisms

There are several drawbacks to hypolimnetic withdrawal. Problems of large amplitude internal waves in reservoirs causing water quality problems downstream have been reported. In particular, a well-documented incident occurred in a Sydney water supply reservoir in 1998 where interface excursions of 35 meters created a 70-day supply shutdown following a water quality problem of *Cryptosporidium* contamination following large rainfall runoffs (Cox et al. 2003). Negative biological effects might also take place downstream due to the discharge of water with a lower temperature, higher nutrients, ammonia, hydrogen sulfide or other toxic compounds. To avoid this negative impact, mixing with epilimnetic water might be employed (Cooke et al. 2005). These adverse effects can also be attenuated by additional chemical phosphorus precipitation in the outlet (Chorus and Mur 1999). If low DO is expected in the outlet, then aeration equipment could be installed. Drawbacks include strong odor caused by hydrogen sulfite, lake water-level drop, if water treated, increase in temperature which may destabilize the water column causing mixing with surface water and transport of phosphorus from the hypolimnion to surface waters.

Methods that directly treat the source of the internal nutrients loads (i.e., the sediments) have also been widely used. We report on a non-chemical method of sediment treatments: sediment dredging.

Sediment dredging

Working mechanism

The primary aim of sediment dredging is to remove the surface sediment layer rich in toxic pollutants (i.e., heavy metals (Garcia-Orellana et al. 2011), or organic pollutants (Martinez and Hornbuckle 2011; Weston et al. 2002), thus decreasing their release from the sediments to the water column. In the last two decades, sediment dredging has also been used to reduce nutrient and cyanobacterial inoculum (Zhong et al. 2008; Kleeberg and Kohl 1999).

Successful and unsuccessful examples

Examples of both successful and unsuccessful control of cyanobacteria by sediment dredging have been reported in the literature and are summarized in Table 1. A typical example of successful restoration took place in Lake Trummen (Vaxjo, Sweden) with cyanobacterial control by sediment dredging. Until 1920, the lake was oligotrophic, but the increase in domestic sewage and industrial wastewater inputs resulted in progressive eutrophication followed by formation of cyanobacterial blooms, fish kills and oxygen demand (Eiseltova 1994). During the years of 1957–1958, the external sources of nutrients were restricted, but without expected effects with respect to water quality. Thus, Lake Trummen was dredged in 1970–1971. After this radical action, the cyanobacterial blooms disappeared and were replaced by a species-rich plankton community. The ecosystem as a whole recovered to a functional unit, characterized by a balance between production and decomposition, and Lake Trummen became available for a variety of recreation activities (Bjork et al. 2010).

On the contrary, unsuccessful remediation action took place in Vajgar fishpond, Czech Republic (Pokorny and Hauser 2002). Sediment removal from the Vajgar pond (44 ha) started in 1991 and finished in 1992. During this period, a total amount of 330.000 m^3 of sediment was removed (whole cost was approximately \$850 k). While during the first year after the removal of sediments, phytoplankton dynamics showed some signs of improvement; 2 years later, phytoplankton structure and dynamics returned to a state as that before the year of 1991. Indeed, although the internal load of Vajgar pond was reduced by the sediment removal, the external loading remained high and there was still regular import of phosphorus and phytoplankton from the intensely managed fishponds in the upper pond catchment (Pokorny and Hauser 2002). Kleeberg and Kohl (1999) reported up to 99 % reduction of P fluxes from dredged cores of 50 cm compared to intact cores under both aerobic and anaerobic conditions from a shallow eutrophic lake Groβer Muggelsee, Germany. In this lake, despite efforts of external load reductions, large cyanobacterial blooms of *Limnothrix redekei*, *Planktothrix agardhii* and *Aphanizomenon flos-aquae* appear every summer. The reduction of P release in the cores was not due to breakdown of organic matter with depth or reduced TP concentrations but to progressed diagenesis. Hence, they suggested that as P fluxes from sediments were mainly determined by P supply via sedimentation, a reduction of external P loading would be more efficient than dredging.

Lake Taihu is a very shallow hypereutrophic lake in China ($z_{mean} = 2$ m) that frequently exhibits serious cyanobacterial blooms in summer (Pu et al. 1998; Pearl et al. 2015). Although external loads have been considerably reduced in recent years, eutrophication in Lake Taihu has not decreased as internal loading is delaying the effect (Jiang et al., 2008). The influence of sediment dredging of the top 30 cm of sediments from Lake Taihu, China, on the release of phosphorus was tested in a laboratory incubation experiment (Zhong et al. 2008). The sediment cores of the dredged sediments at monthly interval between December 2005 and June 2006 had a lower release potential (by 50 %) compared to the controls over a period of 6 months at temperatures close to in situ values. The porosity of the dredged cores was lower due to compaction with depth, and organic matter content was also lower. The authors suggested that dredging was a possible measure to control cyanobacterial blooms in Lake Taihu after external loading was efficiently reduced. Van der Does et al. (1992) also reported on the successful restoration of shallow lake Geerplas, the Netherlands, by dredging. The lake was characterized by a high internal loading, a reduction of external loading alone not sufficient for recovery. A removal by suction of the mud layer (from 1 to 100 cm) was achieved together with the positioning of screens to reduce horizontal mud transport during two attempts in 1989 and 1990–1991. Phosphorus concentrations in the lake decreased to 0.1 mg/L. A simultaneous reduction in 1990 of external loading from 0.38 down to 0.13 P m^{-2} y^{-1} did, however, hide the real cause of improvement.

Often studies on dredging have reported responses associated with combined restoration methods. The restoration of shallow Lake Cockshoot Broad, UK, started in the 1980's. First, the lake was isolated from the sewage effluent contaminated River Bure in 1982, followed by sediment dredging. Sediment removal was combined with other in-lake methods such as local fish community removal, which resulted in chlorophyll-a concentration decreases and aquatic plants re-colonization (Moss et al. 1996). Sediment dredging was combined with a Phoslock® addition in an enclosure experiment in the Netherlands and resulted in reduced cyanobacteria, microcystin and soluble reactive phosphorus concentrations (Lurling and Faassen 2012).

In summary, dredging is a costly restoration measure. In addition to the cost of dredging at the site, dredging also raises the problem of sediment disposal and treatment, both aspects which are also costly and impact on the environment (Peterson 1981). Treated sediments can, however, be used as fertilizers, and the benefit gained from sales after treatment can slightly offset the overall cost. Disposal sites often require hard to obtained permits, especially in lowlands, and require chemical and toxicity testing of the disposed material (Cooke et al. 2005). Dredging should be selected only if internal loading is the dominant nutrient source, if sedimentation rates are low and if the depth associated with high concentration of labile P is known. All the case studies reporting unsuccessful control of cyanobacteria by dredging came from observations that external loading was not concomitantly controlled. Most successful cyanobacterial responses after dredging have been associated with other combined restoration methods (fish removal, inactivation by Phoslock, external control).

Adverse effects on other organisms

The environmental impact of dredging on fauna and flora in sediments is discussed frequently. Basically, disturbance of sediments may act against living organism in two ways: (a) through the physical changes (changes in gravel and organic matter content, direct removal of organisms, etc.) and (b) physicochemical changes since the disturbance of contaminated sediments may remobilize sediment-bound contaminants into the water column where they potentially impact a wide range of species (Burton and Johnston 2010). If a lake is dredged completely, 2–3 years may be required to reestablish benthic fauna (Cooke et al. 2005). Dredged cores have reduced significantly the benthic macrofauna diversity and density (Lewis et al. 2001). Changes in gravel content on the sensitivity of fauna communities have been reported by Cooper et al. (2011) who showed that the sensitivity of macrofaunal communities may increase as both the proportion of gravel increased and the level of natural physical disturbance decrease. Of course, the effects may vary with different communities colonizing different sites. For example, the study of Bolam et al. (2011) investigated in a field experiment the vertical migratory capability of temperate macroinvertebrate species. They found that the survival following sediment burial varied significantly across species. Although the release of toxic substances from resuspended sediment may occur, toxicological responses in the field appear to be rare. On the other hand, the accumulation of contaminants into the living systems is likely even for several months after dredging and disposal (Roberts 2012).

Conclusions and recommendations

The mechanisms responsible for the release of phosphorus from sediments are lake specific, but the dominant process in shallow wind exposed lakes is often resuspended while in iron-rich deeper lakes it is redox-sensitive release. Although accurate quantification of internal phosphorus loads is difficult to determine due to the many factors and processes, which are involved, numerous restoration techniques have been developed and tested.

The selected case studies reporting on both successful and unsuccessful responses to different physical restoration methods, summarized in Table 1, have demonstrated that an essential prerequisite for long-term success is the detailed understanding of the dominant processes at work and their spatial and temporal variability. Indeed, all the unsuccessful applications were associated with insufficient initial knowledge leading to inappropriate method selection or undersized system.

We therefore recommend the following criteria before choosing among the physical restoration methods:

1. Define the critical phosphorus level needed to achieve the predicted outcome
2. Assess the dynamics and relative role of internal loading compared to external loading
3. Assess the sediment characteristics to determine whether internal loading can be controlled
4. Quantify the link between internal load and cyanobacterial biomass
5. Scale the restoration method as a function of the internal load and the size of the lake
6. Evaluate the potential to cause adverse effects to aquatic biota
7. Set a long-term monitoring program before, during and after the restoration

The above criteria are agreement with those proposed by Schauser et al. (2003) and Hickey and Gibbs (2009). Schauser et al. (2003) presented a decision support tool for the selection of an appropriate in-lake method to reduce internal P load. They highlighted that selection criteria should include the evaluation of the magnitude of the external versus internal load, the dynamics of the internal load and the lake morphology. Another decision support framework to choose appropriate management options for reducing internal loads, and thus cyanobacterial blooms, has been provided for New Zealand lakes that are more windy and mix more deeply than continental lakes (Hickey and Gibbs 2009). These authors also show that the choice is a function of the lake and sediments characteristics and of the remediation method characteristics. They also highlighted the need to evaluate the potential to cause adverse effects to aquatic biota. When a lake is being restored, we also recommend a long-term physical, chemical and biological monitoring program including phytoplankton, zooplankton, fish and benthic fauna. Indeed, a lake response to reduction in internal load will take time and will only be effective once external loading is also controlled. Moreover, a cyanobacterial response to reduced loading will only be effective below a threshold associated with cyanobacterial growth switching from light to nutrient limited. Long-term monitoring is required as climate variability together with climate changes will influence the success of the method selected.

Reviewing the different physical methods commonly used to control internal phosphorus release from sediments, we have summarized the applicability, the effectiveness in decreasing internal P load, the cost and the ecological impact on biota in Table 2. Hypolimnetic aeration and oxygenation are a short-term effective method in deep stratified systems where complete destratification is not recommended. It has a relatively high operating cost associated with the generally continuous mode of operation for successful applications. The ecological impacts are mostly beneficial for both zooplankton and fish habitat. Hypolimnetic withdrawal is a long-term low capital and low operating costs method appropriate for small deep stratified systems. It requires discharged water to be partly compensated by high-quality inflow to avoid problems associated with water-level variations and discharged water to be treated to avoid downstream water quality problems of low DO, high reduced elements and nuisance odors. Sediment dredging is a short term, extremely high cost (of removal, disposal and treatment) restoration method and therefore only practically applicable in small and shallow systems. While it directly removes the cyanobacterial inoculum and reduce the labile P content of the sediment, it can have negative impacts on benthic organisms. We conclude that each method has its strength and weakness depending on the system considered and that a combination of restoration methods is often more successful than a single method. It is important to keep in mind that internal phosphorus release is likely to increase from expected climatic changes. Control methods should therefore include expected increase in internal loading in their applicability and potential effectiveness.

Table 2 Comparison of widely used physical methods to control internal phosphorus load

Method	Applicability	Effect on internal P load	Cost	Ecological impact
Hypolimnetic aeration and oxygenation	Require large hypolimnion, avoid in shallow systems <15 m	Highly dependant on proper design and sizing in relation to BOD and sediment Fe sorption capacity	High mostly operating costs, dependant on type of equipments, power rates	Mostly beneficial impact Increase fish habitat and zooplankton density
Hypolimnetic withdrawal	Require small deep stratified system, avoid in shallow systems, require discharged water to be replaced	In most cases, long-term solution	Low capital and operating costs	Impacts associated with decrease in water level Adverse impact on downstream water quality (low DO, high P, high NH4, hydrogen sulfite and metals), nuisance odors
Sediment dredging	Only feasible in relatively small systems (wetlands, ponds, harbors)	Shallow dredging might expose fertile sediments, deep dredging might expose unwanted toxic substances	High capital costs depend on hydraulic or mechanical dredging plus disposal and treatments	Exposition of unwanted toxic substances including heavy metals, release of sediment-bound contaminants, high disturbance of sediment bacterial communities and fauna, removal of cyanobacterial inoculum

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