Aquat Ecol (2016) 50:541–576
DOI 10.1007/s10452-016-9593-0

How to combat cyanobacterial blooms: strategy toward preventive lake restoration and reactive control measures

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Received: 12 May 2015 / Accepted: 23 July 2016 / Published online: 11 August 2016

Abstract Water managers worldwide are facing the serious problem of dense blooms of cyanobacteria in surface waters. In the quest for the optimal method to combat cyanobacterial dominance, many questions and many possible solutions arise. This paper presents a three-phase strategy to fight harmful cyanobacterial proliferation. Water managers who apply this strategy can generate a tailor-made set of measures. The three phases consist of (1) defining the ecological water quality targets of the waterbody in question, (2) assessing its current ecological state, and (3) selecting which measures will yield optimal results. The paper provides assistance in the quantitative diagnosis of the state of both shallow and deep temperate freshwater lakes by means of a lake system analysis, and presents a survey of measures. Measures are divided into two sets. Preventive measures are based on switching to a clear state (shallow lake) or reducing the overall cyanobacterial biomass (deep lake). They are subdivided into nutrient reduction, hydromorphological adjustments, and food web management. Concerning nutrient reduction, in many situations phosphorus management alone should suffice. Nitrogen management can be important to increase the species diversity of macrophytes, or to relieve downstream (marine) consequences. Control measures (including mitigation) have a direct impact on cyanobacterial blooms by biomass removal, flushing, or mixing; however, the number of proven technologies is limited.

Keywords Lake system analysis · Lake assessment · Phosphorus loading · Dual nutrient management strategy · Alternative stable states · Shallow and deep lakes

Guest editors: Petra M. Visser, Bas W. Ibelings, Jutta Fastner & Myriam Bormans/Cyanobacterial blooms. Ecology, prevention, mitigation and control.

Electronic supplementary material The online version of this article (doi:10.1007/s10452-016-9593-0) contains supplementary material, which is available to authorized users.

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Introduction

Dominance of cyanobacteria in phytoplankton communities creates undesirable situations in fresh, brackish, and salty waters worldwide. Dense cyanobacterial blooms may lead to serious health consequences for cattle, wildlife, pets, and humans (Carmichael 1992; Paerl et al. 2001; Huisman et al. 2006). Such blooms prevent the full use of the affected waterbody; for example, cattle cannot drink the water, recreation is prohibited, and water intake for drinking water production is hampered. Climate change may lead to an increase in cyanobacterial blooms in the near future (Paerl and Huisman 2009). Water managers have to act to combat the causes and consequences. The question arises where and how to start the action, what measures are available, and which measures are the most suitable. The set of possible measures to reduce these harmful blooms range from technical applications based on the direct reduction in cyanobacterial biomasses or scums, to intervention in a lake's ecological state. There are limits and caveats to every approach. Cyanobacterial scum layers in small ponds obviously require a different approach compared to cyanobacterial blooms in reservoirs, large lakes, or marine deltas. The type of measure can be defined based on the targets that need to be met, the time frame, the available budget, the influences of different stakeholders, and the set of feasible measures.

Numerous projects have failed to achieve clear waters without cyanobacterial dominance (Søndergaard et al. 2007). To our knowledge, only a limited number of publications on quantified lake assessment also explain why certain measures are chosen to implement. This paper can help lake managers to apply an assessment strategy and select the optimal measures, based on an ecological and technical approach. All types of waterbodies are addressed as "lakes" unless stated otherwise. The procedure presented is based on small- to medium-sized (an area of up to several km^2), lentic, fresh to mildly brackish, temperate inland waters. Both shallow polymictic lakes and deeper monomictic and dimictic lakes are covered.

Implementing the strategy

The strategy consists of three phases (see Fig. 1): (1) defining the ecological water quality targets for the lake in question, (2) assessing its current ecological state by means of a lake system analysis and (3) selecting which measures will yield optimal results. Two main approaches to avoid cyanobacterial proliferation or their unwanted effects are differentiated, namely taking preventive measures and taking control measures. Preventive measures are aimed at achieving long-term improvement of a lake's aquatic ecosystem. Control measures are used to reduce (the nuisance of) cyanobacterial blooms in a direct manner. The latter measures are not designed to improve the ecological or biochemical water quality as such. There is a thin line between control and mitigation. Here they are collectively identified as control measures. Both preventive and control measures should be explored in order to select the most suitable measure(s) in terms of efficacy, side effects, and feasibility.

Parts of this strategy are supported by a knowledge base that is currently under construction: the Lake Ecological and Cyanobacterial Diagnosis and Measures (LECoDaM) knowledge base. LECoDaM displays the two approaches (collapsed) mentioned above: control and prevention (Fig. 2). It contains measures, described in terms of leading principle, preconditions, potential risks, efficacy, implementation, side effects, and cost. Supplementary, the "Nutrients" block supports the quantification of nutrient loading calculations for lake assessment purposes (see "Lake system analysis" section).

Ecological targets

The ecological water quality targets to be met (phase 1, Fig. 1) depend mainly on the function(s) of a lake:

- *Bathing and recreational water*. For recreational use, toxins and particularly cyanobacterial scums pose risks of illness and mortality. Cases of illness and mortality of cattle and pets have been reported worldwide (Chorus and Bartram 1999; Codd et al. 2005; Paerl and Huisman 2008; Faassen et al. 2012; Lürling and Faassen 2013). Incidents with human exposure to cyanobacteria in recreational/bathing waters and subsequent serious health problems have rarely been reported. The EU bathing water regulation specifies that exposure to cyanobacterial blooms "should be avoided"—a course of action that is open to interpretation. National and regional regulations differ worldwide and can be based on

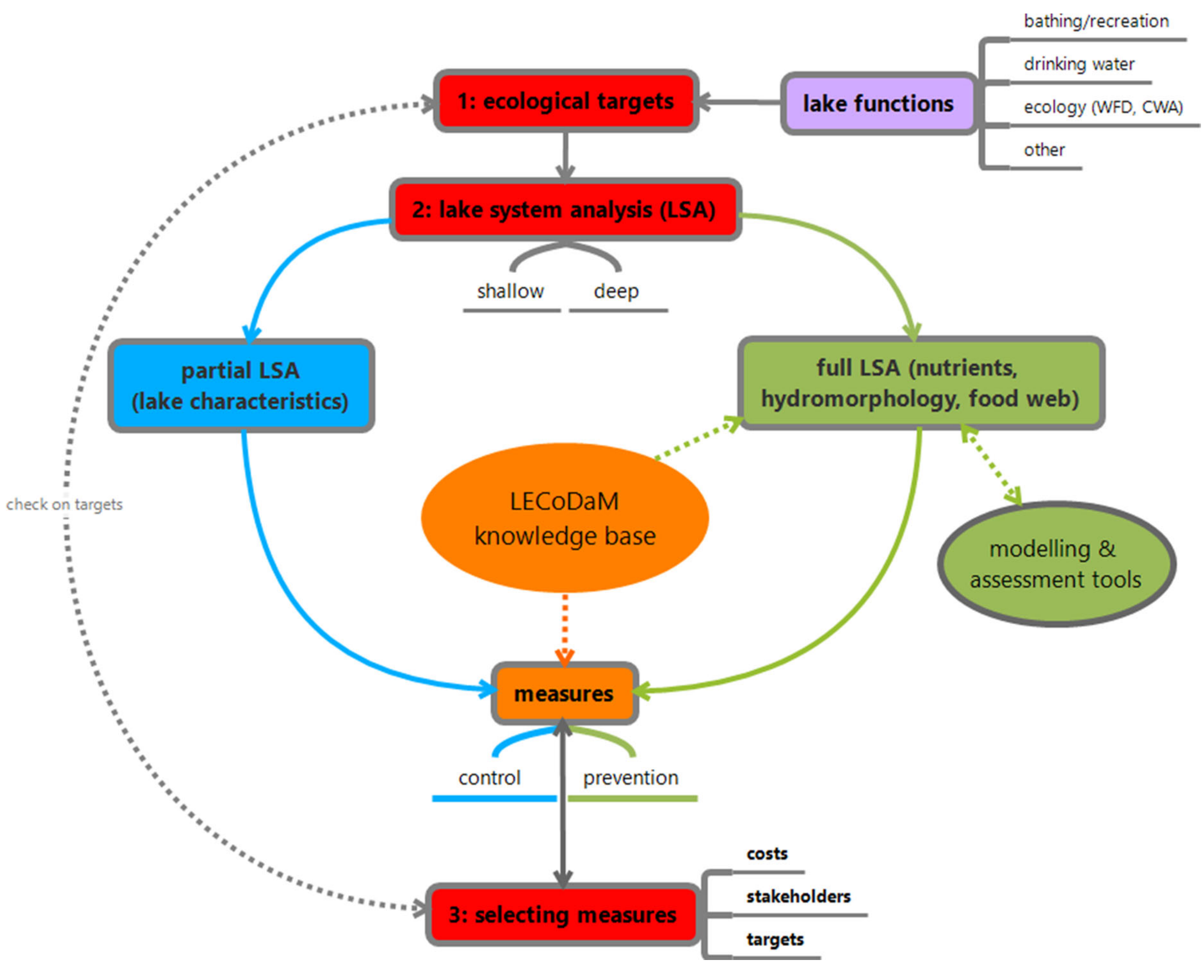

Fig. 1 Three phases of the strategy toward preventive lake restoration (*green/right*) and reactive control measures (*blue/left*)

cyanobacterial cell counts, biomass, or toxicity (Chorus 2012; Ibelings et al. 2014).

- *Drinking water*. A provisional guideline value of 1 μg l^{-1} has been adopted by the World Health Organization (WHO) for microcystin-LR in drinking water (Chorus and Bartram 1999). The potential toxin production of cyanobacteria makes cyanobacterial dominance in drinking water reservoirs undesirable, since it might require an extra purification process to degrade the cells and toxins, for instance by ozone or chlorination (see "Biomass removal" section), or peroxide and/or UV treatment (Cornish et al. 2000; He et al. 2012). Moreover, large cyanobacterial biomasses may cause anoxia near the sediment of stratified reservoirs, which could induce the internal loading of several toxic metals [e.g., manganese (Du 2004)], threatening drinking water quality.
- *Ecology*. Targets might be set by national or regional legislation, for example the European Water Framework Directive (WFD) or the US Clean Water Act (CWA). In general, biodiversity improves ecological values. Especially, cyanobacterial dominance is considered a symptom of poor water quality.
- *Other*. Another lake function can be fishery, which might have targets that conflict with the above-mentioned targets. The cyanobacterial blooms experienced by civilians might force water managers to specify targets like preventing odor from scums or anoxia in urban ponds.

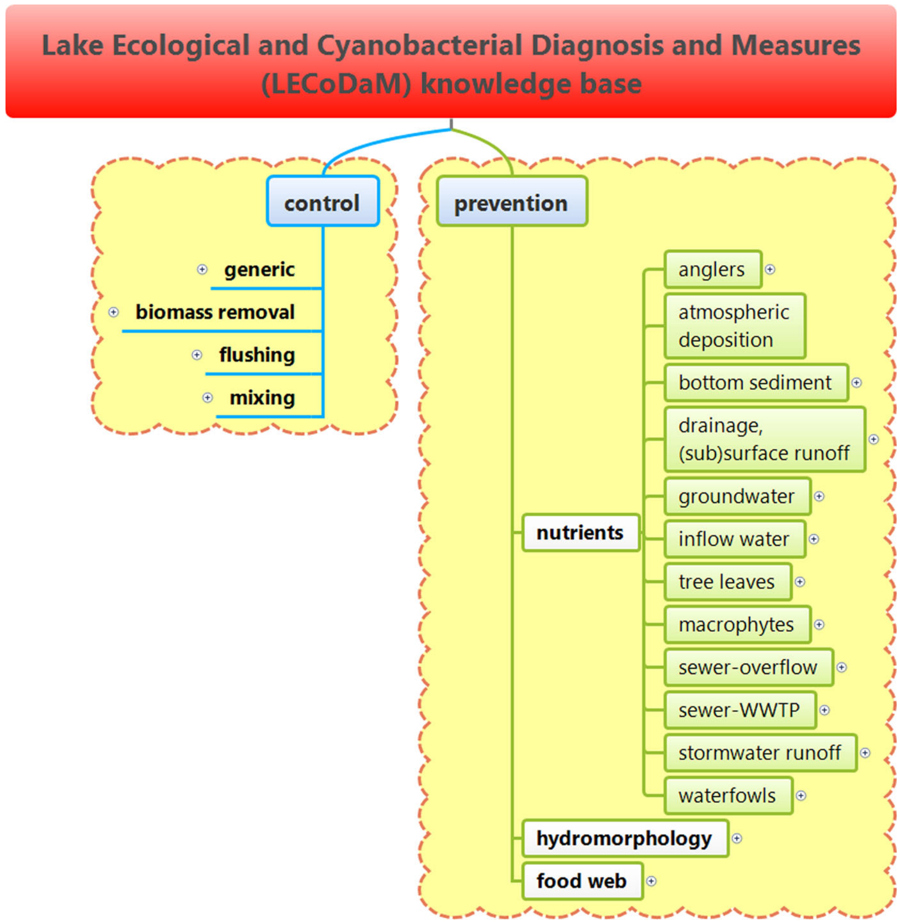

Fig. 2 Structure of the LECoDaM knowledge base in a collapsed form. A "*plus*" sign indicates that a branch can be expanded

Lake system analysis

Scope

A selection from the available spectrum of suitable measures can only be scaffolded by an upfront diagnosis of the lake's aquatic ecology by means of a lake system analysis (LSA, phase 2 in Fig. 1). An LSA can consist of a water balance and nutrient budgets, and information regarding water level management, hydraulic residence time, and other lake specific properties. Defining several preconditions can help to enhance the process toward a LSA and the final selection of measures. Shallow lakes behave different from deep lakes ("Shallow versus deep lakes" section). With respect to nutrients, a single or dual nutrient management strategy can be taken into account ("Phosphorus versus nitrogen" section). Areal nutrient loadings can be quantified ("Quantification of P loading" section) and assessed ("Assessment of P loading" section) in detail.

To improve the understanding of biological interactions, numerous other ecological data might be investigated, strengthening the LSA. These are not explored in this paper. For example, the biomass and community structure of phytoplankton (Brettum and Andersen 2005; Nõges et al. 2010), fish (Benndorf 1990) or macrophytes (Jeppesen et al. 2000), which indicate nutrient limitation and the overall ecological status. Or the light conditions that limit the macrophytes colonization depth. The maximum colonization depth can be estimated with Secchi depth (Canfield et al. 1985; Cooke et al. 2005; Hudon et al. 2000), chlorophyll a (Trolle et al. 2008), extinction coefficient (Hudon et al. 2000) or relative surface irradiation (Lacoul and Freedman 2006). Or the zooplankton-to-phytoplankton ratio, indicating whether zooplankton is counterbalancing phytoplankton (Mehner et al.

2008; Beklioglu et al. 2011). Or the zooplankton size distribution, which indicates a fish community structure based on the fact that planktivorous fish are size-selective feeders with a clear preference for larger zooplankton prey (Benndorf 1990, 2005). Or the presence of bivalves like dreissenid zebra and quagga mussels, which are very effective in clearing lakes (Nalepa and Fahnenstiel 1995). The use of dreissenids in measures is discussed in "Preventive measures" section.

To select optimal preventive measures, a full-scale LSA (full LSA in Fig. 1) is strongly advised. The choice of control measures can be based on less detailed information (partial LSA in Fig. 1). For example, lake characteristics, hydrological management, phytoplankton population composition, bathymetry, and residence time might provide ample information.

Shallow versus deep lakes

In the LSA (phase 2, Fig. 1), a most essential distinction to make is lake depth. Here, shallow (polymictic) lakes are defined as having a mean depth of <3 m, based on lakes that can be largely colonized by macrophytes and that do not stratify for long periods in summer (Scheffer 2004; Burks et al. 2006; Beklioglu et al. 2011). Deep (monomictic or dimictic) lakes are defined as having a mean depth of >10 m, based on the lake's incompletely mixed area compared to the total lake area (Sas 1989). Lakes in between these mean depths are referred to as moderately deep lakes.

In shallow nutrient-enriched aquatic systems, two strikingly different alternative and stable states are frequently observed: A clear water state dominated by benthic macrophytes, and a more turbid state dominated by algae (Scheffer 1990; Janse et al. 1998; Körner 2002; Jeppesen et al. 2005; Smith and Schindler 2009). Many freshwater studies show that regime shifts between these two states can be abrupt (Scheffer et al. 2001; Scheffer and Carpenter 2003). Similar regime changes have been observed in shallow brackish ecosystems (Dahlgren and Kautsky 2004). The clear or turbid state depends on a diversity of ecological factors and stabilizing ecological feedback mechanisms. This ecological resilience is able to keep the system in a clear or turbid system in a wide and overlapping nutrient status (Scheffer 1990; Scheffer et al. 1993; Dokulil et al. 2011). In essence, the hypothesis of alternative stable states is widely accepted and has been proven to be empirically adequate for shallow lakes being small (Scheffer et al. 2001; Scheffer and van Nes 2007) or large (Ibelings et al. 2007; Janssen et al. 2014).

At high nutrient loading in shallow lakes, high turbidity and phytoplankton dominance is an almost certainty. Reducing nutrient loads will not induce a gradual shift to a clear water state, due to ecological resilience. After sufficient reduction in nutrient loading, light conditions will improve due to the reduced algal biomass. The colonization depth for macrophytes will subsequently increase. Higher abundances of macrophytes might enable the food web to make a sudden switch to a stable clear water state, with further stabilizing consequences: Macrophyte-colonized periphyton can take over the primary production from phytoplankton (Strand and Weisner 2001; Vadeboncoeur et al. 2003). Macrophytes also reduce wind fetch-induced turbidity. Macrophytes provide shelter for predator fish, reducing the number of zooplanktivore and benthivore fish species, and increasing zooplankton and thus spring clarity. Spring clarity in its turn will enhance macrophyte germination (Meijer 2000). Macrophytes can produce allelochemicals which might reduce algae growth (Scheffer et al. 1993; Hilt and Gross 2008). On the other hand, macrophytes might enhance denitrification reducing nitrogen levels, enhancing sediment phosphorus release in the process (Hilt et al. 2006; Veraart et al. 2011). The root uptake of macrophytes creates another route of potential internal loading (Welch and Cooke 1995). This might help to induce a cyclic shift from clear to turbid and vice versa (van Nes et al. 2007; Mihaljević et al. 2010).

At low nutrient loading in shallow lakes, there is also one possible stable state: clear, macrophyte dominated, low turbidity. At intermediate nutrient loadings, both options are possible (clear or turbid), due to hysteresis based on lake history and, for instance, the above-mentioned increase in internal nutrient loading. The threshold loading (the "critical" loading, see "Assessment of P loading" section) to switch from turbid to clear is lower than the threshold to switch from clear to turbid conditions.

Deep lakes are in general clearer than shallow lakes due to higher sediment losses (Søndergaard et al. 2005c). Deep lakes are similar to shallow lakes at low and high nutrient loading in terms of planktonic biomass dominance. The influence of fish and macrophytes, though, might be very limited if the littoral zones are marginal compared to the epilimnetic volume of water in the pelagic zone (Jeppesen et al. 1997; Sollie et al. 2008). This minor influence of the littoral food web is the main reason why hardly any nutrient-related hysteresis effects are observed in many deep lakes (Scheffer 2001; Sachse et al. 2014). In typical deep lakes, there are no significant alternate states, but there will be decreasing algal biomass at decreasing nutrient levels in a sigmoid curve (Scheffer 1990; Scheffer et al. 1993). If in deep lakes nutrients are limiting algae growth, lowered internal and external nutrient loadings will result in less algal biomass, though some hysteresis may occur as a result of the overwintering strategy of several cyanobacterial genera in lake sediments, for example genera like *Dolichospermum* (formerly called *Anabaena*), *Cuspidothrix, Aphanizomenon*, *Cylindrospermopsis*, *Gloeotrichia*, and *Nodularia* produce akinetes to overwinter in lake sediments (Kaplan-Levy et al. 2010). Also sunken *Microcystis* cells can overwinter in lake sediments (Verspagen et al. 2005; Misson et al. 2012a, b). Wind and bioturbation by benthivorous fish like cyprinids might induce recruitment from sediment (Verspagen et al. 2004). Thus, high cyanobacterial concentrations leave a trail in their benthic stock. Lake sediment phosphorus release (internal loading) will induce a hysteresis effect if only external loading is reduced (see "Other P sources" section).

In lakes that are moderately deep, both the presence and the absence of distinct alternate states are found. Lake Müggelsee (4.9 m mean, 7.9 m maximum depth) seems to have distinct alternate states (Körner 2001), as does Lake Scharmützelsee, which has a large shallow area (up to 6 m) and a deeper part (9.0 m mean, 29.5 m maximum depth, Hilt et al. 2010). In an analysis of 300 lakes, Körner (2002) showed a more gradual macrophyte abundance in lakes of 5–10 m maximum depth compared to shallower lakes.

Phosphorus versus nitrogen

Reduced nutrient inflows are often a first and essential condition to initiate lake restoration and reduce cyanobacterial dominance in the long term (Benndorf 1990; van Liere and Gulati 1992; Jeppesen et al. 1997). Considering that phosphorous (P) and/or nitrogen (N) are the main limiting nutrients for algae growth, choosing which nutrient(s) to reduce is an important starting point for the LSA. The choice can be based on ecological and financial efficacy. For inland waters, it has been widely accepted that the focus for reducing algae blooms can be on P reduction to switch lakes to a clear state (Sawyer 1952; Vollenweider 1968; Schindler 1974; Rast et al. 1983; Smith 1983; Jeppesen et al. 2002; Schindler et al. 2008). Important aspect is that many cyanobacterial species can use gaseous nitrogen (N_2) as an N source (Reynolds 1984). This N_2 fixing has high energy requirements and might be limited by light availability. Cyanobacteria with N_2-fixing capacities switch immediately to assimilate NH_4 when it is available (Ferber et al. 2004; Finlay et al. 2010; Paerl et al. 2011). But still, inducing an N limitation might shift algae composition toward these N_2-fixing species and thus ineffectively reduce or even promote cyanobacterial blooms. And a single P management strategy has proved to be successful in many restoration programs (Smith and Schindler 2009).

But the value of reducing N loading as additional management measure is subject to debate (Moss et al. 2005; Lewis and Wurtsbaugh 2008; Sterner 2008; Conley et al. 2009; Paerl 2009; Scott and McCarthy 2010; Paterson et al. 2011; Lewis et al. 2011; Paerl et al. 2011; Paerl and Otten 2013; Schindler 2012; Moss et al. 2013; Müller and Mitrovic 2015).

For lake management, it is crucial which nutrient has to be included in the LSA and choice of measures. Important aspects of the debate on N and P in water management are touched upon below. These aspects are the N cycle and measures, nutrient limitation and N:P ratios, macrophyte development, and downstream consequences.

N cycle and measures

The N cycle is complex and has widespread diffusive and atmospheric sources (Carpenter et al. 1998; Robertson and Vitousek 2009; Elser et al. 2010). Sources of N are largely anthropogenic (influent of wastewater treatment plants, agriculture, and fossil fuels), and N can be transported by water, groundwater, or air. Processes controlling N losses to the atmosphere include ammonification, denitrification, nitrous and nitric oxide production, and products of the anammox reaction. Nitrogen (N_2) fixation by cyanobacteria, on the other hand, is an N gaining process (Robertson and Vitousek 2009; Paerl 2009). The net effect of these gaseous fluxes is usually loss of N to the atmosphere, but the amounts of N gained or lost through atmospheric fluxes are much harder to predict or influence compared to P which has no gaseous atmospheric cycle (Schindler 1977; Robertson and Vitousek 2009). Local measures like crop, fertilizer, or fuel management might be beyond the control of lake managers. In essence, N is hard to manage. Nitrogen management will often be a regional scale matter that is beyond the scope of the lake manager, and costly (Schernewski and Neumann 2002; Neumann and Schernewski 2005).

Nutrient limitation and N:P ratios

Total nitrogen (TN)-to-total phosphorus (TP) ratios in freshwater lakes can range from 1 to 250 by mass (Downing and McCauley 1992), but are typically confined to between 5 and 60 (Downing et al. 2001; Xu et al. 2010; Orihel et al. 2012). Mean summer TN:TP ratios are generally high in oligotrophic lakes and low in hypertrophic lakes, suggesting that the ratio reflects the anthropogenic source of nutrients (Downing and McCauley 1992). During the growth season, the N:P ratio generally decreases due to denitrification losses and sediment phosphorus release (Moss et al. 2013).

But at high, non-limiting nutrient levels, neither nutrient concentrations nor nutrient ratios will have any explanatory meaning (Sas 1989; Moss et al. 2005; Paerl et al. 2011; Paerl and Otten 2013). Other factors besides N or P will limit growth. In those cases, even strong nutrient reduction will have no effect, as shown regarding P reduction by Jeppesen et al. (2002), Søndergaard et al. (2005a), and Moss et al. (2005). This underlines the statement that absolute individual concentrations determine nutrient limitation and thus species composition and abundance (Reynolds 1999; Kolzau et al. 2014). This also implies that statistical scrutiny of ecological effects of N:P ratios with data including high non-limiting nutrient levels will generate unforeseeable results. The suggestions that fluctuations in key variables can be a better predictor of plankton community than averages (Pomati et al. 2012) and that studies should focus on smaller temporal scales (Wagner and Adrian 2011) seem valuable. Nutrient loading and concentrations might vary within days, months, and seasons, influencing the possibilities for macrophyte development and cyanobacterial blooms.

Macrophyte development

In terrestrial and aquatic ecosystems, higher species richness can be related to environments with medium or low food availability. Up to a certain point, biodiversity increases with the number of limiting resources, implying that a higher budget of the potential limiting resources will yield a reduced diversity (Tilman et al. 1982; Tilman 1986; Bobbink et al. 1998; Interlandi and Kilham 2001; Miller et al. 2005; Moss et al. 2013). If N and/or P are available in surplus, the species that are highly efficient in N and/or in P will lose competitive power. Without other limiting nutrient resources, the fastest grower will dominate. Dual limitation increases species richness compared to single limitation.

James et al. (2005) found highest species diversity below a mean winter nitrate threshold of 2 mg N l^{-1}. Related to low macrophyte abundance, several papers draw comparable conclusions: a summer TN threshold of 2 mg N l^{-1} irrespective of TP concentrations (ranging from 0.03 to 0.2 mg P l^{-1}) (González Sagrario et al. 2005), an NO_3 threshold of 2 mg N l^{-1} (Lambert and Davy 2011), and a TN threshold of 1–2 mg N l^{-1} (Kosten et al. 2009). Another conclusion of Kosten et al. (2009) was that macrophytes will remain more abundant (>20 %) if either TP or TN is very low. González Sagrario et al. (2005) drew the same conclusions with respect to low TN. Jeppesen et al. (1990) related maximum colonization depth of macrophytes to summer TP ($r^2 = 0.8$). Jeppesen et al. (2000) related species richness unimodally to TP (in the range 0.02–1.0 mg P l^{-1}). The highest species richness was found below 0.05 mg P l^{-1}, and the maximum colonization depth declined at increasing TP.

There seems little disagreement that low nutrients, especially NO_3 and TN, favor the abundance and species richness of submerged macrophytes. It also seems undisputed that a clear water state with macrophytes can be achieved at low TP concentrations at indiscriminative TN levels. Species richness and acceptable P loading might be increased at lower TN levels. Especially for the management of lakes with high acceptable P loadings, it might be relevant to check on N budgets. Such lakes are typically small and shallow (see "Assessment of P loading" section). If macrophytes colonize larger areas, denitrification generally increases in the low oxygen lake sediments, which will reduce N concentrations as a result of measures initiated to reduce P concentrations (Hilt et al. 2006; Scheffer 2004; Moss et al. 2013). In cases with low critical P loads, the partly correlated TN concentrations might not obstruct macrophyte development if P loading conditions are met. Those lakes are typically larger and somewhat deeper shallow lakesand deep lakes (see "Assessment of P loading" section).

Downstream consequences

Controls on production and nutrient cycling in estuarine and coastal systems are physically and chemically distinct from those in their freshwater counterparts. Nutrient sources in estuaries have lower N:P ratios due to upstream denitrification (Howarth and Marino 2006). Nitrogen fixing and nitrate assimilation are slowed down in marine waters (Howarth and Cole 1985; Howarth et al. 1988a, b). Moreover, growth rates of N_2-fixing cyanobacteria are lower (Marino et al. 2002). As a result, at higher salinities (>8–12) significant N_2 fixing by cyanobacteria has not been observed (Howarth and Marino 2006; Conley et al. 2009). This very limited N_2 fixing in saline waters strengthens the growth-limiting effects of inorganic N depletion and makes N effectively the limiting compound for algae growth in saline waters (Paerl et al. 2004; Howarth and Marino 2006; Lewis and Wurtsbaugh 2008; Smith and Schindler 2009). Reducing algae production by P limitation diminishes the upstream algae N filter (Paerl et al. 2004). This combination of the standing practice of reducing only P loads and anthropogenically induced increased N loading, transported by water or air, have exacerbated the N-limited downstream eutrophication in saline coastal and marine waters (Paerl et al. 2004; Robertson and Vitousek 2009; Paerl 2009; Smith and Schindler 2009; Paerl and Scott 2010). In general, the induced stoichiometric N:P imbalance has major consequences for the marine phytoplankton communities. The dominant P reduction has been shown to suppress nuisance blooms of colonial *Phaeocystis* (Burson et al. 2016).

Synthesis: N or P in LSA and measures?

A single P management approach is in most cases adequate. Next to P management, nitrogen management might be relevant to uplift the critical P load in shallow lakes. Species diversity of macrophytes will be impaired at high N concentrations especially if P is not very low. The downstream consequences toward more saline waters might be taken into account if the upstream lakes (and measures) are not stamp-sized compared to saline estuaries and coastal zones. Large-scale nitrogen emission measures are beyond the scope of local lake management. The LSA will improve if the N limitation status of the lake is included. Calculating the effective N loading is in most cases not necessary and moreover difficult due to the influence of atmospheric fluxes. In the strategy presented here, the quantification and assessment of nutrient loading are focused on P (see "Quantification of P loading" and "Assessment of P loading" sections).

Quantification of P loading

The P loadings, which are important to determine the trophic state, are expressed in mg P m^{-2} day^{-1} or g P m^{-2} $year^{-1}$. There are several methods to calculate these loadings (under nutrients in Fig. 2).

Waterborne P sources

The P loading of a waterborne source on a lake is calculated by the product of an inflow (m^3 day^{-1}) and concentration (g P m^{-3}), divided by lake area (m^2). A water balance model can be used to quantify the majority of waterborne nutrient sources. The water balance can be generated with several tools, ranging from spreadsheet models to numeric hydraulic software. Calibrating the model using a tracer, such as chloride, might improve the water balance substantially. Several sources such as runoff (Sharpley et al. 2013), stormwater inflow (Pitt et al. 2012; Langeveld et al. 2012), and the gross (chemical) groundwater interactions (Griffioen 2006) might have a large influence, but are difficult to calculate or measure in terms of inflow or concentration.

Other P sources

Other nutrient sources as shown in Fig. 2 need a different approach to quantify and express in areal loading: bait by anglers (Niesar et al. 2004; Turner and Ruhl 2007), atmospheric deposition (Peñuelas et al. 2013), tree leaves, macrophytes (Cooke et al. 2005), and waterfowls (Post et al. 1998; Hahn et al. 2007, 2008).

Lake bottom sediment P release (internal loading) should be included in the assessment of nutrient budgets (Burger et al. 2007; Hickey and Gibbs 2009). Lowered external loading will increase the sediment–water gradient of phosphorus, enhancing the legacy P release. Not addressing the internal P loading from sediment in final measures can delay the total P load reduction by several years, or even as much as 20 years (Sas 1989; Søndergaard et al. 2003, 2005a, 2013; Jeppesen et al. 2005; Köhler et al. 2005). The biochemical processes in lake sediments are complex (Mitsch and Gosselink 1993; Smolders et al. 2006), and the estimation of the amount of mobile P proves to be difficult. An important trigger for lake sediment P release is depletion of oxygen and nitrate during the organic matter breakdown process. If these substances deplete, other oxidants like manganese, iron, and sulfates take over, releasing phosphates (bound in iron oxyhydroxides) in the process. The bioavailable sediment fractions transported in the water column should be evaluated. Thus, the long-term contribution of P bound in sediments should be determined on the basis of different P fractions (Zamparas and Zacharias 2014).

For internal loading and subsurface runoff, at least pore concentrations and total content of P, Fe, and S can be used to estimate P release. P release will increase strongly at molar Fe:P ratio <1 in pore water (Schauser et al. 2006; Geurts et al. 2010), but a dedicated research might be needed to quantify internal loading. In addition, budget calculations in epilimnion and hypolimnion can give a quantitative indication of the net internal loading.

In deep lakes, due to thermal stratification, the bottom P release typically remains confined to the hypolimnion during the growth season of phytoplankton and submerged vegetation. At the lake's overturn in winter, high P concentrations may boost spring diatom and chlorophyte blooms. The sedimentation of sinking phytoplankton will again transport nutrients from the epilimnion to deeper layers and the lake bottom sediment. As claimed by Sas (1989), though, it might be that on an annual basis there is no net P release at all, but just an annual cycle. Kleeberg et al. (2012, 2013) confirmed a comparable interaction in which >50 % of the earlier released Fe and P can co-precipitate within a couple of days during the lake's overturn. In stratified lakes, based on P:Fe:S ratios, an intact or disrupted "ferrous wheel" might control the net phosphorus release to the epilimnion.

Assessment of P loading

Reducing the P loading might reduce the overall algae biomass. In addition, the expected relative cyanobacterial abundancy decreases at lower chlorophyll a concentrations (Canfield et al. 1989; Sas 1989; Downing and McCauley 1992; Duarte et al. 1992; Downing et al. 2001; Watson et al. 1997). Downing et al. (2001) found that in 99 north temperate lakes, chlorophyll a concentrations of ca. 10 μg l^{-1} might result in an expected 30 % cyanobacterial fraction, increasing to 80 % at chlorophyll a concentrations of ca. 100 μg l^{-1}. Below 0.03 mg l^{-1} TP, the risk of cyanobacteria dominance is 10 %, increasing to 40 % at 0.07 mg l^{-1} TP. Jeppesen et al. (2005) found low cyanobacterial abundances in lake thresholds of <0.050 mg P l^{-1} in shallow lakes and <0.010–0.015 mg P l^{-1} in deep lakes. A chlorophyll a maximum of 10 μg l^{-1} is defined as acceptable for recreational waters as regards toxin content (Chorus 2012).

The calculated lake loadings (see "Quantification of P loading" section) can be assessed by comparing them with the threshold loadings for clear water without cyanobacterial blooms. Shallow (polymictic) and deep (monomictic or dimictic) lakes behave different with decreasing P loading ("Shallow versus deep lakes" section), so does the method to define the acceptable threshold loadings. Also, the conventional terminology for these thresholds differs for shallow and deep lakes. Here the term "acceptable loading" is used if the threshold is applicable to both shallow and deep lakes.

The assessments are based on lakes that are dominated by ecological processes instead of transport. A transport-dominated lake does not generate a eutrophic expression under high nutrient loading but

Table 1 Indication of modeled effects of lake characteristics on critical P loads in shallow lakes

critical P load default lake 0.9–3.0 mg P m^{-2} day^{-1}		Scenarios			Effect on critical P load		
		min	Default	max	min	max	Remark
Wind fetch	m	100	1000	3000	+++	–	
Mean water depth	m	1	2	4	++++	–	Model overrates effect of shallowness ≤1 m
Sediment type	–	Peat	Clay	Sand	–	+	Largest effect on clear to turbid threshold
Marsh zone	%		0.1	100		+	
Hydraulic load	mm day^{-1}	10	20	80	–	++	Largest effect on clear to turbid threshold
Fishery rate	day^{-1}	0	0.00137	0.01	○	○	
N/P load ratio	–	3	10				Switch from N limitation to P limitation

Each scenario has seven lake parameters with respect to the "default lake." The last column gives the effect on the critical P loading. A "min" or "max" shows the effect of a deviation of the default lake on critical P loading, in which "+" indicates an increase, "–" a decrease, and "○" a marginal effect based on Janse et al. (2008)

low (<0.02 mg P l^{-1}) inflow concentrations due to high flushing rates (Cameron Lake paradox: Dillon 1975; Brett and Benjamin 2007).

In shallow lakes, the concept of at least two alternate stable states implies two "critical" P loading switch points (see "Shallow versus deep lakes" section). These loadings to achieve or maintain a clear state are affected by lake size (wind fetch), bottom sediment composition, depth, retention time, and climate (Janse 2005; Scheffer and van Nes 2007). The critical phosphorus loadings can be calculated with the PCLake model (Janse and van Liere 1995; Janse 1997; van Puijenbroek et al. 2004; Janse 2005), which takes into account interactions between macrophyte recovery and decay, fish, zooplankton, and phytoplankton in shallow lakes (Janse 2005; Nielsen et al. 2014). The model has been calibrated on 43 and validated on 9 lakes (Janse 2005; Janse et al. 2008) and has been subjected to a sensitivity analysis (Janse et al. 2010) and used and validated in several studies (Trolle et al. 2014). An online meta-model (in Dutch) for P-limited lakes is available (URL: PCLakeMeta) for quick analyses (Janse 2005; Janse et al. 2008; Mooij et al. 2010).

For the most common Dutch lakes, which Janse (2005) used for calibration, the critical P loading for turbidification is calculated as 2–5 mg P m^{-2} day^{-1} (0.05–0.1 mg P l^{-1}) and the value for clarification as 0.6–1.0 mg P m^{-2} day^{-1} (0.03–0.05 mg P l^{-1}). In terms of phosphorus concentrations, this fits well with critical concentrations as reviewed by Beklioglu et al. (2011) for north temperate shallow lakes (mainly North America and Europe): 0.025 (clarification) and 0.15 (turbidification) mg TP l^{-1}, of which the upper limit can be much higher if N is a limiting nutrient.

For a "default" shallow lake, critical P loadings were calculated to be 0.9 (turbid to clear) and 3.0 (clear to turbid) mg P m^{-2} day^{-1} (Janse 2005). An exploration of the sensitivity of critical P loadings on several system characteristics, based on modeled imaginary shallow lakes, is summarized in Table 1. The results show that the reduction in water depth (2–1 m) or fetch (1000–100 m) increases greatly the critical P loading of the default shallow lake. The opposite, an increase in water depth (2–4 m) or wind fetch (1000–3000 m), decreases the critical P loading to a much smaller extent. This is in accordance with a survey of 800 Danish lakes, in which the smaller lakes (<1 ha) were found to be less susceptible to nutrient loading than the larger lakes (Søndergaard et al. 2005b). The relative coverage of macrophytes was highest in the lakes smaller than 10 ha. The results presented in Table 1 are also in line with a survey of 215 lakes in the Lower Rhine floodplain, in which the smaller lakes (<1 ha) had 50 % coverage of macrophytes ($n = 124$) and the 1–4 ha lakes had 20 % coverage ($n = 52$). Macrophyte coverage was 50–60 % at a lake mean depth of 0–1 m ($n = 109$), and coverage of 20 % was found at mean lake depth of 1–2 m ($n = 89$). At higher lake depths, coverage was well below 10 % ($n = 18$). TP ranged from 0.02 to 0.40 mg P l^{-1} in the highly covered lakes (van Geest et al. 2003). The results fit rather well with the empirically observed critical P loadings of 0.8 and 1.5 mg P m^{-2} day^{-1} (with summer TP of 0.018 and

Table 2 Calculated acceptable P loads of deep lakes

lake	Zmax	Zmean	residence time	acceptable P-load		remark	tool	reference
	m	m	y	$mg/m^2/d$	$g/m^2/y$			
virtual	11		0.25-50	0.05	0.02	without macrophytes	PCLake, SALMO-1D, GOTM	(Sachse et al. 2014 Fig. 8b)
	30			0.5	0.2			
	100			0.5	0.2			
	11			2.2	0.8	with macrophytes		
	30			1.2	0.5			
	100			0.8	0.3			
Lake Ravn	33	15	1.75	1.4	0.5	50% reduction -> Good Ecological Status (WFD), 6.5 µg chlorophyll a	DYRESM–CAEDYM	(Trolle et al. 2008a; Trolle et al. 2008b)
				0.3	0.1	90% reduction -> significant increase depth limit submerged macrophytes		
14 US lakes		10.6 - 313	0.2 - 700	0.30	0.11	permissible level, $q_s = 1$	modified Vollenweider relationship	(Rast and Lee 1978)
				0.59	0.22	excessive level, $q_s = 1$		
				0.55	0.20	permissible level, $q_s = 10$		
				1.1	0.40	excessive level, $q_s = 10$		

The virtual lakes with a z_{max} of 11 m are added as comparison; they represent a lake of moderate depth. The q_s is the hydraulic load (see text)

0.040 mg P l^{-1}) in Lake Botshol. This lake has a depth of 1.5–3 m, an area of 2.87 km^2, a hydraulic load of 45 mm day^{-1}, and an approximate effective fetch of 700 m (Rip 2007; Rip et al. 2005).

For deep lakes, the acceptable phosphorus loadings are expressed as a “permissible” and an “excessive” loading. The approach to define these differs from shallow lakes (see Table 2). Based on inorganic

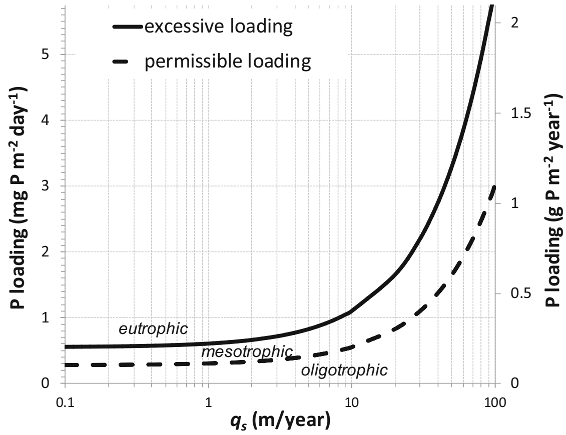

Fig. 3 Permissible and the excessive P loading from the modified Vollenweider relationship versus q_s (hydraulic load; mean depth-to-retention time ratio). From Rast and Lee (1978, formulae 11 and 12)

phosphorus concentrations of 0.01 mg P l^{-1} at the spring overturns (Sawyer 1947, 1952; Benndorf 1979), a sedimentation coefficient depending on mean lake depth, and a steady state reactor model, Vollenweider (1975) derived a permissible TP load. An excessive P load was defined as twice the permissible P load. Rast and Lee (1978) and Rast and Thornton (2004) called this the modified Vollenweider relationship (Fig. 3): $L_c\,(P) = (100 + 10 \times q_s)/365$, in which $q_s = z/\tau$. $L_c(P)$ is the permissible TP load (mg P m^{-2} day^{-1}), q_s is the hydraulic load (m/y), z is the mean depth (m), and τ is the hydraulic retention time (yr).

Internal P loading is not explicitly captured in this model, and the model does not take into account time-dependent variation in P loads. The modified Vollenweider relationship has been validated on 14 US lakes >10 m mean depth (Rast and Lee 1978). Twelve of these lakes have q_s of between 0.8 and 10 m yr^{-1}. This implies that for these lakes, the theoretical P load boundaries will be between 0.30 and 1.1 mg P m^{-2} day^{-1}. This expression is used to assess four Dutch sandpits 10–13 m mean depth and 0.6–0.9 km^2 surface area, with q_s of 1–3 m yr^{-1} (reports Waternet 2010–2014). The yearly excessive and summer permissible P load thresholds seem to be good limits to prevent cyanobacterial blooms in these lakes. The P loads of these lakes are dominated by surface and subsurface runoff, which is relatively low during the growth season. Especially for smaller lakes, both yearly loading and growth season loading should be assessed, since cyanobacterial blooms might be an instantaneous response to loading.

The acceptable P loadings for deep lakes are generally lower than for shallow lakes (Sas 1989; Jeppesen et al. 2003). Sachse et al. (2014) modified the PCLake model for application in a deep lake. The results indicate the stabilizing influence of macrophytes at moderately deep lakes, increasing the acceptable P loadings (Table 2). Furthermore, larger and deeper lakes can contain a larger absolute amount of biomass, yielding to higher scum forming potential. Cyanobacterial concentrations of 10 µg l^{-1} in the center of a lake can impede bathing water functions at the shorelines.

Measures

This section provides an overview of measures, differentiated in preventive and control measures, to be used in phase 3 ("Selecting measures" section).

Preventive measures

Preventive measures can be used to ensure that a lake switches (shallow lake) or gradually transforms (deep lake) to a clear state. To achieve a P loading below the acceptable P loading, measures might include:

- Nutrient measures, here that is decreasing the P loading (see "Phosphorus versus nitrogen" section).
- Hydromorphological adjustments, by either indirectly decreasing the P loading or increasing the acceptable P loadings.
- Direct food web management, to induce a switch to a clear state if the P loading is between the critical thresholds.

The nutrient measures and hydromorphological adjustments generate the preconditions, and direct food web management might help to tip the balance. Hydrological measures are intertwined with the sets of nutrient and hydromorphology measures. Food web measures and hydromorphological adjustments will mainly be successful in shallow lakes; they are less relevant in deep lakes, since the food web influences clarity and cyanobacterial blooms to a far smaller degree (Jeppesen et al. 1997; Mehner et al. 2002, 2008). In Table 3, the most relevant measures

Table 3 Summary of preventive measures, with indication of applicability

#	Cat	Primary target	Measure	Applicability											
				Depth		Size		Cost		Speed		Risk		Support	
				Sh	D	Sm	L	Lw	H	Sl	F	Lw	H	Ng	P
1	FW	Fish	Harvesting zooplanktivores	xx	x	xxx		x	x		xx			x	
2	FW	Fish	Stocking piscivores	xx	x	xxx	x	xx		xx				x	
3	FW	Fish	Enhancing migration	x	x	x	x		x	x		x	x		
4	FW	Mussel	Stocking	xx	xx	xxx	xx	x	x		x	xx	x	x	
5	FW	Macrophyte	Protecting	xx	x	xx			x			x	xx		
6	FW	Macrophyte	Seeding	xx		xx						x			
7	FW	Macrophyte	Stocking	xx		xx			x		x	x			
9	HM	Fish, macrophyte: light	Compartmentalizing	xx		xx	x	x	xx		xx	xx		x	
9	HM	Macrophyte: light	Fetch reduction	xx		xxx	x	x	x	x	x	x	x		
10	HM	Mussel	Substrate enhancement	xxx	xxx	xxx	xx	xxx		x		xx	x		
11	HM	Mussel	Excavation	xx		xx			xx	x			xx		
12	HM	Macrophyte: light	Drawdown permanent	xx		xx	x	xx	xx	xx		x	x	x	
13	HM	Macrophyte: light	Drawdown temporary	xx		xx	x	xx	xx	xx		x	x		
14	HM	Macrophyte: substrate	Capping	xx		xx	x		xx	x	x	x	x		
15	HM	Macrophyte: substrate	Dredging	xx		xx			xxx	x	x	x	x		
16	Nu	Anglers	Feeding discouragement			xx		xxx		xx			xx	x	
17	Nu	Bottom sediment	Periodic desiccation	xx		x	x	x							
18	Nu	Bottom sediment	Active capping	x	x	xx	x		xx		x	x	x	x	
19	Nu	Bottom sediment	Passive capping	x	x	xx	x		xx		x	x	x		
20	Nu	Bottom sediment	Removal	xx		xx			xxx	x	x		x		
21	Nu	Bottom sediment/groundwater	Hypolimnetic withdrawal		xx	x	xx	x	x		xx	xx			
22	Nu	Drainage	Manure management						x	xxx		x	xx	x	
23	Nu	Drainage	P removal					x	xx		xx	x			
24	Nu	Drainage	Adjustable drainage					x	xx		xx		x		
25	Nu	Drainage	Diversion					x	xxx		xx	x			
26	Nu	Drainage	Recirculation					x	xx		xx	x			
27	Nu	Groundwater	Active capping	x	x	xx	x		xxx		x	x	xx	x	
28	Nu	Groundwater	Groundwater controlled surface water level management	x	xx	xx	x	x	xx	x		x	xx	x	
29	Nu	Groundwater	Groundwater management	x	xx			xx	x	x	x	x	x	x	

Table 3 continued

#	Cat	Primary target	Measure	Applicability											
				Depth		Size		Cost		Speed		Risk		Support	
				Sh	D	Sm	L	Lw	H	Sl	F	Lw	H	Ng	P
30	Nu	Groundwater	Permeability reduction (seepage)	x		x	x	x	xxx	x	x	x	xx		
31	Nu	Inflow water	P removal						xx		xx	xxx			
32	Nu	Inflow water	Dilution (of lake)	xx		xx		xx	x		xx	x	x		
33	Nu	Inflow water	Disconnecting/diversion			xx		x	x		xx	x			
34	Nu	Inflow water	Permeability reduction (infiltration)	x		x	x	x	xx	x	x	x	x		
35	Nu	Inflow water	Water level management	xx	x	xx	x	xx	x	xx		x	x	x	
36	Nu	Macrophytes	Maintenance	xx		xx	x	x	x	x		x			
37	Nu	Sewer (overflow)	Diversion					x	xxx		xx	xx			
38	Nu	Sewer (overflow)	Increasing capacity						xx		xx	xx			
39	Nu	Sewer (WWTP)	Diversion					x	xxx		xx	xx			
40	Nu	Sewer (WWTP)	Upgrading efficiency						xxx		xx	xx			
41	Nu	Stormwater	P removal					x	xx		xx	xx			
42	Nu	Stormwater	Correcting faulty connections					x	xx	x	xx	xx	x		
43	Nu	Stormwater	Sediment removal					x			xx	x	x		
44	Nu	Stormwater	Compartmentalizing/diversion					xx	xxx		xx	xx			
45	Nu	Stormwater	Increasing capacity						xx		xx	xx			
46	Nu	Stormwater	Prioritizing discharge locations					x	x		xx	xx			
47	Nu	Waterfowls	Discourage settlement			xx		xx		x	x		x	x	

Category: *FW* food web, *HM* hydromorphology/hydrology, *Nu* nutrients. Applicability: *Sh* shallow, *D* deep, *Sm* small, *L* large, *Lw* low, *H* high, *Sl* slow, *F* fast, *LR* low risk, *H* high risk, *Ng* negative, *P* positive. An "x" represents "true," with a maximum of three. If a category scores "x" on both sides, it indicates a range. The scores postulate an adequate LSA. "Costs" are the estimated relative costs per square meter. Under "Speed," is indicated how fast an implemented measure will have the desired effect. Under "Risk," the inherent risks of failure, unwanted side effects, and irreversibility are combined. Under "Support," stakeholders' support or lack thereof is indicated are presented with relative and partly subjective scores on several items that are relevant to lake management.

Nutrients

A set of measures (see supplementary Figure) can be selected on the basis of the calculated P budgets (see "Quantification of P loading" section). The total number of possible nutrient measures is far too extensive to describe, because almost any nutrient source has multiple optional measures with a wide range of varieties (see Fig. 4).

Some generic measures are applicable at several sources of different origin; for example, diversion can be used for drainage, sewer, stormwater runoff, and inflow water. The measure can have decisive effects, since it can divert 100 % of the sources and also reduces nitrogen loading. In urban areas, however, diversion might not be feasible since it is very expensive to reconstruct infrastructure. A complicating factor is that this method may impair other waterbodies. Moreover, introducing compensating water (thus nutrient) sources may be necessary.

Phosphate removal techniques for inflowing waters are common practice and can be used for (agricultural) drainage, wastewater treatment plant (WWTP) effluent, stormwater, and inflow water. Lamella settlers are typically deployed for stormwater, with a P removal of 0.3 (Langeveld et al. 2012). A majority of chemical P removal is based on communal wastewater treatment plant techniques with metal salts (mainly iron, alum). Removal efficiencies can be as high as >0.9 (Klapper 2003). In this process, chemical sludge is produced with low P recycling options. More high-end options are continuous backwash filters. Surface flow systems like sewage farms, mash zones, wetlands, vegetated buffer strips, and sedimentation ponds are easy to maintain and have little chance of hydraulic failure. They might have some P removal properties, but might in worst cases also act as P sources (Sheppard et al. 2006; Ballantine and Tanner 2010). Vymazal (2011) reviewed various approaches of constructed wetlands and concluded that wetlands are usually not designed for P removal. Langeveld et al. (2012) reviewed subsurface flow systems like sand and soil filters and found removal efficiency of 0.4–0.8. Ballantine and Tanner (2010) reviewed the upgrading of substrate (by amendments) and filter materials for wetlands. This can drastically increase P removal and is an important improvement in lake measures. Grain size distribution, pH, specific surface area, and the availability of Al, Fe, and Ca ions are important aspects (Ballantine and Tanner 2010; Buda et al. 2012; Stoner et al. 2012). These soil passage techniques with P sorbing qualities are not only for treatment of inflow water and stormwater through constructed wetlands: Basically, the same materials and techniques can be used for runoff and drainage water in permeable buffer strips or enveloped drain structures (Buda et al. 2012). Enriched and suitable substrates, large contact areas, and ample contact time further increase P removal, but hydraulic resistance and filter clogging might endanger throughput. This makes size, design, and management crucial for P removal efficiency. Such efficiencies might vary from 0.3 up to >0.9 (Sheppard et al. 2006; Ballantine and Tanner 2010; Groenenberg et al. 2013; Penn et al. 2014). Provided there is ample hydraulic head, especially for subsurface flow or drains, the sorbing beds can be implemented below the surface in wetlands and drains without additional hardware, fully integrating into the natural surroundings. If designed well, the filter contents might have to be replaced only once every decade (Groenenberg et al. 2013). Depending on the design and infiltrating water quality, a pump and a backwash system might be essential.

From a sustainability perspective, natural local minerals as well as residual products (iron sludge or iron coated sand) from drinking water treatment plants (Chardon et al. 2012; Groenenberg et al. 2013), or by-products of power or steel production (steel slag) (Stoner et al. 2012; Penn et al. 2014), can be used without sacrificing removal efficiency. Developments in the use of by-products are reviewed by Buda et al. (2012) and Stoner et al. (2012). But it is difficult to redeem the removed phosphorus to complete the circle. This might need other biological removal techniques to be explored for surface water management.

In phosphate removal techniques for restraining the internal loading, three leading principles can be distinguished: removal, passive capping, and active capping. Such activities have in common that they might impose a decisive impact on the benthos and biochemistry of the sediment.

In small- or moderate-sized shallow lakes, the removal (dredging) of enriched lake sediment is feasible, albeit costly (Xu et al. 2012; Bormans et al.

2015). Dredging will remove the benthic biota including (propagules of) macrophytes (Cooke et al. 2005). It might expose viable seed banks (de Winton et al. 2000; Hilt et al. 2006), expose toxic layers (Cooke et al. 2005), or change the active chemical composition (Fe, P, S) (Geurts et al. 2008). Thus, dredging might reduce or increase, for example, the population and diversity of macrophytes, denitrification, and sediment nutrient release. Bormans et al. (2015) reviewed the applicability of dredging.

Passive capping confines the reactive bottom sediment by means of a physical boundary. This measure will only have lasting effects if external nutrient loading toward the lake is low (see "Other P sources" section). In the case of soft sediments, the capping material may sink into the enriched bottom material, frustrating the goal of decreasing the P release from the sediment. Passive capping can be done by, for example, spraying soil-like sand, gravel, or clay (Xu et al. 2012).

Active capping by the chemical fixation of phosphates is a common approach. It can be combined with passive capping to create a chemical boundary for inflowing seepage water or to ensure the confinement of the capping agent. Active capping can create an active phosphorus sink, removing some external loading as well. Most P adsorbents with substantial uptake capacity are generally enriched in calcium, iron, alum, and/or lanthanum (Douglas et al. 2016). In shallow lakes with an aerobic bottom sediment, iron is commonly used as a P capping agent due to its low cost. In deep lakes, iron is less favorable due to release mechanisms at low oxygen concentrations (see "Quantification of P loading" section). If ample iron is already available, artificial oxygenation can be a way to increase the redox potential and thus the binding capacity of iron (Conley 2012; Stigebrandt and Kalén 2013; Stigebrandt et al. 2014; Gerling et al. 2014). Bormans et al. (2015) reviewed the applicability of hypolimnetic oxygenation. Alum (salts, zeolites), calcite (+nitrate), Phoslock^{TM} (lanthanum-enriched bentonite clay), and modified zeolite are most commonly used (Spears et al. 2013; Yang et al. 2014; Zamparas and Zacharias 2014). Important issues are the choice of appropriate P adsorptive, dose, and the longevity of an application (Zamparas and Zacharias 2014). Longevity might be impaired by low doses (Cooke et al. 2005; Jensen et al. 2015), chemical aging (Søndergaard et al. 2007; de Vicente et al. 2008), or physical redistribution in lake sediment. A single big dose might be less effective than smaller doses due to aging of the active compound (de Vicente et al. 2008; Jensen et al. 2015). Douglas et al. (2016) reviewed solid-phase phosphorus adsorbents, categorized in: naturally occurring minerals, soils, suspended particles or earth materials; natural or synthetically produced materials; modified clay mineral or soils; and mining, mineral processing, and industrial by-products. Spears et al. (2013) used six industrial waste products and concluded that some of them have P sorption properties similar to those of products that are commercially available. Xu et al. (2012) used local soil, being rich in iron and alum, to create a 5–10 cm capping of lake sediment. Sediment P release was halved after 18 months.

An indirect measure to mitigate internal loading or groundwater inflow is hypolimnetic withdrawal of anoxic water from lakes and reservoirs. This is a proven method to reduce P, metals that are undesirable in drinking water supplies (e.g., manganese), ammonia, and hydrogen sulfide (Klapper 2003; Cooke et al. 2005). A suction pipe along the lake bottom transports the water elsewhere, preferably to a waterbody with a lower water level so it can be used as a siphon. An important requirement is that the amount of the discharge is limited, to avoid disturbing the stratification and transporting nutrient rich water to the epilimnion, which might promote cyanobacterial blooms (Cooke et al. 2005; Çahşkan and Elçi 2009; Dunalska et al. 2014). Bormans et al. (2015) reviewed the applicability of hypolimnetic withdrawal.

In sand and gravel pits directly connected to an unconfined aquifer, water level management in symbiosis with natural groundwater level fluctuations will reduce groundwater–surface water exchanges. In artificially managed lakes, adaptive water level management based on hydrological forecasts can minimize the need for inflow water (see also "Hydromorphology" section).

Increasing storage is an option that reduces the inflow of water and nutrient sources. Measures focused on detaining and smoothing out the typical peak flows of, for example, stormwater decreases lake nutrient loadings. They also reduce the spilling of a valuable fresh water flow, which might have to be replenished in periods with less water surplus. Management can be focused on a multiple use of space to contain these stormwaters. Examples are adjustable drains, surface or subsurface stormwater basins, and green infrastructures like permeable pavements, green roofs, and rainwater gardens. In some options, also P concentrations are reduced. The smoothening of high flow rates might add to the P removal efficiencies. Although some of these green infrastructural developments are well on their way, there certainly seems to be room to improve innovation.

Hydromorphology

Hydromorphological adjustments (see Fig. 4) create the preconditions required to increase the acceptable nutrient loading thresholds (see "Assessment of P loading" section). Smaller shallow lakes generally withstand a higher nutrient loading and are clearer than larger lakes (see "Assessment of P loading" section) due to relatively low fish abundances and reduced wind fetch (van Geest et al. 2003; Søndergaard et al. 2005b; Scheffer and van Nes 2007). Optimal ecological resiliency seems to occur in smaller lakes with varying depth profiles. Furthermore, as mentioned in "Assessment of P loading" section larger and deeper lakes have a higher scum accumulating potential.

The reduction in lake size by compartmentalizing increases the acceptable nutrient loading. An additional benefit can be that dominant nutrient sources might be diverted. The reduction in wind fetch will reduce resuspension, which increases light availability at the lake sediment, favoring macrophytes. Compared to compartmentalizing, wind fetch reduction will be preferable for anglers and recreational use. Measures related to size and fetch will have relevant effects at the lower range of compartment sizes ("Assessment of P loading" section, Table 1). They can be constructed by means of dams (fetch reduction and small compartments) or only, for example, rows of poles (only fetch reduction). Promising results are possible by creating dams or even islands with geotextile constructions to store and reuse lake sediment (Lauwerijssen et al. 2015). The dam or island itself will be the substrate for vegetation. Optimal design and an ample settling period (aided by coagulation of the substrate) might allow for the removal of the construction plastics from the lake in due time.

Substrate enhancement relates to the root anchorage of macrophytes and grazing enhancement by freshwater mussels. Soft sediments (as a relay from eutrophic conditions) make macrophytes more susceptible to drag forces created by currents and waves (Schutten et al. 2005). To prevent their easy dislodgment, soft sediments can be dredged or capped with, for example, sand. Dredging can be a promising approach, if accompanied with ample upfront research (see "Nutrients" section).

Dreissenid mussels like the zebra mussel (*Dreissena polymorpha*) and the quagga mussel (*Dreissena rostriformis bugensis*) filter seston (mainly phytoplankton), which may drastically improve light conditions for macrophytes. In the dispersion of dreissenids, a suitable substrate can be a decisive

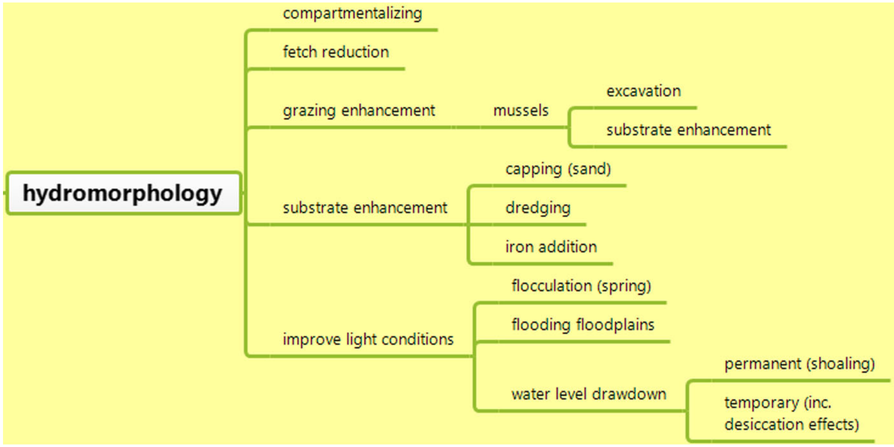

Fig. 4 Overview of hydromorphological measures

factor (Burlakova et al. 2006). For instance, sand, rock, clay, wood, macrophytes, and the shells of other bivalves are suitable substrates (Garton et al. 2013). Karatayev et al. (2014) indicated for quagga mussels in North America, typical lake bottom coverages of 1000–10,000 m^{-2} (2014). Clearance rates can range 50–250 ml h^{-1} per individual (Horgan and Mills 1997; Ackerman 1999; Diggins 2001). This yields a potential filtration capacity of a 1–60 m day^{-1} water column per square meter covered with dreissenids. The mussels enhance the benthic–pelagic coupling by providing shelter and substrate for the benthic community (Vanderploeg et al. 2002). In a survey of >65 lakes, the impact of dreissenid mussels was quantified: for example, turbidity (−41 %), chlorophyll a (−47 %), phytoplankton (−58 %), and zooplankton (−51 %), and in the littoral periphyton (171 %), macrophytes (182 %), and zoobenthos (212 %, excluding dreissenids). Mussels also seem to decrease the nutrient concentrations (Higgins and Vander Zanden 2010; Higgins et al. 2011; Johengen et al. 2013), which might be temporary if biomass reaches a steady population (Bootsma and Liao 2013). Zebra mussels excrete relatively more P than N, decreasing the N:P ratio (Arnott and Vanni 1996; Conroy et al. 2005; Wojtal-Frankiewicz and Frankiewicz 2011). The overall mechanism of dreissenid abundance shows that a dense population is able to significantly increase the acceptable loadings of lakes, inducing clear water states (Mayer et al. 2013; Karatayev et al. 2014).

Several mechanisms distinguish cyanobacteria from other seston related to mussel grazing:

- The overwintering strategy of cyanobacteria in lake sediments (see "Shallow versus deep lakes" section), if disrupted by grazing, might result in hampered recolonization from lake bottom sediments.
- The ability of cyanobacterial buoyancy regulation and its diel vertical migration cycle influences the time spent near mussel populations compared to sinking algae species. From model studies based on *Microcystis*, a vertical migration of 5–8 m seems feasible depending on, for example, colony size (Visser et al. 1997; Wallace and Hamilton 1999). Field data show that *Microcystis* and *Anabaena* have not been found at sufficient depth to reach the (nutrient richer) metalimnion depth (Bormans et al. 1999). It seems likely that in both shallow lakes and the littoral zones of deep lakes cyanobacteria might have an advantage (staying higher in the water column compared to sinking algae) or a disadvantage (actively migrate themselves toward their predators), depending on factors influencing their buoyancy cycle.
- Reduced grazing pressure on *Microcystis* by selective rejection or excretion of viable cyanobacterial cells (Vanderploeg et al. 2001; Bierman Jr. et al. 2005; Knoll et al. 2008; Fishman et al. 2009; Vanderploeg et al. 2013; Tang et al. 2014) or reduced filtration time induced by toxicity (Vanderploeg et al. 2001, 2009) might favor *Microcystis* dominance (Vanderploeg et al. 2002; Knoll et al. 2008). However, the digestion of colonies of *Microcystis,* preferent digestion of single-celled *Microcystis* (Dionisio Pires et al. 2004, 2005), or mixed results (Vanderploeg et al. 2013; White et al. 2011; Raikow et al. 2004) have also been found. The filamentous genera *Aphanizomenon* (Horgan and Mills 1997), *Anabaena* (Horgan and Mills 1997; Knoll et al. 2008; Tang et al. 2014), and *Planktothrix* (Dionisio Pires et al. 2005) can be digested and assimilated.

The impact of mussel grazing on different cyanobacteria is not fully understood. It seems fair to assume that most cyanobacterial genera are effectively assimilated. But especially large-sized *Microcystis* colonies seem to be avoided. If that is the case, this implies that mussel grazing is more effective at preventing cyanobacterial blooms under nitrogen-limiting circumstances. N-limiting conditions will prevent non-N-fixing genera like *Microcystis* from blooming, and the N-fixing genera will be assimilated by the mussel population.

To expand an existing dreissenid population, the most effective depths based on the migration of cyanobacteria are from 8 m up. But locations shallower than 2.5–3.5 m depth largely increase possible predation by waterfowl (van Nes et al. 2008). If the substrate is removable, a periodic harvesting of mussels might compensate for nutrient loading.

To improve light conditions, the flooding of floodplains might add large shallow areas, effectively increasing lake size and sharply decreasing the average depth. It has been shown in Lake Sakadaš that even with elevated organic matter and nutrient

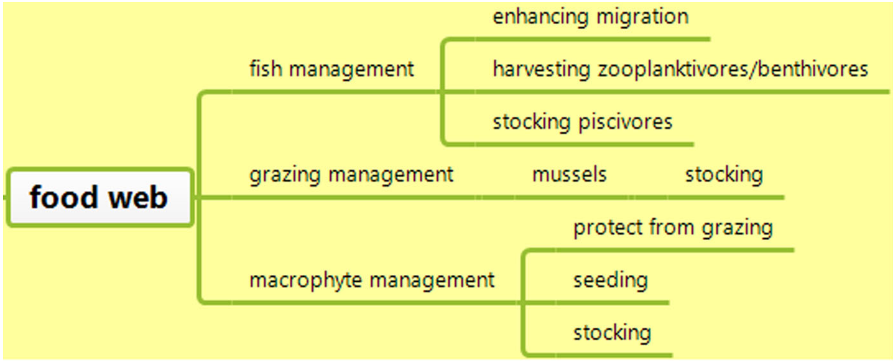

Fig. 5 Overview of food web measures

concentrations that accompany the high floodplain levels, macrophytes rapidly became dominant and phytoplankton biomass remained low (Mihaljević et al. 2010). Although in this case a Secchi depth did not change compared to regular years, it did show a vast and fast influence of water level management on the ecological state of a lake.

Shoaling or lake level drawdown to increase light at the lake bottom is expected to have the highest efficacy in the 1–2 m range (Janse et al. 2008). A twofold drawdown of up to 1.5 m in Lake Tämnaren had a tremendous effect on clarity and macrophytes. After a level increase in 0.3 m, the lake became turbid again with a macrophyte coverage collapsing from 80 to 14 % (Scheffer 2004). A variation on this leading principle is a temporary water level drawdown in spring to enhance light availability (Coops and Hosper 2002; van Geest et al. 2005). A drawdown of 0.3–0.6 m might suffice to foster results (Hilt et al. 2006). Because of different tolerances of temporary desiccation, the temporary drawdowns will also have effects on species composition (van Geest et al. 2005).

Food web

Direct intervention in the food web (see Fig. 5) can have a direct and decisive impact on aquatic biota. Such measures should be able to swing a lake ecosystem from an instable to a stable clear state, provided the nutrient status allows this shift. If successful, food web measures speed up the transition to a clear state and can also be of high efficiency. If only temporarily successful, food web intervention can be considered as a control measure, rather than a preventive measure. Measures consist here of direct intervention on fish, macrophytes, or freshwater mussels, by means of stocking, seeding, or removing species.

If the aquatic ecosystem (especially nutrient loading) is not ready to swing to a clear state, fish management (commonly known as biomanipulation) has proved to be only a temporary measure (Søndergaard et al. 2008; Vašek et al. 2013). After nutrient reduction, Jeppesen et al. (2005) found a natural response of fish community structure and biomass within 10 years. The stocking of piscivorous fish or removal of zooplanktivores and benthivores can greatly reduce turbidity and the destruction of submerged macrophytes, generating immediate results (Angeler et al. 2003; Hilt et al. 2006; Genkai-Kato et al. 2012; Bakker et al. 2013; Pot and ter Heerdt 2014). At least 75 % of the fish remnants of the turbid state should be reduced to obtain clear waters (Meijer 2000). Fish stock reduction to, and maintenance at, <50 kg ha^{-1} should generate a successful result (Gulati and van Donk 2002). Reducing solely the carp population from 300 to 40 kg ha^{-1} has been found to have a marginal effect (Bajer and Sorensen 2015). The stocking of piscivores can be done at any lake size at a relatively low cost, but the removal of fish leads to faster results, and although it is more costly, it has a high efficacy (Meijer 2000). Effects were significant over 6–10 years in 36 Danish lakes (Søndergaard et al. 2008) and >8 years in Dutch eutrophic lakes (Meijer 2000). Data from over 70 biomanipulated lakes indicate a return to a turbid state within 10 years in most cases (Søndergaard et al. 2007), emphasizing that an adequate nutrient status is a precondition to achieve a clear lake.

Macrophytes might respond within a few years after restoring the fish community structure (Pot and ter Heerdt 2014). But according to Bakker et al.

(2013), macrophyte recolonization without food web management might take 20–40 years. The recolonization rate will depend on, for example, turbidity, grazing, fish, and viable propagules. Propagules can survive for 20 years or longer (de Winton et al. 2000; Hilt et al. 2006). Active macrophyte cultivation by stocking or seeding can give instant results in lake restoration, mainly by reducing sediment phosphorus release (Søndergaard et al. 2001; Hilt et al. 2006).

Introducing invasive allochthonous freshwater dreissenid mussels from mussel farms might lead to a switch to a clear state (see "Hydromorphology" section). One of the adverse side effects might be that the switch enhances exploding macrophyte abundances with little species diversity, due to the unaltered nutrient loading (see "Macrophyte development" section). A large mussel population will adapt a fish population structure and might attract diving ducks, and there will be numerous other effects on autochthonous species. In larger lakes, dreissenids can increase the eutrophication in the nearshore area (nearshore shunt hypothesis) (Hecky et al. 2004; Roy et al. 2010; Bocaniov et al. 2014). Especially, quagga mussels are infamous for their 3D expansion, enabling them to clog water transportation pipelines (Chakraborti et al. 2013). Once introduced, it is impossible to eradicate the mussels from a lake ecosystem without drastic (chemical) intervention (Sousa et al. 2014), which makes introducing dreissenids an irreversible measure.

Control measures

Whenever water managers face problems with scums at undesirable locations (e.g., bathing water sites or drinking water intake points), measures leading to fast positive results are desired. The number of measures with a significant negative effect on the growth or accumulation of cyanobacterial biomass can be reduced to a limited set of leading principles: biomass removal, flushing, and mixing (see Fig. 6). Which measure is most suitable depends on the targets that need to be met (see "Ecological targets" section), the problem experienced (toxicity, scum, odor), the genus of cyanobacteria, and the lake characteristics (size, depth, isolated vs. flowing waters, position in relation to prevailing wind direction, etc.). The most relevant control measures and their relative scores are presented in Table 4. Several measures with doubtful effectiveness are reviewed by Lürling et al. (2015).

Compartmentalization

A smaller lake size increases the success rate of applied actions, simply because in smaller compartments control measures are easier to manage. Compartmentalizing within larger waterbodies can be realized with dams (see "Preventive measures" section) or, when rapid or temporary action is required, with oil slick booms with chain-weighted sleeves down to the lake bottom. Combining a controlling measure within a compartment can prove to be optimal especially for bathing or recreational waters.

Biomass removal

Chemical removal Water managers and stakeholders are in general reluctant to use chemical control, because of the negative effects such substances may have on the overall aquatic ecology. In general, algaecides should only be used when cell numbers are low, to avoid excessive release of cyanotoxins or taints following the rupture of the cyanobacterial cells (Jones and Orr 1994; Chorus and Bartram 1999). Matthijs et al. (2016) reviewed applicability of cyanocidal compounds. Many algaecidal and/or cyanocidal compounds are not applicable in lakes due to lack of effectiveness or ecosafety. For example, chlorine (Daly et al. 2007; Zamyadi et al. 2013), potassium permanganate (Li et al. 2014), and ozone (Coral et al. 2013; Zamyadi et al. 2015) are effective in decaying cyanotoxins and the membrane integrity of cyanobacterial cells (Fan et al. 2013). These are not applicable in lakes due to the adverse effects on other aquatic organisms. Their use is mainly restricted to the treatment of process water for drinking water. Copper sulfate and hydrogen peroxide are used as cyanocide in surface water. Their applicability is described below.

Copper sulfate ($CuSO_4$) has been used for decades to control cyanobacterial blooms. It is both effective and relatively easy to apply (Van Hullebusch et al. 2002; Newcombe et al. 2010). Although cyanobacteria are particularly sensitive, $CuSO_4$ is a broad spectrum biocide. It can have adverse effects on other biota, for example zooplankton and fish, and tends to

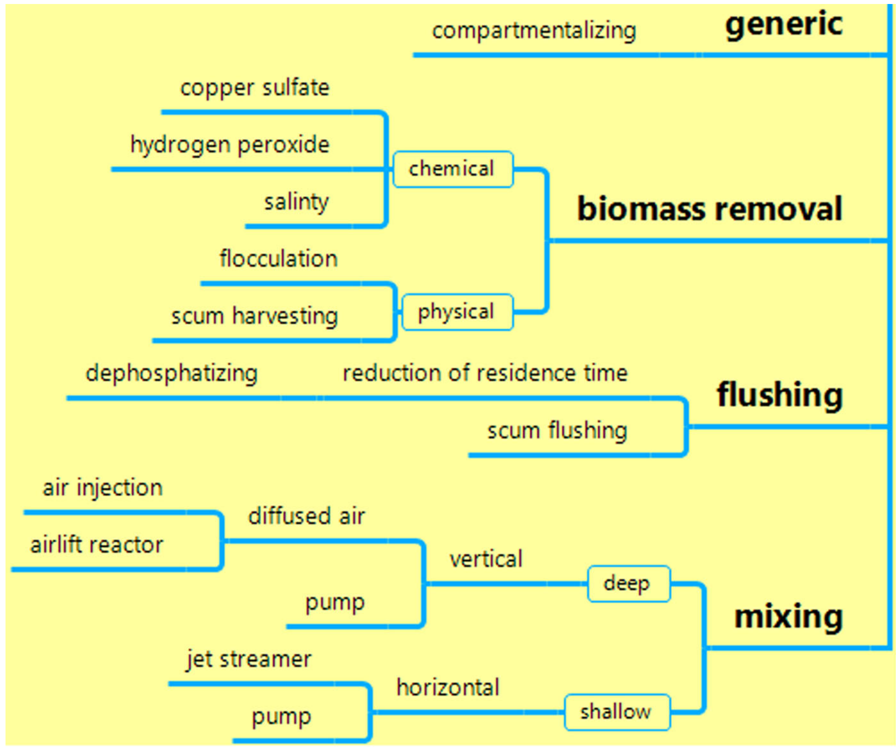

Fig. 6 Overview of control measures

Table 4 Summary of control measures, with indication of applicability

#	Measure	Cat	Applicability									
			Genus		Depth		Size		Costs		Risk	
			Sc	nSc	Sh	D	Sm	L	Lw	H	Lw	H
1	Copper sulfate	BR	x	x	x	x	x	x	x			x
2	Hydrogen peroxide	BR		x	x	x	x	x	x		x	
3	Flocculation	BR	x	x		x	x	x	x	x	x	x
4	Scum harvesting	BR	x		x	x	x		x		x	x
4	Reduction in residence time (+P removal)	Fl	x	x	x		x		x	x	x	
5	Scum flushing	Fl	x	x	x		x		x		x	x
7	Vertical mixing	Mx	x	x		x		x		x	x	
8	Horizontal mixing	Mx	x		x		x		x		x	x

Category: *gen* generic, *BR* biomass removal, *Fl* flushing, *Mx* mixing. Applicability: *Sc* scum forming genera dominate, *nSc* non-scum forming genera dominate, *Sh* shallow, *D* deep, *Sm* small, *L* large, *Lw* low, *H* high. "Costs" are the estimated relative costs per square meter. Under "Risk," the inherent risks of failure, unwanted side effects, and irreversibility are combined. If options vary, both sides are checked

accumulate in lake bottom sediments (Chorus and Bartram 1999; de Oliveira-Filho et al. 2004; Cooke et al. 2005; Newcombe et al. 2010; Jančula and Maršálek 2011; Fan et al. 2013). Copper is a priority substance in the WFD. Despite these considerations, copper is still one of the most used algaecides (Matthijs et al. 2016).

Hydrogen peroxide (H_2O_2) is a potent algaecide and herbicide in general, but particularly selective for cyanobacteria (Barroin and Feuillade 1986; Drábková et al. 2007b; Matthijs et al. 2012; Weenink et al. 2015). H_2O_2 reduced *Anabaena* sp. and *Microcystis* sp. at H_2O_2 doses of 10 and 1.7 mg l^{-1}, respectively (Kay et al. 1982a), and destroyed *Planktothrix rubescens* at 1.7 mg l^{-1} within a few hours (Barroin and Feuillade 1986). At 2 mg l^{-1}, *Planktothrix agardhii* was effectively and selectively targeted (Matthijs et al. 2012). *Aphanizomenon flos-aquae* inhibits nitrogen fixation at 0.9–3.4 mg l^{-1} (EC_{50}, 22 h) (Peterson et al. 1995). Lakes dominated by *Microcystis aeruginosa*, *Aphanizomenon*, and *Dolichospermum* (formerly called *Anabaena*) have been successfully treated with the hydrogen peroxide applied at 2.3–5 mg l^{-1} (Matthijs et al. 2016). In laboratory assays, 1 and 2 mg l^{-1} H_2O_2 imposed mild effects on *Microcystis aeruginosa*, while 4 and 8 mg l^{-1} H_2O_2 reduced photosynthetic efficiency to virtually zero (Lürling et al. 2014).

In general, zooplankton seem sensitive if exposed for longer than 24 h in the range of 4–10 mg l^{-1} (Kay et al. 1982a; Schmidt et al. 2006). Some mollusks (*Physa* sp., *Dreissena polymorpha*) are damaged at concentrations of 10 mg l^{-1} and above (Kay et al. 1982a; Martin et al. 1993; Schmidt et al. 2006). To avoid damaging or killing nontarget species, end concentrations should be limited to 2.5 mg l^{-1} (Matthijs et al. 2012), or 5 mg l^{-1} if decay rates in the lake water are high (Weenink et al. 2015). Decay rates increase with algae biomass, metal concentrations, and irradiance (Drábková et al. 2007a). Toxicity to the aquatic biota is positively related to temperature (Gannon and Gannon 1975) and irradiance (Kay et al. 1982b, 1984; Drábková et al. 2007a). One of the major advantages of hydrogen peroxide is the absence of harmful remnants in the environment, since H_2O_2 decays in water and oxygen in a few hours or days (Quimby 1981; Drábková et al. 2007a; Matthijs et al. 2012; Weenink et al. 2015) and the cyanotoxin microcystin within several days (Matthijs et al. 2012).

Increasing salinity can block freshwater cyanobacterial blooms. *Aphanizomenon* reduces its growth rate to 30 % at 7 g l^{-1} (Teubner et al. 1999). *Microcystis* might bloom up to 12.5–14 g l^{-1} (Verspagen et al. 2006a; Tonk et al. 2007) or 25 g l^{-1} (Robson and Hamilton 2003). *Anabaena* reduces its growth rate to 30 % at 9 or 15 g l^{-1} (Singh and Kshatriya 2002). Here this option is not explored; increasing up to these ranges will have direct and drastic effects on all aquatic biota, may induce increased sediment phosphorus release (Barker et al. 2008), and may introduce marine noxious species such as *Nodularia* (Rakko and Seppälä 2014).

Physical removal One of the successful traits of several genera of cyanobacteria is their ability to regulate their buoyancy, reducing their sedimentation loss rate. Cyanobacteria can form a dense scum (Kardinaal 2007) if the weather circumstances are favorable (e.g., low turbulence). In the case of scums, wind-driven accumulation near shorelines can pose a health threat due to extremely high cyanobacterial and cyanotoxin concentrations. The scum can be removed by superficial skimming (vacuum cleaning). Large lakes might generate a prolonged transport of biomass toward the shores. This makes the method efficient only in smaller waterbodies (e.g., urban ponds) or sites that are compartmentalized. The efficacy depends on the density of the scum layer. In the case of dense layers, the biomass yield can be high. The potential high toxicity reduces the options to dispose of the biomass. It might be processed in a communal wastewater treatment plant, or disposed of in a fermenter.

After dosing a flocculation agent, trapped particles, including cyanobacteria, sink to the sediment. The treatment has stronger effect in deeper lakes, since in shallow lakes the wind-induced mixing may result in the resuspension of the cyanobacterial cells (Van Hullebusch et al. 2002). A side effect might be the removal of zooplankters by flocculation (van Oosterhout and Lürling 2011). The use of clays for the flocculation of *Microcystis aeruginosa* can be effective, as described by, for example, Pan et al. (2006a) for several clay types, and by Verspagen et al. (2006b) for bentonite and kaolinite. The use of in-lake sediments is suggested by Pan et al. (2006a), Zou et al. (2006), and Pan et al. (2006b). In this way, no allochthonous additives are introduced when controlling cyanobacterial blooms and legal requirements are possibly easier to satisfy. Zou et al. (2006) conclude that modification with biodegradable chitosan can turn local soils into effective *Microcystis aeruginosa* scavengers, especially in freshwater. In jar tests, Pan et al. found (2006a) a removal efficiency of up to 95 % in 75 min for *Microcystis aeruginosa*, depending on the salinity and chitosan modification of the used sepiolite and kaolinite. In a 32 m^2 enclosure in Lake Taihu, a removal efficiency of 99 % of chlorophyll a (i.e., from 2000 to 20 μg l^{-1}) and 42 % of dissolved phosphate was obtained with chitosan-modified local soils (Pan et al. 2006b). Salts based on iron [ferric chloride (Fe^{3+}), ferrous chloride (Fe^{2+}), iron sulfate (Fe^{3+})] or alum [aluminum nitrate, aluminum sulfate, and the polymer polyaluminum chloride (PACl)] are widely used for mostly phosphate precipitation but also for direct algae flocculation (Lelkova et al. 2008; De Julio et al. 2010; Jančula and Maršálek 2011). Lelkova et al. (2008) successfully flocculated *Planktothrix agardhii* dosing PACl in a fishpond. Sulfate compounds are less suitable due to the possible formation of iron sulfide (FeS), which reduces the phosphate binding capacities of lake sediments (Hansen et al. 2003). Iron compounds are less suitable in deep lakes due to their redox sensitivity (see "Nutrients" section). The side effect of the precipitation of inorganic phosphates besides the flocculation of algae might decrease the needed frequency of repetition compared to just removing algae biomass. To increase overall efficiency, a treatment of salts combined with clay can be used. This enhances the sedimentation properties and minimizes the amount of cyanobacterial biomass that can escape from the flocs. Lürling and Van Oosterhout (2013) demonstrated that the combination of polyaluminum chloride (PACl) with enriched bentonite clay (Phoslock™) was effective in stripping the water column of dissolved phosphate and floating particles, including cyanobacteria, effectively locking them on the lake bottom. In the above-mentioned case studies, flocculation and sediment capping (see "Nutrients" section) were combined.

Flushing

A bloom of cyanobacteria in ponds, lakes, or reservoirs can be prevented if the residence time of the waterbody can be reduced in relation to the growth rate of the cyanobacterial cells. Based on 15 published maximum growth rates of cyanobacteria [*Microcystis* (6), *Anabaena* (2), *Planktothrix agardhii* (2), *Aphanizomenon flos-aquae* (2), *Cylindrospermopsis raciborskii* (3)], maximum growth rates range from 0.1 to 1.0 day^{-1} (mean 0.5 day^{-1}) (Lee and Rhee 1999; Wallace et al. 2000; Dokulil and Teubner 2000; Wilson et al. 2006; Imai et al. 2008; Lürling and Tolman 2014). This generates a doubling time ranging from 0.7 to 7 days and implies that to control blooms by flushing, residence times should be reduced to a few days or weeks. In natural stagnant waters, this is feasible only in smaller lakes and ponds, or after compartmentalization ("Compartmentalization" section). Sufficient flushing water with a low level of cyanobacteria should be available. In Lake Sloterplas (Amsterdam), a watertight 3100 m^3 compartment was created with oil slick booms reaching the lake sediment as a temporary (2013–2014) bathing water measure. Water supplied from 10 m depth prevented high concentrations of cyanobacteria from entering the compartment. The hydraulic residence time was 2.3 days. This resulted in fewer bathing water close-downs (extending the bathing time by a few weeks) compared to the situation outside the compartment. A possible side effect of using a compartment is that low concentrations of cyanobacteria entering the compartment might accumulate. Creating an outlet construction that provides a top layer exit from the compartment can reduce this risk (report Waternet 2015).

Mixing

Mixing in the horizontal direction is based on dilution, hence preventing scum formation. In the Netherlands, some more or less successful cases are known in a bathing pond (Zwemlust, Utrecht) and a marina (Lake Braassemermeer, Roelofarendsveen). The applied devices were propellers. In Lake Braassemer, a mixer capable of creating a flow of 750 $m^3\ h^{-1}$ kept away cyanobacterial scum up to a distance of around 100 m (unpublished), with the risk of displacing the scum problem elsewhere.

Vertical mixing will distribute the typically buoyant cyanobacterial cells over the entire lake depth. This will limit light availability for cyanobacteria and decrease the sedimentation losses of other phytoplankton, which will in effect decrease cyanobacterial dominance and dilute the remnants of the cyanobacterial population (Visser et al. 1996; Huisman et al. 1999, 2004). Visser et al. (2016) show in a review that vertical mixing is only effective in lakes deeper than a mean of >16 m or a maximum of >30 m for *Microcystis*, or less for other species (Table 1 of Visser et al. 2016). This implies that is not suitable for small lakes. The deeper the mixing, the more effective it is. Efficacy will also depend on the overall morphology of the lake or reservoir. In the case of an insufficient mixing depth, vertical mixing may even enhance cyanobacterial growth, as more nutrients become available from the hypolimnion. The accompanying enhanced oxygen levels near lake sediments might reduce sediment P release and the release of unwanted metals in drinking water reservoirs. On the other hand, the mixing sharply increases hypolimnetic water temperatures to slightly below regular epilimnetic levels, which is not desirable in drinking water production, and disables a potentially valuable energy-saving asset: the use of a lake hypolimnion as a potent cooling agent. Continuous or intermittent mixing should be carried out throughout the growing season of the cyanobacteria (Visser et al. 2016). The construction, maintenance, and energy costs are relatively high. Successful vertical mixing uses 67 kW km^{-2} in Lake Nieuwe Meer (Jungo et al. 2001) and 31–63 kW km^{-2} in Lake Zegerplas (unpublished).

Selecting measures

Based on the ecological water quality targets ("Ecological targets" section), a quantitative lake assessment ("Lake system analysis" section), and the available preventive and control measures ("Measures" section), the selection must narrow down to optimal measures (phase 3 in Fig. 1). Related to the downstream transport of a poor chemical and biological water quality, in many cases preventive measures (lake restoration) should be preferred over control measures. However, a set of preventive measures can proof to be difficult to optimize due to conflicting or synergetic ecological effects.

If an overall ecological improvement is the main target, a preventive system-altering approach is virtually obligatory, since most control measures hardly influence the food web or nutrient status, and these determine the ecological state. Vertical mixing and flocculation might be the exceptions. Flocculation not only removes cyanobacteria, but can also temporary increase the overall light conditions and remove phosphates. If morphological and design conditions are met, vertical mixing is a proven technique that will reduce cyanobacterial biomass substantially. This will increase macrophyte development, but macrophyte species diversity may remain low due to an unaltered nutrient loading.

In drinking water reservoirs, a preventive approach is preferable to ensure few fluctuations in the water quality and a low risk of external pollution. Large catchment areas and high pressure for fast results might make the quest for a preventive approach a serious challenge, invoking the need for control measures. Artificial oxygenation or hypolimnetic withdrawal might be suitable for drinking water reservoirs and have minor negative effects on the drinking water production process. Vertical mixing is effective, although the steep increase in the deep water temperatures might be detrimental for the drinking water industry. The absence of chemical remnants and relatively low costs offer good prospects for the use of hydrogen peroxide.

In waters with other functions—like bathing, recreation, or fishery—or in urban ponds, the targets may vary widely, as will the potential measures. Control measures in those cases might well be more feasible and affordable than preventive measures.

The aspects mentioned in Tables 3 and 4 also generate certain shifts in applicability.

- *Depth*. In shallow lakes, the number of possible preventive measures is greater than in deep lakes, because typical deep lakes lack significant amounts of littoral area. Vertical mixing and flocculation are typical deep lake measures. In shallow lakes, preventive measures can be considered failures if the switch from a turbid to a clear state is not established. In deep lakes, the chances are that the measures will at least reduce the duration or intensity of the cyanobacterial blooms. Consequently, in deep lakes considering a stepwise approach may be useful to achieve the targets without overacting. Especially in shallow lakes, next to engineering an exhaustive reduction in nutrients, indirect measures (hydromorphological adjustments) and direct measures (food web manipulations) to tip the balance to a clear state should be considered.
- *Size*. If a lake is relatively small (<1 km^2), more preventive and control measures are feasible. Smaller shallow lakes have a higher critical loading, which decreases the needed effort of measures. Thus, compartmentalizing increases the amount of options. To directly reduce cyanobacterial blooms and cyanotoxins, flushing to

transport algae is restricted to small lakes (ponds or compartments) or linear waterbodies. Especially in deeper and larger lakes, even if ecological targets are met due to preventive measures, cyanobacterial scums might accumulate in bathing areas, necessitating the need for local control measures as well.

- *Costs*. No rule of thumb is definable, since costs vary widely and they have not been thoroughly examined.
- *Speed*. In most cases, control measures provide instant results upon implementation, whereas it takes years to decades before preventive measures lead to results. Lake sediment P release is a well-recognized factor in slowing down lake restoration. It usually takes longer to implement preventive measures than control measures.
- *Risk*. Although there are ample tools and experiences available, any attempt to regulate aquatic ecology by means of preventive measures has an inherent risk of failure due to uncertainties in quantification and forecasting of ecosystem responses. In prevention, measures based on the diversion of inflows, creating storage, or P removal have the lowest risk of failure, but might need the largest financial budgets, especially in urban areas. The risk of failure seems high concerning internal loading. There are several uncertainties in quantifying the gross and net internal loading. And chemical capping—a common best practice measure—suffers from uncertainties in choice of compound, dose, and longevity. Furthermore, capping or dredging can be destructive to benthic communities. Control measures have fewer uncertainties and might be easier to intensify if results are not satisfactory.
- *Support*. In general, unknown or in-lake chemical measures like algaecides (control) or active capping (prevention by means of chemicals) receive little support from stakholders. The preventive approach includes reducing or cutting off nutrient sources. This might include isolating a lake from fluvial sources like rivers or canals, thus impairing recreational use or fish migration. Switching a shallow lake to a clear state will permanently decrease the fish stock and increase submerged macrophytes. Related to lake use (e.g., fishery, water-related recreation), such issues can prove to be of major importance, and the wishes of stakeholders can be more decisive in balancing measures. Upfront clarity in all the ecological and functional targets helps water managers to find optimal, feasible, and fundable measures.

Synthesis

Lake management to prevent harmful cyanobacterial blooms is a tailor-made process, based on functional targets and a detailed assessment of the lake in question, by means of a lake system analysis (LSA). A focus on P in the quantification of nutrient loading and nutrient measures will usually suffice. The nutrient limitation status of both N and P is important in lake assessment, whereas the achievable limiting situation is important for meeting ecological targets. P loading is far more manageable than N loading. P reduction alone can effectively reduce cyanobacterial blooms, while N limitation can largely be compensated for by N_2-fixing cyanobacteria. However, nitrogen loadings might have negative consequences for marine waters downstream, something that should be considered when implementing large-scale measures. Macrophyte species richness and abundance can make nitrogen reduction relevant to the improvement of the ecological status, especially in shallow lakes. If the littoral zone is very small, or cyanobacterial reduction is the only target, the species richness aspect is less important.

Most of the potential nutrient sources can be adequately calculated and then added up to arrive at a lake P budget. The gross and net internal loading might prove to be hard to quantify. The total calculated lake P loadings can be compared with the calculated critical (shallow, mean <3 m) or permissible/excessive (deep, mean >10 m) loadings, which are lake specific. In general, acceptable nutrient loadings are higher for shallow lakes compared to deep lakes.

Preventive measures are aimed at a stable clear water state with low cyanobacterial dominance. To switch from cyanobacterial dominance, a shallow system has to be turned over to a clear state. Limited nutrient loading is a precondition for successful lake restoration; hydromorphological adjustments and food web measures might help to tip the balance from turbid to clear, or to speed up the restoration process. In shallow lakes, all options available in the measures toolbox are valuable. In deep lakes, if reducing cyanobacterial abundance is the only target, hydromorphological adjustments and food web measures are not very useful. P reduction should suffice. Measures concerning P reduction vary widely and many can be considered as proven technologies. Measures to prevent internal loading seem to be the most challenging. Chemical capping of the lake sediment, being a common best practice measure, holds many uncertainties.

Control measures, here including mitigation strategies, are aimed at artificially reducing the negative impacts of cyanobacteria. As opposed to preventive measures, control measures are limited in number. Low-tech options are to accept the algae blooms but clean up the scums, or to disperse the scums/blooms and reduce the nuisances and risks. Feasible and scientifically proven techniques are based on the use of H_2O_2, flocculation, or vertical mixing.

In both prevention and control, compartmentalizing the waterbody might be opportune. This increases the critical loading for shallow lakes, and for smaller lakes, more feasible measures are available.

Exploring both of these fundamentally different approaches (i.e., prevention and control) increases the ability to select the optimal measures.

Future directions

Whereas ample evaluations of measures are available, there are few evaluations explaining why certain measures are selected and implemented. This paper aims to aid this part of the process. But costs are not incorporated. An overview of investments, economic depreciation, and annual running and maintenance costs can prove to be decisive in selecting.

Sustainability, durability, climate footprint, and the recycling of assets are current and future issues and are also important for lake management. For example, water as a valuable cooling agent, saving freshwater, and redeeming nutrients have only been touched upon, but these environmental issues should be taken into account when selecting measures. A broadened perspective on sustainable water management, materials, and energy may push researchers to discover other balanced sets of measures, with adjusted financial models.

Lake sediment P release alone can exceed the total acceptable P loading. There are uncertainties in quantifying the amount of net P released from lake sediment, its ecological effects, and optimal measures to prevent this internal loading.

The quantification of the acceptable loadings in deep lakes is basically founded on a single precondition of a springtime P concentration threshold. This suggests that there is room for improvement in deep lake assessment.

The invasive spread of the quagga mussel in North America and Western Europe raises questions about their effects and how their presence might interfere with the LSA and selection of measures. High densities of mussels may increase water clarity and benthic pelagic coupling, while high nutrient levels and low macrophyte diversity maintain. The net effect of the grazing properties of dreissenids on cyanobacterial population structure, the buoyancy regulation (including diel vertical migration) of different cyanobacteria taxa, and hydraulics has not been fully covered in the literature. Quantification of the carrying capacity of a dreissenid population and the resulting effects on the cyanobacterial population and ecological targets can be further explored.

Author contributions J. M. Stroom wrote the manuscript. W. E. A. Kardinaal co-authored “Control measures” and contributed to the structure and revisions of the manuscript. We thank the reviewers and editors for their valuable comments.

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