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Harmful Algae

Review

A critical review on operation and performance of source water control strategies for cyanobacterial blooms: Part II-mechanical and biological control methods

Faith A. Kibuye [a,*], Arash Zamyadi [b,c], Eric C. Wert [a,*]

[a] *Southern Nevada Water Authority (SNWA), P.O. Box 99954, Las Vegas, NV 89193, United States*
[b] *Walter and Eliza Hall Institute of Medical Research (WEHI), 1G, Royal Parade, Parkville VIC 3052, Australia*
[c] *Water Research Australia (WaterRA) Melbourne Based Position Hosted by Melbourne Water, 990 La Trobe St, Docklands VIC 3008, Australia*

ABSTRACT

This review summarizes current knowledge on mechanical (artificial mixing, hypolimnetic aeration, dredging, and sonication) and biological (biomanipulation, macrophytes, and straws) methods for the management of cyanobacterial blooms in drinking water sources. Emphasis has been given to (i) the mechanism of cyanobacterial control, (ii) successful and unsuccessful case studies, and (iii) factors influencing successful implementation. Most mechanical and biological control strategies offer long-term control. However, their application can be cost-prohibitive and treatment efficacy is influenced by source water geometry and continual nutrient inputs from external sources. When artificial mixing and hypolimnetic oxygenation units are optimized based on source water characteristics, observed water quality benefits included increased dissolved oxygen contents, reduced internal loading of nutrients, and lower concentrations of reduced ions . Treatment efficacy during oxygenation and aeration was derailed by excessive sedimentation of organic matter and sediment characteristics such as low Fe/P ratios. Dredging is beneficial for contaminated sediment removal, but it is too costly to be a practical bloom control strategy for most systems. Sonication control methods have contradictory findings requiring further research to evaluate the efficacy and applicability for field-scale control of cyanobacteria. Biological control methods such as biomanipulation offer long-term treatment benefits; however, investigations on the mechanisms of field-scale cyanobacterial control are still limited, particularly with the use of macrophytes and straws. Each control method has site-specific strengths, limitations, and ecological impacts. Reduction of external nutrient inputs should still be a significant focus of restoration efforts as treatment benefits from mechanical and biological control were commonly offset by continued nutrient inputs.

1. Introduction

Cyanobacterial blooms cause a globally significant challenge to water quality managers because of the adverse water quality and ecosystem impacts that range from oxygen depletion, reduced transparency, loss of aquatic biodiversity, and their potential to generate cyanotoxins and taste and odor (T&O) compounds.

The prevalence of cyanobacterial blooms has been linked to chronic surface water nutrient enrichment, particularly phosphorous (P) and nitrogen (N). Historically, P has been considered the primary limiting nutrient for eutrophication (Christophoridis and Fytianos, 2006; Søndergaard et al., 2003); however, the growth and toxicity of cyanobacteria can also be controlled by N (Gobler et al., 2016; Paerl et al., 2014). Furthermore, cyanobacteria have unique physiological adaptations allowing them to thrive in a wide range of environmental conditions, including low-nutrient water bodies (Reinl et al., 2021; Sterner

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* Corresponding authors.
  E-mail addresses: faith.kibuye@snwa.com (F.A. Kibuye), eric.wert@snwa.com (E.C. Wert).

et al., 2020).

Sediments can be a sink for external nutrient inputs originating from point and non-point sources . Depending on the biogeochemical characteristics of the sediment-water interphase, sediments can shift to become a source of nutrients to the water column through various internal nutrient loading processes (Orihe et al., 2017). Released nutrients can further maintain source water productivity and promote cyanobacterial blooms even long after external nutrients inputs are controlled (Paytan et al., 2017; Søndergaard et al., 1999).

The ideal cyanobacterial preventative methods are without a doubt limiting chronic external nutrient pollution into aquatic systems. However, watershed-level nutrient management strategies are often challenging. They are defined by cost, stakeholder participation and involvement, varying success levels for nutrient best management practices (BMPs), and spatiotemporal factors that influence nutrient management (Prokopy et al., 2019; Shortle et al., 2020). Consequently, external nutrient programs are often complemented with various source water control strategies used to manage internal nutrient loading or mitigate the growth of cyanobacteria. Source water control strategies refer to the management of nutrients or cyanobacteria at the lake or reservoir.

Internal nutrient loading can be controlled through the addition of nutrient sequestering metals (e.g., iron (Fe), aluminum (Al), lanthanum bentonite), physical/mechanical control methods (e.g., dredging, mixing, and hypolimnetic aeration), and biological control strategies (e.g., biomanipulation and planting of macrophytes). The proliferation of cyanobacteria has been controlled by using chemical agents such as algaecides (e.g., copper sulfate, hydrogen peroxide), physical methods like sonication, and biological strategies such as straws.

Chemical control methods are commonly used to provide rapid improvements in water quality and are relatively cheap (Matthijs et al., 2016). However, current chemical methods are limited by offering short-term bloom control, resulting in cell lysis and the generation of toxic metabolites (i.e., cyanotoxins and T&O compounds) and the fact that they leave behind ecologically relevant treatment residuals, e.g., Cu, Al, Ca, Fe, and La in water and sediments (Fan et al., 2014; Kibuye et al., 2021; Willis and Bishop, 2016) . Given these limitations, non-chemical control methods, i.e., physical and biological control strategies, are often sought after.

Effective source control of cyanobacteria and nutrients using physical and biological control strategies can depend on various site-specific characteristics, e.g., external and internal nutrient loading rates. When external nutrient inputs are not managed, different source control strategies are unsuccessful as source water conditions reverse to antecedent conditions (Jensen et al., 2015; Tu et al., 2020). Therefore, efforts to limit external nutrient inputs in aquatic systems should go hand in hand with *in-situ* nutrient and cyanobacterial control strategies. Other factors influencing successful cyanobacteria and nutrient control include operational aspects, environmental conditions, source water characteristics, water quality, and bloom conditions.

Here, we review the application, relative success, factors influencing treatment performance, and the ecological impacts of common physical (dredging, mixing, hypolimnetic aeration, and sonication) and biological (biomanipulation, macrophytes, and straws) methods implemented to control internal nutrients and cyanobacteria in surface water sources. This review is the second part of a two-part review on source control strategies for cyanobacterial blooms. Part one (Kibuye et al., 2021) reviewed chemical control strategies used to manage cyanobacteria in source water, highlighting (i) their modes of action, (ii) cases of successful and unsuccessful treatment, (iii) factors influencing performance, (iv) and the current knowledge on environmental impacts.

2. Biogeochemical processes of internal nutrient loading

Most lakes and reservoirs are commonly impacted by external nutrient inputs from various anthropogenic sources or riverine sources.

At steady-state, dissolved and particulate P entering a lake/reservoir is retained through natural sedimentation processes (Søndergaard et al., 2003). Lake sediments are composed of aggregates of minerals, organic, and inorganic species. Based on biogeochemical characteristics, some fractions of P inputs can be permanently bound to the sediment layer. In contrast, other P fractions are potentially mobile since they are bound to minerals, organic matter, and inorganic species in the sediment layer such as Fe , Al, calcium (Ca), manganese (Mn), and clay (Søndergaard et al., 2003). This inorganic P fraction occurs as a readily soluble orthophosphate anion found in sediments either as part of a mineral or as precipitated phosphate salts such as $CaO(PO_4)_3(OH)$, $FePO_4$, $AlPO_4$, and $Ca_3(PO_4)_2$ (Reynolds and Davies, 2001).

Depending on biogeochemical conditions, the inorganic P bound to metal species and minerals accumulated in sediments can be released to the overlying water column, effectively delaying improvements in water quality conditions following management efforts to reduce external P inputs. Redox conditions in the hypolimnion can cause reductive dissolution of the previously bound P in the form of soluble reactive phosphorus (SRP) as well as soluble states of metals such as Mn^{2+} and Fe^{2+} etc. into the water column (Christophoridis and Fytianos, 2006; Gibbs and Howard-Williams, 2018; Søndergaard et al., 2003). This reductive dissolution can increase under high pH conditions as hydroxyl ions compete with P and lowers the P binding potential of the oxygenated sediment layer (Søndergaard et al., 2003). P flux in anoxic conditions further increases at high temperatures (Gibbons and Bridgeman, 2020). Other mechanisms involved in releasing P from sediments include desorption of organic P from mineral surfaces, hydrolysis or mineralization of P-containing organic matter, and dissociation of ternary P complexes from organic matter (Orihe et al., 2017). Particle-bound fractions of P that settle in the sediment layer can also be resuspended into the water column due to bioturbation, wind, or rainfall-induced perturbations (Barbosa et al., 2020; Søndergaard et al., 2003).

The internal loading of P can be significant in some cases and maintain lake productivity even when external nutrient inputs are controlled. In Lake Erie, total P sediment flux was estimated to be range between 300-1250 metric tons per year, amounting to 8-20% of external P inputs to the lake (Paytan et al., 2017). The importance of internal P loading is further expected to vary between lakes. Robertson and Diebel (2020) found that internal P loading ranged from 3-60% of summer P inputs in a chain of four shallow lakes within the same drainage basin. Therefore, net internal P loading should be quantified first to understand how lakes and reservoirs will react to nutrient reduction strategies.

Internal nutrient loading can be quantified using three methods: (i) total and dissolved P load estimates derived from *in-situ* collected water samples, (ii) gross load estimates from P mass balances based on retention models, and (iii) gross load estimates from release rates measured in laboratory-incubated sediment cores (Burnet and Wilhelm, 2021; Robertson and Diebel, 2020; Song et al., 2017; Wang et al., 2021). Based on a cost analysis of the above quantification methods by Burnet and Wilhelm (2021), *in situ* water column samples and load estimates from sediment cores were the least expensive. However, *in-situ* sampling methods can be advantageous as this technique is more representative of whole-lake dynamics.

2.1. Mechanical/ physical control

Mechanical or physical control strategies typically include measures taken to manage in-lake processes of thermal stratification and subsequent consequences such as oxygen depletion and internal nutrient loading. As shown in Fig. 1, various physical control methods have been used to manage internal nutrient or contaminant loading and hypolimnion hypoxia. Artificial mixing and oxygenation processes oxygenate and destratify the water column. In contrast, hypolimnetic oxygenation or aeration processes increase oxygen levels while maintaining thermal stratification in the water column. In some cases, the nutrient-rich

Table 1
Summary of field and pilot-scale studies reporting on the performance of physical/mechanical control strategies for cyanobacterial control.

System	Source Characteristics	Treatment Goals & Design	Short term Response	Long Term Response	Ecological & Water quality Impacts	References
Hypolimnetic oxygenation	A: – Z_{Max}: 23 m Stratified: Yes	-Goal: Minimize Fe, Mn, and increase DO HOx flow: 1580 kg O_2/d	-Increase in sediment O_2 uptake -Sediment oxic-zone depth related to the O_2 gas flow in the water column	-Hypolimnion Mn level increased when HOx was off for 48 h. -Water quality was impaired when the system was off for weeks	-Lower reduced chemical species in the water column	(Bryant et al. 2011a; Bryant et al. 2011b)
Hypolimnetic oxygenation	A: 0.08-9.4 km^2 Z_{Max}: 9-38 m Z_{Avg}: 4-14 m Retention: 0.6- 12 yrs. Stratified: Yes	-Liquid O_2 transformed to gas, then transported to the deepest areas of the lake through pipes or tubes and released as small bubbles from different types of diffuser systems	-38-88% decline in TP and ammonia -Increased O_2 levels -Better trophic state in some lakes -35% decline in algal biomass in one lake	-Higher P when HOx was stopped in the lakes after 7 and 13 yrs. -Lower P accumulation 10-15 yrs. in the lake with low external P inputs	-Increase in hypolimnetic water temperature by 1-2°C	(Liboriussen et al. 2009)
Hypolimnetic oxygenation	A: – Z_{Max}: 15-70 m Z_{Avg}: – Vol: 1110-29,330 ha	-Control of internal TP, SRP, anoxic byproducts. -Oxygenation performed using bubble plume diffuser	-90% fewer T&O complaints in 3 yrs. of operation -50-66% lower SRP and TP months after startup - High nutrients 2 yrs. later at one lake but were lower than pre-treatment conditions -Lower Mn and Fe and high DO levels	-With controlled external P loading, the trophic state index for TP was lower (41) after 3 yrs. at one site -Maintained DO levels in successive years. DO spread laterally & longitudinally in the hypolimnion	–	(Mobley et al. 2019)
Hypolimnetic oxygenation	A: 3.1 km^2 Z_{Max}: 25 m Z_{Avg}: 5.8 m Vol: $18*10^{-2}$ km^3 Retention: 3 months Stratified: Yes	-Improved oxygenation system pumping 1m^3/s from 3 m surface to 12 m in the hypolimnion initiated in 1990. The system was turned off in 2012.	- Annual Chl-a and P levels did not deviate from the declining trend when the system was turned off.	-DO was lower in the years when the system was turned off -Trophic status did not deviate from the long-term trend of oligotrophication	-Significant alteration in the hypolimnetic temperatures, thus may limit habitat for cold-water species	(Kuha et al. 2016)
Hypolimnetic aeration	A: 26 km^2 Z_{Max}: 33 m Z_{Avg}: 6.8 m Vol: $18*10^{-2}$ km^3 Retention: – Stratified: Yes	-Oxygen-rich epilimnetic water is pumped into the hypolimnion	-Increased degradation of organic material -Degradation of organic material increased oxygen consumption -Destratified water column	- No significant change in deep zones surface sediment after 7 yrs.	-Increase in water column temperature and nutrient cycling	(Niemistö, Silvonen, and Horppila 2019)
Hypolimnetic oxygenation	A: 0.61-3.7 km^2 Z_{Max}: 15-62 m Z_{Avg}: 9.8-21 m Vol: $13-36*10^6$ Km^3 Stratified: Yes	-Design oxygen addition of 1000-4200 kg/d with a transfer efficiency of 80-98%	-Increased hypolimnetic DO -Hypolimnetic oxygen demand decline during year-long operation	Long term and effective water quality improvements with year-round system operation and when oxygenation is started before the onset of stratification	–	(Gantzer et al. 2019)
Hypolimnetic oxygenation	–	-Line diffuser systems operated intermittently at the beginning of stratification for mercury control	-Decreased Methylmercury in bottom waters during oxygenation -Methylmercury levels unchanged in surface water	–	-Declining fish tissue concentration of mercury -Mixing of nutrients in the water column	(Seelos et al. 2020)
Aeration	A: 2.2 km^2 Z_{Max}: 80 m Z_{Avg}: 56 m Vol: $5.7*10^7$ m^3 retention: 5 months Stratified: Yes	-Water lifting aerators were installed with gas flow of 20-50 m^3/h	-Completely mixed column with high DO -Anaerobic layers below epilimnion and above sediments due to algal decomposition	–	–	(Huang et al., 2019)
Aeration	Capacity: 28 Million m^3	-Two water lifting aerators were installed	-Increased DO in the bottom layers -Suppressed release of nutrients and ions in the first month -Low algal density near the aerators	–	-Mixing increased temperature in the lower layers	(Ma et al., 2015)

Table 1 (*continued*)

System	Source Characteristics	Treatment Goals & Design	Short term Response	Long Term Response	Ecological & Water quality Impacts	References
Aeration	A: 0.08 km^2 Z_{Max}:10 m Z_{Avg}: 4.3 m Vol: 5.7*10^7 m^3 Stratified: Yes	-Artificial destratification system operated prior to summer stratification	-Elimination of stratification -Isothermal conditions in the water column	-No significant differences TP, TN, and SRP in the water column between aerated and non-aerated conditions -Distribution of nutrients and ions in the water column that expanded algal growth	–	(Balangoda 2016)
Aeration	A: 1418 km^2 Z_{Max}: 95 m Z_{Avg}: 5.8 m Vol: – Retention: – Stratified: Yes	-Goal: Controlling reduced Fe and Mn. -Eight water-lifting aerators were installed. The low compressed gas flow rate was <20 m^3/h, and the high gas flow rate was 20-50 m^3/h	-Oxygenated and mixed water column -Effective control of Fe -Sediment Mn accumulation -Anoxic upper sediment and release of Fe and Mn when the system is off	–	–	(Li et al. 2019)
Dredging	A: 7.64 km^2 Z_{Max}:– Z_{Avg}: 1.2 m Vol: –	-5-10 d dredging for four months in 2017-18	-Decline in sediment release rates of C, N, P -Increased DO levels -Decline in cyanobacterial cell densities	–	-Decline in the abundance of bacterioplankton	(Wan et al. 2020)
Dredging	A: 82.7 km^2 Z_{Max}:– Z_{Avg}: 2 m Vol: –	-Dredging conducted in 500 m radius at 1.1 m depth with sediment volume of 863,900 m^3 removed	-Reduced sediment layer Ca-P contents and organic matter -Increased organic P & Fe-P	–	–	(Li et al., 2020)
Dredging	A: 19.91 km^2 Z_{Max}:– Z_{Avg}: 2.2 m Vol: –	-30-80 cm dredging depth -Dredging done using 4010 cutter 121 suction dredger and 0.3 m^2 grab dredger equipped with mud barge.	-Decrease TP, TN, and phytoplankton biomass and increased clarity in the lake -91% *Cylindrospermopsis raciborskii* cell density decline 1 yr. after dredging	-Water quality parameters returned to pre-dredging conditions after 2 yrs. -More than 90% increase in cell density after 2 yrs.	-Negative impacts on lake ecosystem health immediately after dredging, then improved 1-2 yrs. later and returned to initial health status.	(Jing et al. 2019); (Li et al. 2020)
Dredging	A: 0.09 km^2 Z_{Avg}: 1 m TP: ~1000–2000 µg/L in water and ~2500–4000 mg/kg in sediemnts	-Ponds was dredged to 2 m depth using a backhoe excavator	-95-99% decline in TP release rates. -Reduced SRP and pH in the water column and higher DO -Increased Ca, Al, Fe, and Mg in sediment -Reduced P sorption capacities	–	–	(Oldenborg and Steinman 2019)
Dredging	A: 2338 km^2 Z_{Max}:– Z_{Avg}: 2.2 m Vol: 4.4*10^9 m^3	-Dredging was performed in the upper 30 cm depth. -The dredged area was 93.65 km^2	-Decrease in SRP and Fe/P in sediments -SRP still higher in summer months post-dredging -Reduced biological activity in dredged sediments -High NH$_4$-N fluxes in dredged sediments -Limited effects on nitrates	-No noticeable improvement in water and sediment quality after 6 yrs.	–	(Chen et al., 2018); (Zhong et al. 2018)
Dredging	Canal L: 6 km W: 17.5-35.8 m	-Dredging was done using cutter suction dredgers for 41 d. The dredged part was 2,602 m long and 2.5 m in depth, and 91,920 m^3 of sediment removed	-Increased DO and oxidation-reduction potential -Shannon diversity of microeukaryotic and macrobenthic communities decreased	-No recovery in altered composition and structure of bacteria and microeukaryotes after 1 yr. of monitoring	–	(Zhang et al. 2017)
Sonication	A: 0.81 km^2 Z_{Max}:17 m Z_{Avg}: – Vol: 757 million	-6 months of sonication	-90% reduction of algal density after > 1 wk. of sonication	–	–	(Schneider, Weinrich, and Brezinski 2015)

Table 1 (*continued*)

System	Source Characteristics	Treatment Goals & Design	Short term Response	Long Term Response	Ecological & Water quality Impacts	References
	gal					
Sonication	A: 0.14-0.18 km^2 Z:6 m Z$_{Avg}$: 4.3 m	-5 months of sonication using 40 W, 40–50 kHz transducers	-No significant variations between test sites and controls.	–	–	(Purcell et al., 2013)
Sonication		-Sonication at 36 kHz -300 W, 108 kHz -450W, and 175 kHz-650 W	-80% decrease in sound pressure with 0.5 m increase in distance from the transmitter -Decline in cells density	–	-Release of toxins	(Park, Son, and Lee 2019)

HOx: Hypolimnetic oxygenation; SD: Secchi Depth; A: Surface Area; Z$_{Max, Avg}$: Maximum, average depth; Vol: Volume; DO: Dissolved Oxygen; DOC: Dissolved Organic Carbon; Cond.: Conductivity; Al: Alkalinity; –: Not reported; P: Phosphorous; TP: Total Phosphorous; SRP: Soluble reactive P; TIN: Total Inorganic Nitrogen

sediments are removed from the source water through dredging. In this section, these physical control methods are discussed in addition to sonication methods of algal control. Case studies for these control strategies are summarized in Table 1.

2.1.1. Artificial mixing

Mechanism of nutrient and cyanobacteria control. Artificial mixing is the mechanical manipulation of water circulation within a lake/reservoir to weaken or eradicate the density stratification of the water column. The circulation induces downward water movement that shifts heat and oxygen towards the bottom layers of the water column (Gibbs and Howard-Williams, 2018). Water column mixing allows for simultaneous destratification and oxygenation by causing hypolimnion water to mix uniformly with the water column leading to a near-isothermal state and dissolved oxygen (DO) increase in eutrophic lakes (Fast et al., 1973; Hasan et al., 2013). Oxygenated conditions during mixing can suppress internal nutrient loading to promote nutrient-limited conditions. However, the internal nutrient reduction is expected to play a minor role in systems where external nutrient loading is not controlled.

Cyanobacteria have a competitive advantage over other phytoplankton as they can modulate their buoyancy in the water column to enhance their access to light and nutrients (Visser et al., 2016). When the water column mixing rate exceeds the vertical floatation velocity of cyanobacteria cells, the cells are destabilized and sink to light-limited depths in the water column (Huisman et al., 1999; Visser et al., 2016). Lower light dose to cyanobacteria will result in a suppressed growth rate and a shifted competitive advantage to diatoms and green algae that are well adapted to fluctuating light conditions (Huisman et al., 2004; Visser et al., 2016). Numerical modeling work by Chen et al. (2018) indicated that during mixing, phytoplankton biomass density is controlled by reduced growth rates as a result of light limitation in deep reservoirs (>20 m) as opposed to control due to sedimentation losses in shallow reservoirs. The shift of cyanobacterial dominance from the surface to bottom layers with suppressed growth has been reported in some field-scale studies (Ma et al., 2015; Zhou et al., 2019).

Impacts on water quality. Various studies have indicated that water column mixing resulted in high DO levels and complete destratification signified by homogenous temperature and DO distribution in the water column (Huang et al., 2019; Ma et al., 2015; Visser et al., 1996; Zhou et al., 2019). Following a one-month operation of water lifting aerators in a stratified eutrophicated reservoir, DO concentrations increased from 0 mg/L to 5 mg/L (Ma at al., 2015). The oxidized conditions further resulted in a reduction in the release of internal pollutants noted by concentrations of dissolved P, N, Fe, and Mn that declined by 59%, 68%, 59%, and 84%, respectively (Ma at al., 2015). Similarly, Zhou et al. (2019) noted a 48%, 67%, and 23% decline in total nitrogen (TN), total

phosphorus (TP), and total organic carbon (TOC), respectively, following mixing.

Since mixing destabilizes cyanobacteria, there is often an increase in sedimentation of organic matter, i.e., cyanobacteria biomass (Zhou et al., 2019). At high sedimentation and decomposition rates, an oxygen deficit is created that limits the vertical exchange of DO (Huang et al., 2019; Niemistö et al., 2019). If the sediment oxygen demand exceeds the maximum supply through aeration, the sediment surface remains anoxic and the internal loading of nutrients may not be controlled (Gächter and Wehrli, 1998). With low sedimentation rates, aeration seemed to increase the degradation of the settling organic matter (Ma et al., 2015; Niemistö et al., 2019). Ma et al. (2015) further observed that higher oxygen levels can increase microbial metabolism, thereby accelerating the organic matter degradation rates shown by declining chemical oxygen demand during aeration.

Mixing causes perturbation in the water column and sediments and can cause resuspension of nutrients resulting in the mixing of nutrient-rich pore-water across all depths (Balangoda, 2016; Becker et al., 2006; Fast et al., 1973; Tammeorg et al., 2020; Visser et al., 2016). This distribution of nutrients in the water column can expand algal growth resulting in unsuccessful cyanobacteria control (Balangoda, 2016; Steinberg, 1983; Tammeorg et al., 2020). The continued external nutrient deposition has also been a significant factor for unsuccessful aeration and mixing (Bormans et al., 2016; Heo and Kim, 2004; Tammeorg et al., 2020). Thus, external load reduction should be prioritized in source water management, even when implementing internal nutrient control initiatives.

Impacts on cyanobacteria control. Long-term monitoring indicated that aeration increased phytoplankton richness with an even distribution of available species and a shift from cyanophyte dominance to cryptophytes (Barbiero et al., 1996; Bürgi and Stadelmann, 2002; Hawkins and Griffiths, 1993; Heo and Kim, 2004; Steinberg, 1983; Wang et al., 2021; Yan et al., 2020). The changes in species composition resulting from mixing are primarily driven by shifts in the competitive balance between cyanobacteria and other phytoplankton. In well-mixed waters, diatoms and green algae are predicted to be dominant as they have a competitive advantage promoted by shading of sinking cyanobacteria (Huisman et al., 2004, 1999). In contrast, weak mixing reduces phytoplankton diversity since blooms of buoyant cyanobacteria will cause light limiting conditions for sinking on neutrally buoyant phytoplankton species (Huisman et al., 2004).

The extent of light limitation as a control mechanism varies between colonial and filamentous cyanobacteria species. Colony-forming cyanobacteria can adjust their buoyancy in the water column to exploit light, nutrients, and inorganic carbon at a broader range than filamentous cyanobacteria that commonly result in benthic mats. Jungo et al. (2001) reported that average floatation velocities for the filamentous cyanobacteria *Anabaena* (1.2 cm/h) and *Planktothrix* (0.8 cm/h) were significantly lower than those of *Microcystis* (2.7 and 11.2 cm/h).

Therefore, higher mixing velocities are needed to entrap colonies with high floatation/rising velocities such as *Microcystis* in the bottom layers than for unicellular or filamentous species (Jungo et al., 2001; Visser et al., 2016). Like diatoms and green algae that have low rising velocities (Chen et al., 2018; Huisman et al., 2004), mixing may increase filamentous and unicellular species due to reduced settling rates. Furthermore, light limitation may be of less importance during the control of unicellular or filamentous cyanobacteria as they are typically adapted to low light conditions (Burford and O'Donohue, 2006).

Effect of continuous vs. intermittent mixing on cyanobacteria control. Simulation studies of wind-induced mixing of the water column have shown that intermittent mixing promoted biomass accumulation and competitive dominance of *Microcystis* blooms while continuous mixing completely destabilized *Microcystis* cells leading to bloom collapse (Yang et al., 2020). Similar findings have been noted in the field as the continuous operation of artificial mixing devices decreased cyanobacteria biomass and prevented bloom formation (Jungo et al., 2001; Visser et al., 1996). Continuous mixing can successfully control cyanobacteria by maintaining well-oxygenated conditions and suppressing sediment release of nutrient release (Burns, 1994).

Intermittent operation of artificial missing devices is often preferred where energy savings are of interest. However, depending on environmental conditions, cyanobacteria can rapidly increase in the water column in periods without mixing (Antenucci et al., 2005; Becker et al., 2006; Burns, 1994; Jöhnk et al., 2008; Tsukada et al., 2006). The operation of artificial mixers can also resuspend nutrients in the water column (Balangoda, 2016; Becker et al., 2006; Fast et al., 1973; Tammeorg et al., 2020; Visser et al., 2016) to foster cyanobacteria in periods without mixing. Furthermore, incomplete destratification leads to the formation of a warm unmixed layer where the combination of high nutrients, temperature, and light promote cyanobacteria proliferation (Becker et al., 2006; Burns, 1994; Tsukada et al., 2006). The challenges of intermittent aeration can quickly be overcome when the operation is shifted to a continuous system, as was the case for one reservoir where the shift to year-round aeration triggered by stratification controlled cyanobacterial blooms for an extended period of 7 yrs. (Burns, 1994).

In some cases, continuous operation of artificial mixers has not yielded the desired improvement in water quality conditions (Antenucci et al., 2005), while intermittent mixing has successfully controlled cyanobacteria (Steinberg and Zimmermann, 1988). Successful use of intermittent operation requires site-specific optimization based on the source water characteristics, cyanobacteria species, and external and internal loading dynamics (Jungo et al., 2001; Steinberg and Zimmermann, 1988). The timing of the mixing operation may also be an essential factor for systems receiving intermittent mixing. When bubble plume operation is initiated before peak chlorophyll-a, which can align with peak cyanobacterial biomass, there was a more significant reduction in chlorophyll-a (Imteaz and Asaeda, 2000). Suppose mixers are not optimized to source water characteristics, bloom conditions, and nutrient dynamics; even continuous operation of mixers may not successfully control blooms or improve source water quality. Therefore, as summarized by Visser et al. (2016), for successful cyanobacteria control, the operation of mixing devices should be optimized to (1) attain adequate vertical mixing, (2) achieve ideal mixing depth to allow light limitation of cyanobacteria, and (3) scale based on source water size and geometry.

Environmental impacts. There are limited explorations on the impacts of whole column mixing on the ecosystem. The reviewed papers have shown beneficial ecological changes, such as promoting green algae, diatoms, and zooplankton (Becker et al., 2006; Steinberg, 1983; Yan et al., 2020). Aeration also increases water column temperature (Niemistö et al., 2019). Although the overall impacts of water column warming are not widely investigated, there are potential impacts on

cold-water fish habitats. Since whole column mixing can result in the suspension of bottom lake constituents in the water column, exposure risks to aquatic organisms should also be assessed.

2.1.2. Hypolimnetic oxygenation and aeration

Hypolimnetic oxygenation or aeration techniques are implemented to combat hypolimnetic anoxia by maintaining or increasing DO levels in the hypolimnion while preserving thermal stratification. Maintaining oxic conditions in the upper sediments suppresses the release of P and other reduced species (i.e., Fe, Mn, and NH_4). Minimal destratification reduces the mixing of nutrient-rich hypolimnion water to the epilimnion (Beutel and Horne, 1999; Bormans et al., 2016). Thus, a variety of water quality benefits have been reported following oxygenation. In one drinking water utility, there was >90% decrease in taste and odor-related customer complaints (Mobley et al., 2019). A eutrophic Canadian lake shifted from eutrophic conditions to mesotrophic conditions characterized by lower chlorophyll-a (9μg/L) and TP levels (25 μg/L) following one year of oxygenation (Webb et al., 1997). In an Italian lake, declining algal biovolume and long term water quality benefits were observed for 10 yrs. (Toffolon et al., 2013). Extensive reviews have recently been published on the performance of hypolimnetic oxygenation/aeration systems, successful and unsuccessful cases of cyanobacterial control (Bormans et al., 2016; Preece et al., 2019). Therefore, this section will focus on factors influencing the performance of hypolimnetic oxygenation systems.

Effect of organic matter sedimentation. Hypolimnetic oxygenation or aeration treatments typically focus on maintaining hypolimnetic DO levels above 2 mg/L to diminish the redox-dependent release of P, Fe, Mn, etc. (Gächter and Wehrli, 1998). Several case studies reported an increase in hypolimnetic DO following treatments (Bryant et al., 2011b; Gächter and Wehrli, 1998; Gerling et al., 2014; Horne et al., 2019; Jaeger, 1994; Mobley et al., 2019; Munger et al., 2016).

Despite oxygenation of the water column, Niemistö et al. (2020) noted that an oxygen deficit at the sediment-water interphase occurred due to a significant increase in the sedimentation rates of organic material. Following an analysis of 11 lakes of varying depths (48–372 m), high oxygen fluxes were reported since only 2.6–6.3 mm of the sediments were penetrated by oxygen despite high DO levels in bottom waters (Steinsberger et al., 2020). The rates of accumulation of organic matter in the sediments regulate the flux of reduced substances from sediments (Steinsberger et al., 2017). Sediment oxygen demand during aeration is created by a combination of two main oxygen sinks: sediment oxygen uptake, which is dominated by the rapid aerobic mineralization of freshly deposited organic matter, and the flux of reduced substances from the sediment layer (Müller et al., 2012). Studies by Matzinger et al. (2010) indicated that aeration/oxygenation control measures may not mitigate organic matter deposits in the sediments. If the sediment oxygen demand equals or exceeds the maximum supply in an aerated eutrophic lake, the sediment surface will stay anoxic despite the oxic state of the hypolimnion water (Gächter and Wehrli, 1998).

Effect of maintained anoxic states in the sediment. The internal P cycling and reduced species dissolution will be unaffected when oxygenation does not produce an oxic state in the sediment surface. This phenomenon was observed by Gächter and Wehrli (1998) and they concluded that increased hypolimnetic DO concentrations did not increase the P retention capacity of the sediment in the studied lakes. Similarly, Moosmann et al. (2006) found that the difference in P retention capacity between aerated/oxygenated lakes and lakes with anoxic hypolimnion was insignificant.

In eutrophic systems, DO penetrates and is consumed only at a very thin layer at the sediment surface due to the large pool of reduced materials in the sediment and the diffusive boundary layer (Lorke et al., 2003). Since the bulk sediments may remain in their anoxic states, the

redox-dependent release of nutrients, Fe, and Mn can continue despite oxygenation. Gantzer et al. (2009) studied the behavior of Fe and Mn following the installation of an oxygenation system in a reservoir. They observed that although there was low Mn (0.05 mg/L) in bulk hypolimnion water, Mn levels in the near benthic region were high (> 2 mg/L) regardless of the near sediment DO levels > 5 mg/L. Continued release of Fe and Mn from the sediments even during well-oxygenated conditions was also reported by Munger et al. (2019). When high sediment and pore water DO levels are maintained, released Mn and Fe can be retained within the benthic region to keep bulk hypolimnion water levels low (Bryant et al., 2011b).

2.1.2.8. Influence of sediment characteristics. In addition to the oxic state of the hypolimnion water and sediments, P retention is also influenced by sediment composition and characteristics. Jensen et al. (1992) showed that the Fe:TP of sediment is a measure of free sorption sites for orthophosphate ions on hydroxyoxide surfaces; therefore, if this ratio is kept above 15 then the surface remains undersaturated with sorbed P. Due to a near-saturation of sorption sites as indicated by low Fe:TP in a lake, P was liberated even in oxic conditions (Christophoridis and Fytianos, 2006). Tu et al. (2020) further noted that periods of high sedimentary retention of TP and labile P fractions occurred in Fe and Mn enriched sediment layers during seasonal oxic conditions.

In anoxic sediments, there is a likely reduction of sulfate (SO_4^{2-}) to sulfide (S^{2-}) that dissolves ferrous phosphates and precipitates Fe (II) as iron sulfide (FeS) (Gächter and Müller, 2003). The formation of FeS can prevent the reoxidation of Fe and subsequent sorption of P. Therefore, the molar ratio of the available reactive Fe(II):S^{2-}:PO_4 in the anoxic sediment is also vital to P binding (Gächter and Müller, 2003). This blockage of Fe as an insoluble sulfide can have consequences such as the formation of hydrogen sulfide in the hypolimnion (Kowalczewska-Madura et al., 2018) and cause a lack of Fe for the P binding and a resultant high P mobility (Kleeberg et al., 2012).

In addition to Fe, Al-rich sediment can also contribute to permanent retention of P due to sorption on aluminum hydroxide ($Al(OH)_3$) that is not affected by redox changes in the sediment-water interphase (Kopáček et al., 2005). However, as noted by Markovic et al. (2019), the sediment content of $Al(OH)_3$ must be high enough for effective P sorption, and this sorption may be limited by relatively high pH that can result in the dissolution of $Al(OH)_3$. The dependence of Fe and Al sediment contents on P sorption has prompted Fe and Al addition in lakes to increase P retention. Gołdyn et al. (2014) reported significant water quality improvements such as transparency, increased oxygen content from zero to 1 mg/L, and chlorophyll-a declined by 72% when Fe treatments were combined with oxygenation. A German lake where trophic states remained unchanged despite deep water aeration due to low Fe concentrations in the water column implemented phosphate precipitation using Fe in addition to aeration and observed declined P levels (Jaeger, 1994).

2.1.2.9. Influence of hypolimnetic oxygenation system operation. Due to budget constraints, some systems opt for the seasonal operation of hypolimnetic oxygenation i.e. when hypolimnetic oxygen demand is high as opposed a continuous operation. Numerous studies have shown hypolimnetic oxygenation can substantially improve water quality if operated before the onset of stratification/hypolimnetic oxygen demand or operated continuously (Bryant et al., 2011a, 2011b; Preece et al., 2019). Long term (7-13 yrs.) bubble plume oxygenation system in three lakes decreased hypolimnetic oxygen demand in all the lakes after several years, but the annual decrease in hypolimnetic oxygen demand was more pronounced in lakes that were oxygenated year-round as opposed to those that initiated oxygenation once DO had declined to >5 mg/L (Gantzer et al., 2019).

The impacts of discontinued oxygenation can be rapid and are observable even following short-term shut-offs such that sustained water quality improvements rapidly dissipate in cases where oxygenation systems are implemented for only short durations. An oxygenation system implemented to control Mn in a drinking water reservoir noted a substantial increase in soluble Mn levels when the system was shut off in as little as 48 h (Bryant et al., 2011b). Inoue et al. (2017) further noted that although oxygenation can affect P binding at the sediment surface, dramatic P release is expected within days of discontinuing the oxygenation system. Five Danish lakes exhibited varying responses characterized by improvements in water quality and cyanobacteria biomass, while some lakes exhibited no responses as indicated with relatively constant chlorophyll-a levels (Liboriussen et al., 2009). In this study, the authors concluded that the oxygenation system yielded minimal effect on P accumulation and there was an observed increase in P with halted oxygenation.

Varying response to water quality and cyanobacteria control due to operational characteristics can also be attributed to varying external nutrient loading dynamics. The long-term benefits are often significant if external P loading is also controlled. For example, the P trophic index in a reservoir declined in just three years with controlled external nutrient loading (Mobley et al., 2019). After 23 yrs. of oxygenation in a Finnish lake, it was concluded that the shutdown of the oxygenation system would not impact the lake's trophic status when the external P loading was lowered or maintained (Kuha et al., 2016).

2.1.2.10. Influence of source water characteristics. Functioning of hypolimnetic oxygenation systems can be limited by water column depth, although successful implementations have been performed in both shallow lakes (max depth 9–10 m) (Gerling et al., 2014; Toffolon et al., 2013) as well as deep lakes (>20 m) (Preece et al., 2019). However, hypolimnetic oxygenation and aeration may be limited if the source is extremely shallow, even with stratification, since there may be a low-density gradient to control thermocline erosion (Bormans et al., 2016).

Source water size also is important when sizing and designing oxygenation systems as larger lakes/reservoirs may have cost limitations. The size and depth may also influence the type of system selected. For instance, bubble plume diffusers may be more suitable for deep lakes where bubbles dissolve in the hypolimnion and the plume-generated momentum is significantly low to minimize thermocline erosion (Chen et al., 2017). When a line diffuser oxygenation system was used in small reservoirs, Seelos et al. (2020) noted unanticipated transport of bottom layer constituents to the water column. Thus, shallow and small reservoirs may require oxygenation systems that limit vertical turbulence.

2.1.2.11. Environmental impacts of hypolimnetic oxygenation and aeration. A common impact of aeration is the increase in the water column temperature. Hypolimnetic oxygenation in a shallow dimictic lake resulted in a 9°C increase in water temperature and a weakened thermal stratification that triggered a premature summer lake overturn (Toffolon et al., 2013). Kuha et al. (2016) noted an average temperature increase of 0.092°C/d while temperature increases ranged from 0.5-6°C in Danish lakes (Liboriussen et al., 2009). The causes of deepwater heating are not well understood but are assumed to be due to exothermic reactions in the sediment zone due to DO level increase (Toffolon et al., 2013). This increase in temperature may impact aquatic life. Studies have reported that restoring hypolimnetic oxygen expanded cold-water fish habitats (Preece et al., 2019). In a study assessing the influence of hypolimnetic oxygenation on trout, Cross et al. (2017) noted no enhancement in trout population indicators such as habitat volume and prey availability while oxygenation resulted in optimal conditions for smelt population in another lake (Ruuhijärvi et al., 2020). Depending on source water characteristics and how the oxygenation system is operated , there may be unintended mixing of bottom water in the water column that can result in fish exposure to metals (Seelos et al., 2020).

2.1.3. *Dredging*

Dredging is a geoengineering method that involves excavating sediments in surface water sources and relocation to a disposal site. By directly removing the nutrient-rich sediment surface layer, dredging can control the internal loading of nutrients into the water column, thereby used to control cyanobacterial blooms. Dredging is a controversial nutrient control method since it is relatively costly and causes anthropogenic disturbance on the sediment-water interphase that influences aquatic ecosystems (Paerl et al., 2020). The high cost for dredging projects is often associated with logistical challenges such as water draining, pumping, treating the overlying water, and transportation of the dredged sediment material to a disposal site (Oldenborg and Steinman, 2019).

The primary advantage of dredging as a control strategy compared to other nutrient control methods is the removal of contaminated nutrients from the system, reducing the potential for future internal nutrient release. In addition to creating a nutrient-limited environment, dredging results in increased DO levels, altering the sediment-water biogeochemistry, such as impacting the bacterioplankton community, thus resulting in both a direct and indirect control of cyanobacterial proliferation (Wan et al., 2020). While some dredging case studies have reported significant improvements in water quality, a good amount of studies have also reported unsuccessful trials.

2.1.3.12. *Impacts on water quality and cyanobacterial control*. An immediate water quality impact of dredging is the increase of turbidity in the water column. Dredging and dredged material can create turbid plumes that impact water quality at the dredged site as well as nearby aquatic systems depending on the characteristics of the dredged material and local hydrodynamic conditions (Evans et al., 2012; Fisher et al., 2015; Pledger et al., 2021). Although elevated turbidity during dredging may only be short-term (Pledger et al., 2021), high turbidity in drinking water sources may pose challenges during water treatment and impact product water distributed to customers (Gauthier et al., 2003). Dredging could also reduce surface water turbidity in the long run since limited sediment resuspension is expected following sediment removal (Li et al., 2020).

Sediment dredging may be a valuable technique for reducing sediment release of nutrients and heavy metals and improving DO concentrations (Chen et al., 2018; Oldenborg and Steinman, 2019; Sun et al., 2019; Zhang et al., 2017). Other post-dredging observations also included a decline in cyanobacterial cell densities, chlorophyll-a, and bacterioplankton (Wan et al., 2020). Most of these studies only monitored spatiotemporal water quality variations for limited periods (up to one year), hence not portraying treatment longevity of dredging. Dredging benefits in Lake Taihu were observed for 6 yrs. and pointedly decreased soluble Fe and SRP and labile Fe/P in sediments, however the results showed no apparent long term improvement in water and sediment quality after dredging, likely due to a mixing phenomenon between the dredged and the non-dredged sites that were located in the same bay (Chen et al., 2018).

Dredging commonly resulted in reduced internal P loading in most cases, but may not eliminate it. Oldenborg and Steinman (2019) analyzed sorption isotherms indicating that in oxic conditions, the sediment can still act as a source of soluble P when the average overlying SRP concentrations are < 40 µg/L. Even in post-dredged sites, higher SRP diffusion flux in the sediment-water interphase was still pronounced in summer months but was notably lower when compared with non-dredged sites (Chen et al., 2018). In a small scale dredging study (500 m radius, 1.1 m depth), dredging effectively minimized summer release of SRP, however, authors noted that the newly formed sediment layer contained increased levels of Fe bound P and organic P by 136% and 48%, respectively thereby enhancing the potential release of P into water column especially during anaerobic conditions (Li et al., 2020). Furthermore, the P adsorption capacity of the fresh sediment layer can

be decreased significantly (98%) after dredging due to potential sediment saturation with TP (Oldenborg and Steinman, 2019). This decline in P adsorption capacity is linked to reduced organic matter content following a deep dredging depth. Organic matter content correlates with P adsorption capacity of the sediments as it facilitates the formation of Al and Fe complexes that increase sites for P sorption (Darke and Walbridge, 2000). Sediment sorption capacity in post-dredging sediments may also vary due to site-specific water-sediment characteristics and dredging parameters. For instance, sediment cores from Lake Taihu indicated an increased P sorption potential in post-dredged sediments (Yu et al., 2017).

Treatment benefits from dredging projects are commonly negatively influenced by continued external loading that leads to P resupply to sediments and eventual release into the water column (Wen et al., 2020). One-third of Dongqian Lake (mean depth= 2.2 m) in China was suction dredged in the 30-80 cm depth and the impacts were observed during a 5 yr. period. Jing et al. (2019) noted a decreased nutrient and chemical oxygen demand concentrations in addition to increased water clarity in the entire lake that were maintained for two years before returning to pre-dredging conditions. When a lake frequented by *Cylindrospermopsis raciborskii* blooms was dredged, there was a 91% decline in cell density the following year, though cell densities were 30 times higher two years after dredging (Li et al., 2020). This lake recovery was projected to likely result from continued external TP inputs in the lake. Jing et al. (2015) recommended that sediment dredging should only be an option once external nutrient loading sources have been completely eradicated to have long-lasting treatment benefits. Given the costly nature of partial and whole lake/reservoir dredging projects and the fact that benefits are primarily short-term, researchers have recommended that dredging should be combined with other P load reduction or restoration strategies to achieve long term control benefits (Jing et al., 2019; Yenilmez and Aksoy, 2013).

2.1.3.13. *Influence of source water morphology*. When sediment dredging appears to be a reasonable nutrient management option for a source, morphometric and water quality characteristics such as sediment depth and distribution, chemical composition of the sediments (e. g., nutrients, metals, organic matter, etc.), external nutrient input, rate of nutrient or contaminant release, and sedimentation rates can be used to elucidate on potential sediment accumulation after dredging (Peterson, 1982).

Most of the dredging case studies were conducted in small shallow lakes (mean depth 2 m) and dredging depths ranged between 30-100 cm. Dredging is costly, slow, and likely unfeasible for reducing internal P loading in large lakes, where nutrient-rich sediments are highly mobile and widely distributed across the lake surface area (James and Pollman, 2011). For large lakes, sectional dredging may only remove small portions of the large sediment sizes inside the lakes yielding insignificant or very short-term benefits (Qin et al., 2019). Furthermore, when only small sections of a large lake are dredged, natural lake mixing dynamics hinder long-term sediment removal benefits (Chen et al., 2018). Partial dredging combined with other P load reduction methods may offer long-term management benefits (Yenilmez and Aksoy, 2013). Still, external nutrient sources should be managed before dredging projects (Jing et al., 2015).

2.1.3.14. *Environmental impacts of dredging*. Dredging can significantly reduce benthic macroinvertebrates, macrobenthic fauna, and aquatic plants (Barletta et al., 2016; Erftemeijer and Robin Lewis, 2006; Pledger et al., 2021). The impacts on aquatic life can originate from processes such as physical removal or damage, burial due to sedimentation, loss of habitat, and effects from increased turbidity (light attenuation) and toxic substances in the water column (Wenger et al., 2017; Wilber and Clarke, 2001).

The removal of sediments can alter the morphology and hydraulic

conditions of the dredged site and modify the long-term characteristics of the ecological community (Ellery and McCarthy, 1998). Following long-term monitoring in dredged and non-dredged sites in a lake, Jing et al. (2019) indicated that the ecosystem health index in dredged areas was lower than those in the non-dredged parts of the lake immediately following the dredging project. Over time, however, the negative impacts of dredging in the lake declined during lake recovery. In contrast, the composition and structure of bacteria and microeukaryotes were altered following dredging, though there was no recovery observed 1 yr. after river sediments were dredged (Zhang et al., 2017).

Dredging also presents challenges with the disposal of sediment materials that are costly and impacts the environment. Some disposal options include landfilling (Oldenborg and Steinman, 2019), in lake disposal that can create sediment islands (James and Pollman, 2011), and application of dredged sediments as fertilizer when trace metal concentrations and other pollutant levels are low (Staninska et al., 2014). Depending on the characteristic of the dredged material, these disposal options present additional environmental contamination pathways and risks (Bormans et al., 2016; Chen et al., 2003).

2.1.4. Sonication

Sonication treatment implements high frequency (>20 Khz) ultrasound for the control of cyanobacterial blooms. This is a relatively new technology that gained popularity because of its installation simplicity, ease of operation, lack of residuals, and minimal ecological impacts (Ahn et al., 2007; Heng et al., 2009; Park et al., 2017a, 2019; Purcell et al., 2013b; Schneider et al., 2015). Most studies on sonication are limited to single species during laboratory and pilot-scale studies, thus knowledge gaps exist on the optimization and scalability of sonication in large water bodies.

2.1.4.15. Mechanism of cyanobacteria control. The cyanobacterial inhibition mechanism of ultrasound irradiation is through mechanical damage to cellular structure and functionality (Phull et al., 1997; Rajasekhar et al., 2012a). Ultrasonic irradiation in water results in acoustic cavitation within the cell that entails the rapid generation of bubbles with localized hot spots of high temperatures (5,000°C) and pressures (500 atm) leading to their implosive collapse (Suslick, 1990). Such extreme conditions disrupt and collapse gas vacuoles responsible for regulating buoyancy in cyanobacteria cells (Reynolds, 1972). The disruption of vacuoles results in cell sedimentation (Lee et al., 2001) and subsequent reduction in photosynthetic activity due to minimal light exposure. Sonication also inhibits photosynthetic activities in cyanobacteria by damaging the intracellular and extracellular antenna complexes responsible for capturing photons for photosynthesis (Lee et al., 2001; Zhang et al., 2006a) . Additionally, ultrasonic irradiation results in the formation of free radicals from decomposing water vapor (Suslick, 1990) that can attack and weaken cyanobacteria cell walls depending on applied ultrasonic frequencies (Wu et al., 2012). However, recent work by Kurokawa et al. (2016) indicated that such chemical effects are only minor mechanisms for algal cell disruption.

2.1.4.16. Selectivity on cyanobacteria. Sonication treatment is advantageous because of its reported selectivity to cyanobacteria with gas vacuoles over other species lacking such structural characteristics. Selectivity to gas-vacuole cyanobacteria was demonstrated by Tang et al. (2004), whereby sonication at 1.7 MHz and 0.6 W cm^{2-} resulted in a higher cavitation yield and inhibited cell division and propagation in the gas-vacuolate *Microcystis aeruginosa* while the *Synechococcus* PCC 7942 that has no gas vacuoles grew as fast as the control. Similarly, when *M. aeruginosa* and *Anabaena circinalis* and a green algal species, *Chlorella* sp. were exposed to ultrasound, the former species had a more significant reduction in cell concentrations than the green algal species that lack gas vacuoles (Rajasekhar et al., 2012b). In a pilot-scale study, Ahn et al. (2007) reported the selective reduction of cyanobacteria

followed by diatoms and green algae dominance in the treated pond. The potential to selectively deactivate cyanobacteria species from aquatic ecosystems can make sonication a relatively safer strategy than chemical control strategies that often result in accidental impacts to non-harmful algal species and other non-target aquatic organisms.

2.1.4.17. Factors influencing ultrasound efficiency. The efficiency of ultrasound for cyanobacteria control is influenced by the applied frequency, intensity (power), and duration of exposure. Lower frequencies allow sufficient time for the cavitation bubble to peak resulting in stronger shockwaves during implosion that result in more efficient cyanobacterial removal with less power (Rajasekhar et al., 2012a). In contrast, high frequencies can result in shorter bubble life, minimizing cavitation's mechanical effects on cyanobacteria (Mason et al., 2011). While laboratory-scale studies investigate a wide range of frequencies (Tekile et al., 2017), most field-scale studies employ relatively low frequencies ranging from 22–50 kHz though some are as high as 200 kHz (Nakano et al., 2001). Higher intensities are expected to increase algal cell destruction (Zhang et al., 2006b) . However, the frequency and intensity are used to determine the exposure time, often ranging between a few seconds to several hours (Park et al., 2017b; Rajasekhar et al., 2012a; Wu et al., 2011). Although there have been several sonication studies at varying frequencies, power, and exposure time settings, there are no recommended combinations for optimal field-scale cyanobacteria control. Ultrasound operational parameters determined from laboratory-scale experiments may be unideal for full-scale assessments in larger reservoirs for extended duration of operation due to associated high energy costs (Purcell et al., 2013b).

2.1.4.18. Field and pilot-scale studies. Despite evidence to support the efficiency of ultrasound to control cyanobacterial growth in lab and pilot scale settings (Dehghani, 2016a; Rajasekhar et al., 2012a; Wu et al., 2011), contradictory field-scale findings are pointing out that ultrasound has minimal algal growth inhibition potential (Lürling et al., 2014; Lürling and Tolman, 2014). The consensus is that only a few tests have been conducted in full-scale reservoirs (Park et al., 2017a; Rajasekhar et al., 2012a) that present mixed results compared to those in controlled experiments that conclude ultrasound as an effective algal control technology.

When ultrasound treatment was implemented in three reservoirs in the UK for about eightmonths, Purcell et al. (2013b) found no significant differences in cyanobacteria, green algae and diatom cell counts, and chlorophyll-a between the controls and ultrasound treated reservoirs. In contrast, Schneider et al. (2015) found that four ultrasonic buoys installed in an 800,000 m^2 New Jersey reservoir for six months effectively controlled cyanobacteria when programmed correctly. Nonetheless, the operational parameters were not reported. Even though the water in the control reservoir was pumped into the treated reservoir and may have caused interferences in the treated reservoir, findings indicated a 93% reduction of *Aphanizomenon* based on cell counts and 63% removal of chlorophyll-a during the ultrasound application period (Schneider et al., 2015).

In Nakano et al. (2001) study, an 80-acre shallow recreational lake was treated with a combination of a jet circulator, aeration system, and ultrasonication with two localized 100 W, 200 kHz ultrasound transducers for a 5 s duration of exposure. As indicated by the decline in chlorophyll-a, suspended solids, and increased transparency, findings showed improved water quality. However, given the integrated treatment design in this study, it is difficult to discern how much the sonication units contributed to water quality improvements. Besides, when flushing rates in the lake were reduced in the final year, the *M. aeruginosa* bloom reappeared. Similarly, in the pond study by Ahn et al. (2007), findings did not offer compelling evidence of cyanobacteria control by ultrasound since reduced algal growth in the treated pond was only 7% of the control and green algae and diatoms dominated

Table 2

Summary of field and pilot-scale studies reporting on the performance of biological control strategies for cyanobacterial control.

Biological Control Type	Source Characteristics	Fish added/removed; Macrophyte/Straw Type	Treatment Period	Short and Long term response on cyanobacteria and Nutrients	Ecological & Water quality Impacts	References
Biomanipulation Fish removal	A: 14.8 km^2 Z_{Max}:5.4 m Z_{Avg}: 4.3 m	*Rutilus rutilus Abramis brama*	-First treatment: 1989-1992 -Treatment was repeated in 2005-2012	-First treatment benefits lasted till the mid-90s, then turbid conditions returned. -Insignificant changes in zooplankton mass and TP -Significant decrease in cyanobacteria and toxins -In the long term, *Daphnia sp.* became more dominant while cyanobacteria and toxins remained low	-Improvement in trophic levels	(Ekvall, Urrutia-Cordero, and Hansson 2014)
Biomanipulation Fish removal	A: 11 km^2 Z_{Max}: 13 m Z_{Avg}: 3 m	Cyprinid removal in and nutrient management programs	1992-1994	-Improved water transparency -Microcystis replaced with various phytoplankton -Development of submerged macrophytes -25-55% lower nutrient levels	–	(Annadotter et al. 1999)
Biomanipulation Fish stocking	A: 5.33 km^2 Z_{Max}:13.5 m Z_{Avg}: 7.4 m Vol: 39.2*10^6 m^3	*Sander lucioperca L. Esox lucius L, Silurus glanis L., Anguilla anguilla L.*	1981-1993 -Biomanipulation in conjunction with external P control	- Increase *Daphnia sp.* biomass -Phytoplankton increased and later dropped -Microcystis bloom occurred when external P was not controlled -Long term increase in *Daphnia sp* and improvements in water clarity	–	(Kasprzak et al. 2007)
Biomanipulation Fish removal and Piscivore addition	A: 1.36 Km^2 Z_{Max}:12 m Z_{Avg}: 6 m Vol: 8.2*10^6 m^3	*Rutilus rutilus Abramis brama*	195-2002 -Biomanipulation in conjunction with external P control	- Increase *Daphnia sp.* biomass -Bloom of filamentous cyanobacteria -Water clarity unchanged in the long term	–	
Biomanipulation Fish removal and Piscivore addition	A: 0.64 km^2 Z_{Max}:– Z_{Avg}: 3.1m Vol: –	*Esox lucius L. Stizostedion luciperca (L.)*	The reservoir was drained, and all fish removed, followed by stocking between 1993-1995	- Increase *Daphnia sp.* biomass -Cyanobacteria biomass dominated in the first year -Chrysophytes and diatoms dominated in the third year -Cyanobacteria abundance persisted in the long term	–	(Kozak and Gołdyn 2004)
Biomanipulation and Macrophytes	A: 0.6 km^2 Z_{Max}:2.5-3.0 m Z_{Avg}: – Vol: –	–	November 2015- July 2016	-Improvements in water clarity -Zooplankton to chl-a ratio higher 2 yrs. post biomanipulation -Lowered NH$_4$-N, chl-a, TP, TN, and dissolved P -Treatment benefits observed 2 yrs. post biomanipulation	-Adjustment of the aquatic ecosystem of the treated lake	(Z. Chen et al. 2020)
Biomanipulation and Macrophytes	A:- Z_{Max}:– Z_{Avg}: 2 m Vol: 2.4*10^7 m^3	*Ceratophyllum demersum L. Sarsilatona serricauda*	February 8th-18th 2019. Mesocosms were used to study the effect of macrophyte addition, zooplankton, and nutrient addition	-Macrophyte treatments showed significant cyanobacterial biomass reduction (85%). -Macrophyte more efficient for phytoplankton and cyanobacteria control than zooplankton addition -Comparable nutrient management for both macrophyte and zooplankton addition	–	(Amorim and Moura 2020)
Macrophytes	–		July-August 2008		–	(Wang et al., 2012)

Table 2 (*continued*)

Biological Control Type	Source Characteristics	Fish added/removed; Macrophyte/Straw Type	Treatment Period	Short and Long term response on cyanobacteria and Nutrients	Ecological & Water quality Impacts	References
		Lindernia rotundifolia, Hygrophilia stricta, & Cryptocoryne crispatula		-60% reduction in algal biomass in the planted microcosm -Reduction in nutrients and improvements in water quality		
Macrophytes	–	*Ceratophyllum demersum L.*	2010	-Nutrients, chl-a, and turbidity reduction with varying macrophyte coverage	–	(Dai et al. 2012)
Macrophytes	A: 120 km^2	*Nupar advena*		-Natural habitats contained richer diversity of fish and plants than habitats where macrophytes were planted	–	(Slagle and Allen 2018)
Barley Straw	A: 0.5 km^2 Z_{Max}: 8.2 Z_{Avg}: – Vol: –	Barley dose: 4.5-7g/m^2- 290 bales (2007) and 330 bales (2008)	2007-2008	-50% in cyanobacteria, but increases of 63% -88% for green and browns algae -74% and 61% decline in chl-a and turbidity in the second reservoir	–	(Purcell et al., 2013)
Barley Straw	A: 309 km^2 Z_{Max}: 2 m Z_{Avg}: – Vol: –	Applied for the control of *Aphanizomenon flosaque*	–	-Suppressed growth of *A. flosaque* -Significant decline in DO levels	–	(Haggard et al. 2013)
Barley Straw	A: 0.01 km^2	Applied to control filamentous and planktonic algae	14 weeks	-Decline in phytoplankton -No significant differences in DO and nutrients in treated ponds and control -No algal growth inhibition -Zooplankton community unaffected	–	(Boylan and Morris 2003)

A: Surface Area; $Z_{Max, Avg}$: Maximum, average depth; Vol: Volume; DO: Dissolved Oxygen; –: Not reported

both the treated and control ponds at the end of the study.

Challenges in the field-scale application of sonication can be attributed to complex environmental conditions that influence the cyanobacterial biomass concentration and growth rates in various aquatic systems, including nutrient loading rates, light exposure, water temperatures, and source water hydraulic residence time (Park et al., 2017a). Investigations on how the size and shape of the water body influence the efficiency of cyanobacterial attenuation by sonication are needed. In pilot-scale studies, Park et al. (2019) found that distance from the transmitter was an essential factor as higher growth inhibition was noted near the ultrasonic device, and existing field tests are performed in relatively small water bodies (Nakano et al., 2001; Purcell et al., 2013b).

Additionally, further investigations are needed to establish the methodology for determining optimal ultrasound operational parameters in the field. Schneider et al. (2015) observed varying responses to algal growth in the reservoir by changing the ultrasound programs; however, their operational parameters were not reported. Using arbitrarily selected operating conditions for field applications leads to inconsistencies and uncertainties in ultrasound efficiency as a field solution to cyanobacterial control. Moreover, there is a need for determining optimal conditions with minimal energy costs (Park et al., 2019; Purcell et al., 2013a). Based on the duration and design of the field trials reported above, the treatment longevity of sonication is not established and requires further investigations.

Determination of ideal operational conditions is further essential to minimize algal toxin release during field applications. Depending on the intensity, duration of exposure, and frequency, ultrasound irradiation on toxic blooms can cause the release of intracellular toxins due to cell lysis or the secretion of more toxins as a defense mechanism to sonication (Dehghani, 2016b; Zhang et al., 2006). Generated toxins may be simultaneously degraded with the bloom (Ma et al., 2005; Park et al.,

2019; Rajasekhar et al., 2012b) via chemical mechanisms by the hydroxyl radicals generated during sonication (Song et al., 2005), microbial degradation (Bourne et al., 1996; Krausfeldt et al., 2019), or photodegradation when toxins are present in the photic zone (Kurtz et al., 2021). Thus, it is essential to find optimal conditions that minimize toxin release or degrade existing or generated toxins.

2.1.4.19. Environmental impacts. Ultrasound has been termed an eco-friendly cyanobacterial controlled strategy (Rajasekhar et al., 2012a; Wu et al., 2011), although some evidence suggests potential environmental impacts. In a *Daphnia*-ultrasound treatment, all the adult *Daphnia* used in the experiment died rapidly within 15 min of ultrasound irradiation at low frequencies (<50 kHz) (Lürling and Tolman, 2014). Furthermore, ultrasound-induced cavitation (<1 W/cm^2, and 1 MHz) resulted in damage of fish skin for exposures times as low as 90 s (Renkel et al., 1999). Since ultrasound has been used to control various zooplankton (Holm et al., 2008) and aquatic plants (Wu and Wu, 2006), ultrasound treatments in source water may not be entirely environmentally friendly. Field applications of ultrasound may require high intensities and power to achieve efficiencies reported in small-scale laboratory experiments (Purcell et al., 2013a). The associated ecological impacts are not established yet necessary for determining the suitability of ultrasound as a field-scale solution for cyanobacteria, especially for prolonged (few months to years) exposures that are typical exposure durations for ultrasound treatment (Nakano et al., 2001; Purcell et al., 2013b; Schneider et al., 2015).

3. Biological control

Biological control methods are intended to streamline ecosystem and food web structure to create unfavorable conditions for cyanobacteria

(Fig.1). To this end, filter-feeding organisms such as zooplankton, mussels, bivalves, and fish can be added to feed on cyanobacteria (Boon et al., 1994; Gao et al., 2017); fish that feed on filter-feeding organisms or resuspend sediments can be removed (Ekvall et al., 2014); fish can be added to consume other zooplanktivorous fish; or macrophytes can be planted to promote nutrient uptake and generate allelopathic chemicals that control cyanobacteria (Rojo et al., 2013; Zerrifi et al., 2020; Zhou et al., 2019). Organisms such as viruses, bacteria, and fungi have the potential for cyanobacteria control; however, these techniques have not been scaled up for field applications (Pal et al., 2020; Sharip et al., 2020). Based on the extent of field application, this section has focused on biomanipulation strategies, submerged macrophytes, and barley straws. Mesocosm and field-scale studies for each strategy is summarized in Table 2.

3.1. Biomanipulation

Biomanipulation is the most common biological control method for algal blooms. This method involves increasing the pressure on phytoplankton communities by reducing or removing planktivorous fish (Shapiro, 1990; Shapiro and Wright, 1984) or by increasing grazers and zooplankton populations (Ger et al., 2014; Kâ et al., 2012). By increasing pressure on phytoplankton, the goal is to reduce their populations through increased consumption by other feeders. Removal of zooplanktivorous and benthivorous fish and the addition of piscivores are the most frequently applied biomanipulation methods. The effects of biomanipulation on cyanobacterial control have been extensively reviewed (Triest et al., 2016); hence this section will highlight recent studies and factors to consider during biomanipulation control that can influence successful cyanobacterial management.

3.1.1. Field-scale studies and factors to consider during biomanipulation treatment

The success of biomanipulation strategies is dependent on several factors, e.g., that zooplankton or the selected fish species can consume and assimilate carbon from cyanobacteria efficiently for them to grow and reproduce; their consumption rates match or exceed the overall growth rates of cyanobacteria; and that the consumption rate of cyanobacteria is not influenced by the size, morphology and generated toxins (Boon et al., 1994). When the cyanobacterial growth rates exceed the grazer feed rates and/or target density of the grazers, there are chances of insignificant cyanobacterial control (Ferrão-Filho et al., 2020).

Biomanipulation can be limited by continual nutrient influxes that can lead to the return or maintenance of phytoplankton communities. For the long-term benefits of biomanipulation, it is necessary to maintain reduced external P loading. Benndorf (1987) suggested no more than 0.6–0.8 g TP/m^2 yr loading limit for successful biomanipulation of a lake. When P availability is not controlled, authors have observed unchanged total phytoplankton biomass despite high biomass of the filter feeder *Daphnia*. Furthermore, high nutrient can promote inedible algae and cyanobacteria, thereby diminishing impacts from the biomanipulation (Kasprzak et al., 2007).

3.1.2. Cyanobacterial species-specific response

Some species possess reduced vulnerability to grazers; therefore the effect of grazers in controlling cyanobacteria is specie-dependent. Cyanobacterial cell/filament size, toxicity, and poor nutritional value are inducible or obligate grazing defense mechanisms by cyanobacteria (Martin-Creuzburg and von Elert, 2009; Moustaka-Gouni and Sommer, 2020). For instance, a recent review summarized evidence to suggest that *Microcystis* blooms have a lower mortality rate from grazing effects than other algae (Harke et al., 2016). Yang et al. (2006) reported the defensive transformation of *Microcystis* from unicellular to colonial morphology when subjected to a protozoan grazing to deter further grazing. Other studies have suggested that *Microcystis* may have little

nutritional benefits to grazers and that *Microcystis* generate peptides that inhibit digestive proteases in the grazers, thereby limiting zooplankton grazing (Agrawal et al., 2005; Agrawal and Bagchi, 2001). Other considerations such as the likelihood of zooplankton to adapt to increasing frequencies, durations, and intensities of blooms such that they will coexist with cyanobacteria instead of controlling their dominance (Ger et al., 2014). Findings from Sarnelle (2007) pointed out the potential for cyanobacteria to develop resistance to grazing effect such that a high-density *Daphnia* grazing can reduce cyanobacteria dominance but not eliminate it.

An *in-situ* experiment by Urrutia-Cordero et al. (2016) showed that large-bodied *Daphnia* suppressed the growth of *Aphanizomenon, Dolichospermum, Planktothrix,* and *Microcystis* species. However, *Dolichospermum flosaquae* and *Dolichospermum lemmermanii* have smaller filaments and were more susceptible to grazing than their counterparts *Dolichospermum crissum*. Lack of cyanobacterial control was attributed to species phenotypical characteristics that reduce their vulnerability to grazers (Urrutia-Cordero et al., 2016) and the potential for grazers to acclimate and co-exist with cyanobacteria (Amorim and Moura, 2020). Grazers may fail to feed if cyanobacterial species, especially filamentous species, can surpass the optimal size range for food based on grazer body size (Burns, 1968; Hansen et al., 1994). Severiano et al. (2018) also found that increasing the pressure of zooplankton biomass did not affect filamentous cyanobacteria.

3.1.3. Type and quantify of filter feeders and forage fish

Another factor that can influence biomanipulation attempts is the type of filter feeder employed. In general, *Daphnia* sp. are considered efficient grazers because of their ability to control phytoplankton of wide size range (Ger et al., 2014), but this may not be the case for other grazers. In a 64 ha shallow reservoir dominated by *Planktothrix agardhii* bloom, Kozak and Gołdyn (2004) removed all fish and re-stocked with predatory fish to consume planktivorous and omnivorous fish populations introduced with inlet tributaries. However, the macro-zooplankton, including Cladocerans and Calanoids grazed only on small-sized planktons. Consequently, there was a stimulated growth of large phytoplankton organisms as they benefited from the nutrients released from the small-sized planktons resulting in a failed biomanipulation experiment (Kozak and Gołdyn, 2004).

Initial conditions such as grazer to phytoplankton ratios can influence a biomanipulation performance in the field. Mesocosm studies have indicated that *Daphnia* sp can prevent bloom when present at a high enough abundance before cyanobacteria dominance (Paterson et al., 2002; Sarnelle, 2007). When Ferrão-Filho et al. (2020) assessed the performance of *Daphnia* in controlling *Raphidiopsis raciborskii,* they found that the addition of *Daphnia* at 20 individuals/ml resulted in a significant reduction of biomass. Increasing grazing pressure was efficient in controlling *M. aeruginosa*, but ineffective for filamentous cyanobacteria that discourage grazing due to morphological characteristics (Severiano et al., 2018). Findings from Sarnelle (2007) further pointed out that a high-density *Daphnia* grazing can fail to eliminate cyanobacteria due to resistance to grazing effect.

The type of forage fish added to a lake may further contribute to the biomanipulation results. Mixed findings were reported post-biomanipulation in a 12.6 ha American lake after a two-year biomanipulation effort entailing restocking the lake with piscivores (Shapiro and Wright, 1984). Despite a significant increase in lake transparency and decline in chlorophyll-a after treatment, the lake reverted to pre-biomanipulation conditions due to the re-emergence of the *Aphanizomenon flosaquae* bloom. One of the considerations presented by Shapiro and Wright (1984) was the need for a different type of forage fish instead of the bluegill sunfish that was selected for the biomanipulation. Such considerations are valid and may play an important role on whether a biomanipulation control is successful or not, though only few studies investigating such influences. In a mesocosm experiment, silver carp (*Hypophthalmichthys molitrix*) showed an enhanced

ability to control *Microcystis* because they could more effectively collect and digest gelatin packed *Microcystis* colonies (Šetlíková et al., 2020) than the bighead carp (*Hypophthalmichthys nobilis*) that often prefer to feed on zooplankton than algae (Yi et al., 2016). Therefore, some research should be conducted before biomanipulation control to determine ideal forage fish, filter feeder to add depending on the target bloom and biological interactions at the source.

3.1.4. Environmental conditions and climate factors

Climatic parameters may impact biomanipulation control due to influences in zooplankton and phytoplankton balances and assemblages, among other biological interactions in aquatic systems (Jeppesen et al., 2007). For instance, attaining equilibrium of algal and zooplankton populations can be challenged by rapidly forming algal blooms facilitated by both anthropogenic and climatic factors (Boon et al., 1994). In a warm-temperate lake in Turkey, biomanipulation through fish removal and prior nutrient control resulted in improved water clarity and abundance of submerged plants (Beklioglu et al., 2003). There are few investigations on the efficiency of biomanipulation in subtropical and tropical lakes. Some factors that may influence performance in these regions range from prolonged algal growth seasons, an abundance of small and large fish that feed on zooplankton, lake morphology, temporal variations in precipitation, temperature, and salinity (Amorim et al., 2019; Amorim and Moura, 2020; Jeppesen et al., 2007; Van Colen et al., 2017).

3.1.5. Longevity of biomanipulation

Most studies provide monitoring data on the effects of biomanipulation in the short term (<2 yrs.). A comprehensive data analysis on more than 70 restoration projects conducted in shallow eutrophic Danish lakes reported that the most substantial effects were observed for 4-6 yrs. and the long-term effects ranged between 8–10 yrs. since lakes returned to turbid states after 10 yrs. following initial treatment (Søndergaard et al., 2007). Long term monitoring in a 14.8 km^2 basin of a Swedish Lake pre- and post-biomanipulation entailing the removal of cyprinid fish and reduced predation on zooplankton communities resulted in a 55% increase in the relative abundance of *Daphnia* sp, which led to an overall decline in the toxin-producing cyanobacteria biomass (Ekvall et al., 2014). It is worth noting that this was a repeated treatment effort 10 yrs. following an initial biomanipulation attempt whose effects lasted for about 5 years.

The long-term effects also depend on sufficient control of external P loading with minimal internal P loading (Benndorf, 1987). Annadotter et al. (1999) also report findings from a shallow 1100 ha Swedish lake with significant historical eutrophication and lake management and restoration history. While there was considerably reduced phytoplankton biomass and increased transparency in the lake following a 2-yr. cyprinid fish reduction program, these effects can be due to a joint restoration effort to reduce internal and external nutrient loading. In cases with internal nutrient issues, a combination of biomanipulation initiatives with other nutrient control methods such as nutrient sequestering have yielded improved water quality and aquatic ecosystem health for up to 7 yrs. (Jůza et al., 2019).

3.2. Macrophytes

Macrophytes are essential primary producers that can play a crucial role in maintaining clear water states in lakes and reservoirs. The composition and abundance of submerged macrophytes have thus been implemented as integrators of the dynamic physical and chemical conditions in lakes (Søndergaard et al., 2010).

3.2.1. Mechanisms of cyanobacteria control

Submerged macrophytes can control cyanobacteria through three main processes. Firstly, macrophytes take up nutrients available in the sediments and water column to promote nutrient-limited conditions that

are not conducive to cyanobacteria (Preiner et al., 2020; Robach et al., 1995). Furthermore, macrophytes coverage in lakes prevents resuspension of sediments and nutrients during allochthonous disturbances such as rainfall and wind to promote clear water states (Barbosa et al., 2020). Secondly, macrophytes provide shelter for the zooplankton that are phytoplankton grazers (Søndergaard and Moss, 1998). As such, they indirectly control cyanobacteria by allowing grazers to dominate (Ekvall et al., 2014). The third control mechanism is the secretion of secondary metabolites known as allelochemicals that are inhibitory to phytoplankton (Mohamed, 2017).

Effective implementation of macrophytes for cyanobacterial control in lakes and reservoirs requires adequate macrophyte cover, stability of produced allelochemicals in aquatic systems, and sensitivity of targeted cyanobacteria species to the generated allelochemicals (Hilt and Gross, 2008). The establishment, distribution, and abundance of macrophytes are influenced by site-specific factors such as source water characteristics, solar irradiance, general water chemistry and sediment properties, water velocity, and biotic interactions and competitions (Gasith and Hoyer, 1998).

3.2.2. Effect of source water characteristics

Morphometric features such as depth, size, and bed slopes can indirectly and directly impact photosynthetically available solar irradiance and influence macrophyte development. Jin et al. (2020) planted macrophytes at varying water strata in a pond and found that planting depth significantly influenced biomass height and productivity. The growth of submerged macrophytes at the 5 cm water strata was inhibited due to high light intensity, while those in the 150 cm strata died due to low solar irradiance (Jin et al., 2020). Submerged macrophytes are expected to grow at depths of two or three times the Secchi depth; hence deep sources with reduced water clarity would have limited macrophyte cover (Gasith and Hoyer, 1998). Data on macrophytes, water quality, and lake morphology from 274 lowland lakes in Poland was analyzed by Kolada (2014). The average macrophyte colonization depth was 4–5 m with a maximum depth of 9 m and the total phytolittoral macrophyte coverage was over 70% on average in shallow lakes and was double the coverage in deeper lakes.

The slope of the littoral zone can also influence macrophyte biomass establishment and distribution. A gently sloped littoral allows the deposition of fine materials and modulates the wave action in favor of sediment stability to establish aquatic macrophytes (Duarte and Kalff, 1986; Gasith and Hoyer, 1998). Additionally, sloping in the littoral zone can indirectly affect light availability and influence macrophyte growth (Van Nes, 2002). Kolada (2014) found that shallow and deep lakes with gentle slopes had over 20% more macrophyte coverage than ribbon-shaped lakes with steep slopes.

Fluctuations in water level driven by rainfall, surface runoff, or reservoir drawdown can physically impact macrophytes resulting in their die-off and subsequently influence nutrient cycling in the littoral zone. For instance, rewetting of a reservoir following drawdown can mobilize nutrients in the desiccated sediment in the macrophyte beds and nutrients stored in dead or dormant macrophyte tissues (Lu et al., 2017a). Decomposing macrophyte litter due to water level fluctuations resulted in increased P and N concentration in the water column, although the impact on P levels was greater than N (Lu et al., 2017b). Microcosm decomposition experiments on *Potamogeton crispus* indicated that during the plant senescence and decomposition, there was the leaching of primarily (80%) soluble P from plant tissues into the water column (Wang et al., 2018). This increased water column P concentration can increase source productivity by facilitating phytoplankton growth.

3.2.3. Effect of water quality conditions

Macrophytes and phytoplankton exhibit competitive growth interactions based on existing nutrient, light, and temperature dynamics in aquatic systems. Using a eutrophication model incorporating

phytoplankton, submerged macrophytes, and nutrient dynamics in a lake, Asaeda et al. (2001) demonstrated that high nutrient inflow and the continuous depletion and accumulation of nutrients in aquatic systems promote frequent phytoplankton blooms that in turn creates shading effect on macrophytes. The shading effect from the phytoplankton minimizes available solar irradiance for photosynthesis, thus impacting the overall growth, productivity, and even establishment of macrophytes (Jin et al., 2020; Tan et al., 2019).

Therefore, bloom control using macrophytes may be limited in highly eutrophic systems as phytoplankton blooms induce turbid conditions. Following a three-year evaluation in a eutrophic lake, Jupp and Spence (1977) linked the adverse decline in *Potamogeton filiformis* to seasonal nutrient inputs that yielded blooms of *Anabaena* sp. in the lake, subsequently resulteing in high chlorophyll-a and attenuation of solar irradiance for the macrophytes. Data from 300 Danish lakes were analyzed, and overall macrophyte coverage in shallow lakes (<3 m) decreased with increasing TP, TN, suspended sediments, and chlorophyll-a levels (Søndergaard et al., 2010). A pooled dataset of 782 lakes indicated that the proportion of lakes with adequate (>30%) macrophyte cover decreased with increasing TP concentrations, especially in systems with TP ranging between 0.05-2 mg/L and macrophytes were rare in systems with TN levels above 1-2 mg/L (Kosten et al., 2009). The high nutrient levels promote phytoplankton development, increasing water column turbidity from the resultant high chlorophyll-a levels in the water column. Thus combined N and P management in aquatic systems control is essential to limit the competitive dominance of phytoplankton over macrophytes and increase macrophyte diversity (Moss et al., 2013).

Climatic conditions may also influence water quality conditions in relationship to macrophyte coverage. Regions with warmer climates and high nutrient influxes to aquatic systems are predicted to have lower macrophyte cover (Kosten et al., 2009). Similarly, DO concentrations at the sediment-water interphase are lower in warmer than cooler lakes, implying higher anoxic P release from eutrophic sediment in warm lakes and a corresponding higher phytoplankton biomass in warmer lakes than in cooler lakes with similar external nutrient loadings (Kosten et al., 2011). Thus, more field studies may be needed to expound on the role of macrophytes in the control of algae and how that performance varies between climatic zones with the various confounding environmental factors.

Watershed or catchment-specific characteristics such as catchment area, upstream land-use, and sediment characteristics can also influence macrophyte cover, biomass, and community composition. In an analysis of morphological, water quality characteristics, and macrophyte distribution data from 19 lakes, Schindler's ratio, a measure of a lake's susceptibility to pollutant inputs from the catchment area, expressed as the ratio of catchment area plus lake areas to lake volume presented strong relationships with macrophyte species in the lakes (Stefanidis and Papastergiadou, 2012). In another study of 33 lakes, percent woodland cover in watersheds explained significant variation in both macrophyte abundance and community composition in the lakes, while percent agriculture explained variation in macrophyte cover between lakes (Kissoon et al., 2013). Similarly, when macrophyte cover was assessed in 53 Wisconsin (US) Lakes in relation to land use, agricultural development near the lakes explained the variance in the richness and abundance of macrophyte communities (Sass et al., 2010). Overall, agricultural land uses have been linked to non-point nutrient contributions that establish eutrophic conditions in lakes, while high forested land cover in watersheds promotes clear water conditions that can encourage conditions suitable for macrophyte growth. Land use adjacent to shorelines of lakes have also been noted to be correlated with increased invasive or emergent macrophytes (Alahuhta et al., 2014; Sass et al., 2010); thus their dominance in sources can be indicative of changes in water quality and hydro-morphology in the littoral zone (Alahuhta et al., 2014). Based on these findings, both lake and watershed-scale measures should be considered when lake managers are looking to improve macrophyte cover and community composition in lakes for bloom management.

Water quality characteristics such as pH, alkalinity can also influence macrophyte species and community establishment. Species that can utilize bicarbonate as a source of photosynthetic carbon have a competitive advantage in alkaline waters compared to macrophytes that lack this ability that will prefer less alkaline sources (Alahuhta et al., 2013; Alahuhta and Heino, 2013). Factors such as conductivity and watercolor were also noted to explain macrophyte taxonomic composition and richness (Alahuhta et al., 2013).

3.2.4. Effect of macrophyte cover

Enclosure studies by Dai et al. (2012) found that 20–50% coverage of *Ceratophyllum demersum L.* resulted in significant reductions in nutrient concentrations, turbidity, and chlorophyll-a than the control set up. This study does not indicate the mechanisms through which the macrophytes resulted in the observed water quality improvements, however the reductions in nutrient levels may imply that the underlying mechanism is nutrient uptake. Other researchers have concluded that higher macrophyte densities are better for a stabilized water clarity. Meijer et al. (1999) found that in the majority of clear lakes, low algal biomass coincided with more than 25% of lake surface area macrophyte coverage, and Jeppesen et al. (1990) indicated that more macrophyte coverage in lakes yielded better results, such as higher water transparency. Through *ex-situ* experiments, Mowe et al. (2019) also found that the highest density of macrophytes was more efficient in controlling cyanobacteria abundance apart from two macrophytes *Persicaria barbata* and *Ludwigia adscendens* that were able to control cyanobacteria abundance at low densities. Statistical analyses for a 1-yr period in Bartoan Broad, a shallow 72-ha eutrophic lake in the UK indicated that low macrophyte coverage of 5% did not directly impact the phytoplankton biomass, but provided refuge for Daphnia populations which resulted in population stabilization (Lau and Lane, 2002).

Macrophyte dominance in lakes is sometimes a consequence of other control strategies such as nutrient management efforts and biomanipulation. Thus in some field-scale studies, observed improvements in water quality can be as a result of prior management strategies or due to increased macrophyte coverage, or both, however macrophyte increase has indicated improvements in water quality and ecosystem health. For example, Jeppesen et al. (1990) report that biomanipulation entailing fish removal resulted in a long-term effect of permanent macrophyte populations and although water clarity coincided with macrophyte density in Meijer et al. (1999) study, the results may be associated with biomanipulation effects. In a mesocosm study, macrophyte addition was found to be more effective than biomanipulation (Amorim and Moura, 2020) and combined biomanipulation and macrophyte planting resulted in improved water quality and ecological states and macrophyte cover was correlated with the diameter of zooplankton and density of mollusks (Chen et al., 2020). In a full-scale study where biomanipulation was combined with nutrient sequestration, increased macrophyte cover resulted from successful nutrient control initiatives (Jůza et al., 2019).

3.2.5. Allelopathic interactions

Macrophytes release allelochemicals as a survival and resource-competition mechanism to have the advantage over phytoplankton. The wide range of allelochemicals produced by macrophytes inhibits phytoplankton by negatively impacting photosynthesis, intracellular redox balance, and damaging cellular structure (Legrand et al., 2003; Zhu et al., 2021). Although most studies on allelopathy of macrophytes are conducted in controlled settings with plant extracts or purified plant compounds (Hilt and Gross, 2008; Legrand et al., 2003; Zhu et al., 2021), some studies have demonstrated that allelopathic chemicals can control cyanobacteria in mesocosm setups (Amorim and Moura, 2020; Hilt et al., 2006; Mulderij et al., 2006; Švanys et al., 2014).

Allelopathic effects can be influenced by the differential sensitivity of

phytoplankton to produced allelochemicals and the types of macrophyte and phytoplankton present in a system. A previous review found that planktonic algae were more inhibited by allelopathic chemicals than epiphytic species (Hilt and Gross, 2008). The lack of allelopathic inhibitory effects on epiphytic cyanobacteria growing on macrophytes may be explained by potential adaptation and natural resistance gained by epiphytes against allelochemicals (Mohamed, 2017). Pakdel et al. (2013) investigated the allelopathic effects of *Chara australis* and *Potamogeton crispus* on microalgae. The cyanobacterium *Anabaena variabilis* was more impacted than the green algae *Scenedesmus quadricauda*. In an experiment with over 20 macrophyte extracts on cyanobacteria, inhibitory effects on cyanobacteria were observed in less than 1% of the experimental setups consisting of *Chara globularis, Ceratophyllum submersum, Elodea nuttallii, Hydrilla verticillata, Myriophyllum heterophyllum, Myriophyllum. spicatum*, and *Vallisneria americana* extracts (Maredová et al., 2021). Allelochemicals may further fail to inhibit cyanobacteria in the presence of mixed cultures. Chang et al. (2012) found that interactions of polyphenolic allelochemicals with green algae in mixed green algae and *M. aeruginosa* culture reversed the growth-inhibiting potential of the allelochemicals into growth enhancement effects, thereby increasing *M. aeruginosa* growth rates (Chang et al., 2012). Thus studies on allelopathy for eutrophic systems should incorporate the impact of interactions between species.

Availability of resources may impact the production of allelochemicals and the sensitivity of phytoplankton to macrophyte allelochemicals. Macrophytes produce more allelochemicals when under stress conditions brought by light limitations, nutrient deficits, extreme temperatures, and fluctuations in water levels (Reigosa et al., 1999). Allelopathic polyphenols are known to inhibit extracellular alkaline phosphate (Hilt and Gross, 2008). Since extracellular alkaline phosphate are produced by phytoplankton during periods of nutrient limitation, lower allelopathic effects have been observed on cyanobacteria under P deficiency (Hilt (nee Körner), 2006). Cyanobacteria can also compete for limited resources by producing several secondary metabolites that may have allelopathic impacts on other cyanobacteria species, green algae, diatoms, and angiosperms (Leão et al., 2009). Through a series of experiments, Chia et al. (2018) found that the availability of nutrients greatly influenced the allelopathic interactions between the *Microcystis* and *Anabaena*. Thus, environmental variables such as nutrient availability influence phytoplankton biomass regardless of whether macrophytes are present or absent (Hao et al., 2021).

Aquatic environments may also be subjected to more than one allelopathic macrophyte. Since the efficiencies of growth inhibition allelochemicals vary between macrophytes, whether aquatic systems with mixed macrophyte species are expected to have synergistic algal control should be investigated. Rojo et al. (2013) demonstrated that mixed meadows of macrophytes produced synergistic effects that impacted aquatic communities directly by minimizing algal biomass and indirectly by grazing of filter feeders.

3.2.6. Natural vs. planted macrophytes

The composition and abundance of macrophytes in a lake are influenced by various chemical and physical conditions. Hence, macrophytes have been used as indicators of water quality conditions and in the classification of lakes (Søndergaard et al., 2010; Szpakowska et al., 2021). Additionally, macrophyte abundance can increase as an ecological response to source water management, such as controlling external and internal nutrient inputs and biomanipulation (Jeppesen et al., 1997; Kosten et al., 2009; Meijer et al., 1999). Macrophyte planting was only done in mesocosm studies. Successful planting of macrophytes as a biological control method for cyanobacterial blooms in lakes and reservoirs may be challenging large sources. Whether planted macrophytes survive and become dominant can be vastly impacted by source water quality conditions and morphological characteristics. Slagle and Allen (2018) evaluated whether habitats where spatterdock (*Nuphar advena*) would support richer communities and greater numbers of fish than

unrestored habitats in Lake Apopka, Florida. They found that natural habitats had the greatest relative abundance of fish and more diverse flora, while planted habitats and unrestored habitats generally held lower abundances and diversities. Planting more than one macrophyte species may promote more ecological diversity (Slagle and Allen, 2018).

3.3. Straws

Applying straws such as barley and rice straws in lentic systems is considered an alternative cyanobacterial control strategy. The mode of action of barley straws for cyanobacteria control is not entirely understood and has been a subject of much debate. However, various researchers have indicated that the release of allelopathic compounds during the aerobic decay of straws is an potential mechanism for controlling algae. Findings from a field study that applied $50 g/m^3$ of straws in a reservoir in the UK showed that barley straws yielded about 90% reduction on algal counts compared to previous years with no application (Everall and Lees, 1996). Chemical analysis of water samples upstream and downstream of the straw application site indicated the release of various trace organic compounds known to have acute toxicity to algae (Everall and Lees, 1996). Everall and Lees, (1997) replicated the study by applying 25 g/m^3 in a reservoir and similarly observed a 90% reduction in algal cell counts as early as 12 d post-application. Additionally, toxicologically relevant quantities of phenolic compounds were generated immediately following the straw application and after 3-months post-application (Everall and Lees, 1997). More recently, a series of experiments indicated the release of hydrogen peroxide during photooxidation of lignin and quinone produced from decomposing barley straws (Iredale et al., 2012). Further clarifications on the mode of action of barley straws are deemed necessary.

Barley straws do not provide immediate improvements in water quality. When a potable supply reservoir was treated with 38 g/m^3 to control diatom and cyanobacteria, algal cell counts increased by 27% the following month and only began to decline (~36%) two months after the initial treatment (Barrett et al., 1996). This may be associated with the time required for complete decomposition. Laboratory tests on decomposition have indicated that barley straws' anti-algal characteristics begin after ~2 months of decomposition and are increased steadily to a maximum of six months (Gibson et al., 1990).

While barley may not provide short-term benefits, studies have shown that it offers long-term algal control. In Barrett et al. (1996), monitoring continued for two years with two subsequent applications of 6.5 g/m^3 and the algal counts after the initial application were 75% lower by average. Algal levels were found to remain low for five years after the initial treatment with a twice per year low dose application rates (Barrett et al., 1999).

The decomposition of straws may create an oxygen demand in the water column. Therefore, successful application may require oxygen-rich systems as low oxygen levels can slow or hinder the straws from releasing algal inhibitory substances. For instance, previous studies have applied straws near the water inflow (Barrett et al., 1996; Everall and Lees, 1997, 1996) likely to provide oxygen-rich conditions for decomposition. Low DO levels were proven to inhibit the anti-algal effects of straws applied in ponds, but better algal inhibition was observed in ponds that were equipped with aerators (Boylan and Morris, 2003). Similarly, experiments designed to control *Anabaena flosaquae* demonstrated that while barley suppressed algal growth, DO levels declined significantly to near-zero levels, especially when systems were not aerated (Haggard, et al., 2013). Accordingly, this control strategy may impact and depend on underlying water quality.

The sensitivity of algae to decomposing barley cannot be linked to specific structural or taxonomic algal characteristics. While some studies have indicated selectivity to cyanobacteria, the success for algal control is somewhat anecdotal without a proper rationale for the claimed selectivity. When barley straws were employed to control *Microcystis* blooms in three reservoirs in the UK (Purcell et al., 2013b), there was a

Table 3
Summary and comparisons of mechanical and biological methods for cyanobacterial control.

Strategy	Strengths	Limitations	Considerations	Environmental Impacts
Artificial mixing	-Increases water column DO -Suppress release of reduced species (P, Fe, Mn, NH$_3$) -Cyanobacterial control via sedimentation losses and light limiting conditions -Destratifies water column -Can offer proactive and reactive control -Offers long term benefits (up to 7 yrs.) with continuous operation	-Resuspension of sediment layer nutrients/ions in the water column -Sedimentation of organic matter -Installation and operational cost	-Ineffective in shallow lakes/ reservoirs with a large surface area -May require continuous operation -Can be ineffective when external nutrients are not controlled	-Beneficial ecological impacts -Warming of the water column may impact cold-water habitats
Hypolimnetic oxygenation & aeration	-Raise DO in bottom layers -Suppress release of reduced species (P, Fe, Mn, NH$_3$) -Avoids mixing of nutrients in the water column -Can offer proactive and reactive control -Long term benefits (10-23 yrs.)	-Resuspension of nutrients in the water column -Sedimentation of organic matter -Installation and operational cost	-Success impacted by system optimization based on source geometry and depth -May require a continuous operation or precise timing for effective performance -Can be ineffective when external nutrients are not controlled -Can be influenced by sediment composition of Fe, Mn, and Al	-Beneficial ecological impacts -Warming of the water column may impact cold-water habitats
Sediment dredging	-Removal of nutrient-rich sediments -Increase in DO levels	-Operational cost -Sediment disposal challenges -Suspension of sediment-bound nutrients and metal ions	-Ineffective when external nutrients are not controlled -Not ideal for large lakes/reservoirs	-Disturbance of microbial communities -Suspended sediment-bound materials may be toxic to aquatic life -Disposed of sediments may cause further ecological impacts
Sonication	-No treatment residuals -Potential selectivity to cyanobacteria	-Limited field-scale trials and treatment longevity evaluation -Potential toxin release -Limited process control in the field	-Treatment efficiency can be impacted by source water size /geometry -Field-scale success may require operation at high frequencies and power -Limited field-scale application history and success	-May impact aquatic organisms
Biological control	-Pelagic & benthic bloom control -Proactive & reactive strategy -No metabolite release -No treatment residuals in the water column -Planting of macrophytes can also control internal nutrients -Long term benefits (4-10 yrs.) with biomanipulation -Minimal environmental impacts	-Biomanipulation and straws not suitable for nutrient control -Straws are not effective in sources with low DO -Treatment effect is localized in areas of straw application -Performance of straws is delayed until the onset of decomposition -Limited directives on ideal macrophyte cover and fish removal/ addition rates	-Higher biomanipulation success in shallow than stratified lakes -Macrophytes may be limited in eutrophic sources -Influenced by continued external nutrient inputs	-Long term shift of aquatic ecosystem composition

reduction of 50% in cyanobacteria, but increases of 63% and 88% for green algae and browns, respectively in one reservoir, and approximately 74% and 61% decline in chlorophyll-a and turbidity, respectively in another reservoir. In the former reservoir, the bloom reoccurrence increased chlorophyll-a levels to 180 µg/L (Purcell et al., 2013b). Similarly, *M. aeruginosa* strains were found to be susceptible to barley straws in a lab-scale study, but *Anabaena cylindrica* and *Oscillatoria animalis* were resistant (Martin and Ridge, 1999). Straw extract additions did not suppress *Prymnesium parvum* in lab experiments (Grover et al., 2007) while field-scale studies have indicated the susceptibility of diatom species such as *Asterrionella formosa* and *Tabellaria sp*. (Barrett et al., 1996). Furthermore, barley straws promoted the growth of resistant algal strains (Ferrier et al., 2005; Martin and Ridge, 1999) though the reasons behind the stimulatory effect are not established. Therefore, success in field applications may be limited by the targeted bloom type, among other factors. Given the implications on source water management, further studies may also be necessary to elucidate the variations on susceptibility to barley straw.

Although barley is accepted as an effective control method (Iredale et al., 2012; Purcell et al., 2013b), the selectivity to different algal strains, variations in findings in field results, the overall lag before algal-control, and the varying views on the mode of action for algal control, points toward the need for further studies to understand this method for field-scale algal control. Additionally, no studies have assessed whether barley straws result in the release of intracellular toxins or the impacts on generated algal toxins. Impacts on water quality should also be investigated in depth. For instance, barley straw addition in mesocosm studies resulted in a significant increase in dissolved phosphate levels (Haggard et al., 2013). Consequently, there is an underlying question on the appropriate barley straw dosage to attain algal suppression without water quality impacts and with underlying water quality restrictions for application such as oxygen levels. Typical field-application dosages have ranged from 1–50 g/m^3 (Barrett et al., 1996; Everall and Lees, 1996; Purcell et al., 2013b), though descriptions on dose selection criteria are lacking. Given the lag in algal control using straws, further research on straw application times based on targeted cyanobacteria growth stage can benefit utility managers considering barley as a control strategy.
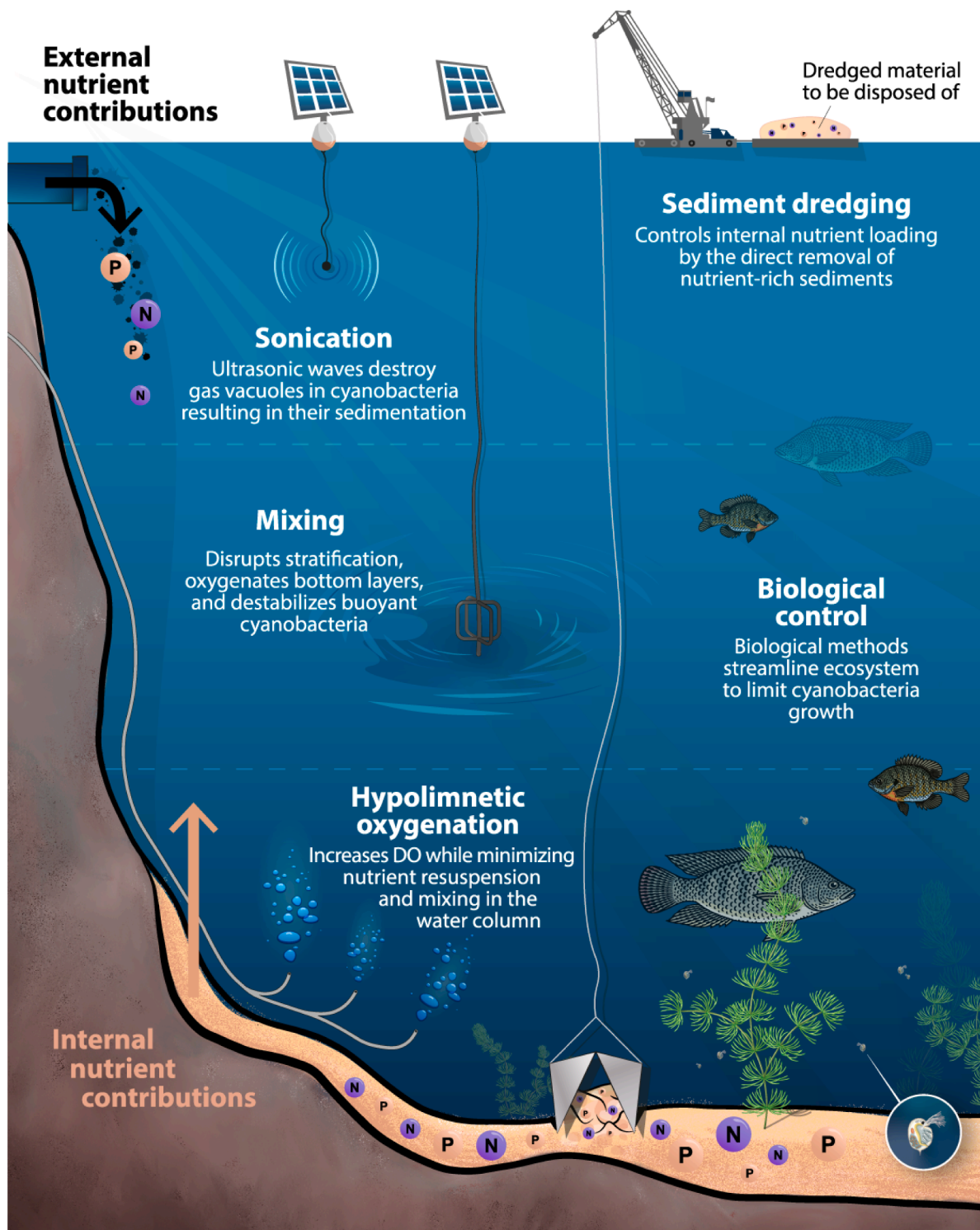
Fig. 1. A conceptual diagram summarizing the mode of action and the effect of sonication, dredging, mixing, hypolimnetic oxygenation, and biological control methods. External nutrients, i.e., nitrogen (N) and phosphorous (P) can originate from discharges from wastewater treatment plants, septic systems, stormwater runoff from agricultural and urban land uses, etc. Depending on water quality conditions, sediment characteristics, and the type of nutrient sequestering chemical used, bound P can be resuspended to the water column.

3.4. *Environmental impacts of biological control methods*

Biological control methodsare considered eco-friendly techniques as there are no toxic chemicals used during treatment and pose limited impacts on the environment. The spatiotemporal scales of ecosystem response following habitat alterations can exceed restoration initiatives and assessment scales (Minns et al., 1996). Therefore biological control methods should also be accompanied by long-term water quality and ecological impact assessment.

Biomanipulation strategies such as fish removal and addition commonly result in habitat alterations in aquatic ecosystems. The reviewed studies commonly reported improvements in water quality conditions and diversity in aquatic organisms, however most studies do not evaluate the ecological response to the biomanipulation strategies.

Biomanipulation can influence changes in competition, predation, and behavioral response to predation and competition in the restored habitat (Pine et al., 2009). Biomanipulation by fish addition can sometimes entail the introduction of non-native fish species. The establishment and dispersal of non-indigenous species can alter trophic interaction and food web structures in a lake and cause long-term effect on native species and even dominance of the invasive species (Burns et al., 2014). Single biomanipulation initiatives focusing on filter-feeding fish, omnivorous fish, and piscivores can impact ecosystem balance by influencing energy flow in the food webs; hence impacts on the food web can be avoided by conducting an integrated biomanipulation based on zooplankton and filter-feeding fish (Peng et al., 2021).

Macrophytes are commonly associated with improved water quality conditions. But their production of allelochemicals may impact other aquatic vegetation. Restoration initiatives involving macrophytes suffer from a lack of long-term evaluation; thus, sustained ecological impacts may not be assessed. Invasive species of macrophytes may alter the ecosystem balance and affect water quality (Sarvala et al., 2020). More studies should evaluate the long-term ecological impacts of planting macrophytes and the influence of invasive species.

The application of barley straws is considered a cost-effective and environmentally friendly control strategy. While field studies did not focus on environmental impacts, most studies did not report negative alterations in the aquatic ecosystem. Other studies have observed improvements in the invertebrate and aquatic plant assemblages since the presence of rotting barley allowed for invertebrate colonization and the consequential absence of algae resulted in the recolonization of higher plants (Caffrey and Monahan, 1999; Welch et al., 1990).

4. Summary and recommendations

In this review, we have provided an overview of mechanical (mixing, hypolimnetic oxygenation, dredging and sonication) and biological (biomanipulation, macrophytes, and straws) methods applicable in the direct and indirect control of cyanobacteria in lakes and reservoirs. While each control strategy offers a wide range of treatment benefits, they are also accompanied by various limitations and ecosystem impacts (Table 3). Nevertheless, the control strategies only provide temporary relief symptoms of water quality degradation such as anoxia, redox-dependent nutrient and contaminant loading, and cyanobacteria biomass. Reduction of external nutrient inputs should still be a primary focus of restoration efforts as treatment benefits were commonly derailed by continued external nutrient inputs. Therefore, an assessment of external versus internal nutrient roles in source water impairment should be conducted before selecting and implementing mechanical/biological control strategies. Internal nutrient loading should be the primary nutrient source to sustain treatment benefits achieved by either mechanical or biological control strategies. Care should be taken when selecting a mechanical or biological control method as restoration should be adequately scaled and optimized as a function of the internal loading rates and source water characteristics.

Mixing and hypolimnetic oxygenation are promising control strategies, although their major drawback is the high energy uses that come with the recommended continuous year-round or cyanobacterial-season operation. The mechanisms of cyanobacterial control using sonication are not established, and field-scale studies offer mixed results. Based on the review, sonication lacks suitable optimization and success in field-scale applications. Biomanipulation methods to reduce eutrophication and stabilize aquatic communities through the removal of planktivorous fish and benthivores and in some cases the selective addition of piscivores. Future studies on biomanipulation should provide detailed information on phytoplankton communities, species of cyanobacteria controlled, selected control methods such as the quantity of fish removed/added, short- and long-term ecological response of the biomanipulation effort, and water quality and nutrient dynamics pre-and post-biomanipulation. Based on the reviewed articles, it is noted that

aquatic macrophytes can support the restoration of eutrophic sources and might have the ability to control cyanobacterial growth through allelopathy and/or depletion of nutrients. Macrophytes are linked to improved water quality and are often abundant in lakes and reservoirs as a result of successful management. More research is needed to clarify the extent of success achieved by planting macrophytes in lakes and reservoirs. Furthermore, there is limited guidance on ideal macrophyte coverage for optimal cyanobacterial control. Future studies on mechanical and biological control methods should be accompanied by adequate monitoring and reporting of site-specific characteristics, water quality parameters, and ecological impacts before, during, and after implementing the control strategy. This delineation of pretreatment conditions, short-term benefits during treatment, and long-term post-treatment response are beneficial towards making accurate comparisons between studies and recommendations for utility managers regarding optimization of various treatment parameters based on water quality and source water geometry.

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