SUPPRESSION OF HARMFUL ALGAE BLOOMS IN RESERVOIRS: LESSONS LEARNED FROM PROJECTS

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ABSTRACT

Suppression of harmful algae blooms (HAB) in drinking water reservoirs is a common and growing concern for utilities worldwide. Effects of HAB include risk of cyanotoxin exposure, formation of taste and odour compounds, and operational impacts to water treatment plants.

The evidence from projects is that suppression of HAB formation in reservoirs is feasible and practical. Evolution of engineering and science over decades has matured into a suite of effective technologies. Successful reservoir management technologies operate through nutrient denial, and (2) buoyancy disruption of cyanobacteria.

Summer is the central season of thermal stratification in reservoirs. Warm surface waters (epilimnion) float on colder, denser bottom waters (hypolimnion). Sediments consume dissolved oxygen, creating anoxia in the hypolimnion. This shift in reservoir geochemistry solubilizes iron, manganese, ammonium, and phosphate from sediments into the water column. It often causes formation of hydrogen sulphide. Increase in phosphate concentrations is the central driver of HAB, but ammonium, iron, and hydrogen sulphide are common co-drivers.

Phytoplankton need abundant light and dissolved nutrients to form HAB. Whatever nutrients are available in the photic (light penetration) zone are quickly scavenged by fast-growing diatoms or green algae, which then sink into the deep. Cyanobacteria control their buoyancy, acquiring nutrients from deep waters and light from the surface on a daily cycle built-in to their physiology.

Destroying anoxia denies cyanobacteria their deep water "pantry" by keeping nutrients locked in sediments. Access to deep nutrients is critically important to cyanobacteria blooms. Diatom blooms often occur in the fall or winter when cooling causes nutrient-rich bottom waters to rise to the surface. Summer anoxia is a major diver of these blooms.

Pure oxygen injection is far more effective at destroying anoxia than is aeration or mixing. By practical application of the law of partial pressures a pure oxygen bubble has about five times greater mass transfer capacity of oxygen to water than does a bubble of air. Comparison of reservoirs that have gone through aeration and oxygen injection phases clearly demonstrates the superiority of pure oxygen.

It is possible to disrupt cyanobacteria buoyancy by destratification. If the reservoir mixes from to top bottom frequently enough, then cyanobacteria cannot sink or float fast enough to maintain their life-strategy advantage in the reservoir ecosystem. Demonstrably effective, theory sets limits to application of this method.

Attention to reservoir geochemistry is critical. Providing abundant dissolved oxygen will not sequester phosphate in sediments if there is not enough ferric iron to bind it. Thus, iron amendments are essential in iron-deficient reservoirs. In shallow reservoirs, increasing the dissolved aluminium concentration may be necessary to bind phosphate

into sediments. Ecologically safe criteria for doing so were established by the US EPA in 2018 after many years of research.

Together, these technologies, all of which are in the public domain, now form core means to suppress HAB formation in drinking water reservoirs and point toward future improvements.

KEYWORDS

Phosphorus, HAB, cyanobacteria, internal nutrient loading, oxygen, anoxia

PRESENTER PROFILE

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1 INTRODUCTION

Suppression of harmful algae blooms (HAB) in drinking water reservoirs is a common and growing concern for drinking water utilities worldwide. Effects of HAB include risk of cyanotoxin exposure, formation of taste and odor compounds, and operational impacts to water treatment plants. Perhaps a better term than HAB would be HPB – harmful phytoplankton blooms. Phytoplankton are a diverse group of organisms that are can be lumped into cyanobacteria (prokaryotes) and true algae (eukaryotes), but the term HAB is well-established and will be used herein.

Most HAB are from cyanobacteria. True algae also form HAB. For example, diatoms blooms, such as *Cyclotella* and *Fragilaria,* can blind filters. This paper focuses on cyanobacteria. Nevertheless, there is overlap of in-basin physical, biogeochemical, and ecological factors that drive HAB in general. Ecologically engineered control of cyanobacteria blooms may therefore benefit management of true algae HAB.

This paper is not a general survey of the means of controlling HAB. Rather, it based on lessons learned from projects using a “bottom up” ecological perspective. Such an approach views a reservoir as a dynamic ecosystem in which internal nutrient dynamics profoundly affect water quality and are subject to engineered controls. Other technical means of managing HAB, whatever their merits, are not within the compass of this paper.

1.1 CYANOBACTERIA ECOLOGY

Bloom-forming cyanobacteria share a common trait fundamental to their physiological ecology: buoyancy regulation (Huisman et al., 2018). Where light penetrates the water column, cyanobacteria fix dissolved inorganic carbon for respiration and cell growth. Carbon beyond immediate cellular needs is stored as carbohydrate granules, causing cells to become negatively buoyant and sink out of the light. Colonies of cyanobacteria can migrate downward or upward for several meters in a few hours. In deeper, nutrient-rich water cyanobacteria take up excess nutrients. Respiration of stored carbohydrate creates carbon dioxide bubbles that are trapped in intracellular vesicles. Cells become positively buoyant, rising into the light. This diurnal cycle of cyanobacterial colonies sinking and rising in the water column is a fundamental driver of bloom formation.

The nutrient dynamics of freshwater cyanobacteria share a nutrient limitation with all other freshwater algae: phosphorus. Perhaps the most common water quality management goal in lakes and reservoirs is therefore to limit phosphorus inputs to the greatest extent possible. It is important to note that diatoms and other true algae readily outcompete cyanobacteria for dissolved inorganic phosphorus (Nicklisch, 1999). Algae grow faster than cyanobacteria up to water temperatures of 25 to 30°C depending on taxa, but at higher temperatures lose this advantage (Lürling et al., 2013, Paerl and Otten, 2013). The principle advantage of cyanobacteria over algae is buoyancy regulation. Non-motile algae sink of the light unless mixed back into it by wind. Motile algae, such as dinoflagellates, can migrate in the water column, but are not a widespread freshwater HAB problem (Paerl et al., 2001).

Cyanobacteria differ from algae in fixing nitrogen and having a much stronger nutrient dependence on iron (Molot et al., 2014). Almost all bloom-forming cyanobacteria fix nitrogen from water in specialized structures called heterocysts (Kumar et al., 2010). Cyanobacteria are thus rarely nitrogen-limited. Microcystis is an important exception, requiring dissolved nitrogen in the water to form blooms. There is an emerging scientific consensus that iron behaves as a macro-nutrient for bloom-forming cyanobacteria (Molot et al., 2014), but not for algae because of strong differences between prokaryotic and eukaryotic cellular physiology. The availability of dissolved or colloidal iron in water can be co-limiting with phosphorus in cyanobacteria blooms.

1.2 INTERNAL NUTRIENT LOADING

The fundamental driver of internal nutrient loading is anoxia at the sediment surface. Typically, anoxia is caused by thermal density stratification of the water column in warm weather. A warm, well-mixed surface layer (epilimnion) floats over a cooler, denser bottom later (hypolimnion). Bacteria at the sediment surface consume dissolved oxygen (DO). Cut off from the atmosphere, there is no replenishment of DO in the hypolimnion. Depletion of DO may occur shortly after thermal stratification sets up or not until early fall depending on basin-specific factors. Stimulation of HAB by hypolimnetic anoxia may occur while thermal stratification exists or after fall/winter turnover of the water column enriches surface water with nutrients.

Regardless of timing, anoxia becomes a key factor in the nutrient budget of basins. At a large scale, the Baltic Sea is the world’s largest estuary and has the largest (anoxic) dead zone (Diaz and Rosenberg, 2008). Despite a 50% reduction of external nutrient loads, the anoxic internal loading drives cyanobacteria blooms and is now 70% of the basin’s phosphorus budget (Stigebrandt, 2018). At a small scale, thermal stratification and anoxia in shallow stormwater ponds drives the summer nutrient budget and causes blooms (Palmer-Felgate et al., 2011). Occupying a place between these extremes of scale, drinking water reservoirs are subject to similar dynamics.

Reservoirs are a sink for phosphorus inflows from the watershed. Phytoplankton scavenge phosphate from the water, converting it to organic phosphorus. Ultimately, phosphorus either flows out of a reservoir or ends up in the sediments. Decay converts organic phosphorus to phosphate. At any time during internal cycling of phosphorus, ferric iron will bind with phosphate. Ferric iron is insoluble and will ultimately precipitate to the sediment surface, sequestering iron-bound phosphate where it is unavailable to phytoplankton if the sediment surface is aerobic.

Anoxic sediments release phosphate back into the water column by reducing insoluble ferric iron (Fe^{3+}) to soluble ferrous iron (Fe^{2+}). Iron-reducing bacteria drive this process. But there is also abiotic reduction of ferric iron by sulphide (HS^-). A few millimetres below the sediment surface, sulphate-reducing bacteria produce sulphide. Although the free-energy yield of iron reduction is far more than sulphate reduction, sulphate-reducing bacteria have a system-dynamic advantage. They produce sulphide that reduces ferric iron and sequesters it in insoluble ferrous sulphide (FeS) and pyrite (FeS_2). Sulphate-reducing bacteria essentially “rob” iron-reducing bacteria of their energetic advantage by consuming ferric iron electron acceptors. A feedback loop quickly sets up where sulphide production quenches competition for organic carbon electron donors that both sulphate-reducing and iron-reducing bacteria need (Austin et al., 2016).

This same process also provides sulphate-reducers with a strong competitive advantage over manganese-reducing bacteria. Reduction of manganese and iron typically occur simultaneously from anoxia hypolimnia in proportion to sediment iron and manganese content. Although manganese is not a driving nutrient for HAB, manganese enrichment of the hypolimnion is clear sign of an overall biogeochemical dynamic that favours HAB.

These biogeochemical dynamics enrich the hypolimnion with phosphorus, iron, and ammonium. In high-sulphur waters, sulphide may scavenge enough iron to counteract iron enrichment. Ammonium is driven from sediments by ion-exchange. Ammonium (NH_4^+) is loosely bound to organic matter in sediments. Anoxic efflux of ferrous iron (Fe^{3+}) and manganous manganese (Mn^{2+}) into sediments displaces NH_4^+ from organic absorption sites, causing it to diffuse into the water column.

The impacts of anoxia described above operate in a seasonal timescales. These are driving mechanisms of HAB after the onset of density stratification. There are longer time scales that create this dynamic. Over a scale of years to decades a reservoir that once enjoyed good water quality and rare HAB can evolve to something much worse by aging of sediments (Katsev et al., 2006). Termed diagenesis in the limnology literature, it a process in which sediments accumulate organic material and reduced minerals, which increases the sediment oxygen demand. A hypolimnion that once stayed aerobic from spring to fall, accumulates days of anoxia, then weeks, and then months. At some point, there is a "regime shift" (Scheffer, 1998); that is, basin dynamics evolve from not favouring HAB to favouring HAB. Regime shift is often abrupt because of the asymptotic nature of positive feedbacks, appearing as a rude surprise to reservoir managers if the warning signs anoxic biogeochemistry are not recognized.

Sediment diagenesis is a product of years-long feedbacks (Katsev et al., 2007, Katsev et al., 2006). Algae sink to the bottom and decay, creating an oxygen demand that that is predicted by spring chlorophyll-*a* concentrations (Walker, 1985). Over years, internal nutrient loading, however slight, can rachet-up spring phytoplankton growth, which then ratchets up sediment oxygen demand (SOD). The regime shift tends to occur soon after SOD reaches a critical point which causes prolonged anoxia in the hypolimnion (Scheffer and van Nes, 2004, Wang et al., 2012).

Sediment diagenesis creates dead zones. Although "dead zone" is a term typically used for anoxic regions in seas and oceans, this term is apt for reservoirs and lakes. Few, if any, reservoirs have been built where water quality projections have considered progression of sediment diagenesis and creation of dead zones. Most reservoirs have been constructed in the last 100 years. As reservoir sediments age, seasonal dead zones have become ubiquitous worldwide and are major drivers of HAB.

1.3 HAB CONTROL THEORY

In reservoirs, cyanobacteria need access to internal nutrient loads to build bloom biomass. Access to those nutrients entails buoyancy regulation. Thus, cutting off access to internal nutrients or disrupting buoyancy regulation curtails cyanobacteria HAB formation potential. Quenching of internal nutrient loading will also curtail other bloom types, such as fall algae blooms that occur at reservoir turnover when deep, nutrient-enriched water mixes with surface water.

Quenching internal nutrient loading to improve reservoir water quality is not a new idea (Cooke et al., 2005), but the technical means of doing so have evolved, as has the science of HAB dynamics. From a management perspective, anoxia should not be tolerated anywhere within a reservoir at any time. Technically, pure oxygen is substantially more effective than aeration for ensuring that the sediment surface is aerobic as discussed below. Pure oxygen technology in reservoirs is widely applied in the United States with 38 linear diffuser (Mobley Engineering Inc., personal communication) and 7 super-oxygenation (Eco2 Technologies Inc, personal communication) systems installed since the early 1990s. Pure oxygen, however, is rare outside the US.

Oxygen alone may not be enough to prevent a substantial internal nutrient load (Hupfer and Lewandowski, 2008). Because ferric iron is needed to bind phosphate, simultaneous injection of oxygen and ferric iron may be necessary and has proven effective at quenching internal nutrient loading (Walker et al., 1989, Austin et al., 2019). Addition of iron may seem paradoxical to controlling cyanobacteria HAB. Afterall, the evidence for iron as a macronutrient for cyanobacteria is clear. However, as a long as strongly oxidizing conditions are forced in the hypolimnion, iron will scavenge phosphate without becoming available as a nutrient to cyanobacteria.

The theory of cyanobacteria buoyancy disruption was developed by Huisman et al (2004). In simple terms, blooms cannot form when cyanobacteria are mixed vertically at rates greater than rising or sinking velocities. Destratification aeration can do this. However, efficacy depends on depth. In shallow reservoirs that intermittently stratify, destratification aeration will not control HAB by buoyancy disruption because cyanobacteria will mix back into light often enough to maintain biomass growth. Nevertheless, under technically appropriate conditions described by Huisman et al buoyancy disruption can stop or prevent cyanobacteria blooms.

Geochemical augmentation is an emerging technology to control HAB in shallow reservoirs. The basic concept is to continually dose the reservoir with aluminium salts at ultra-low, soluble concentrations. Aluminium concentrations are kept below chronic toxicity criteria recently promulgated by the US EPA (2018). Aluminium hydroxide complexes in solution scavenge phosphate from water and eventually settle out with organic particles. Geochemically, this method is much like ferric iron injection in deep reservoirs. The key differences are that (1) aluminium sequestration of phosphate is insensitive to anoxia and (2) that managing concerns of chronic toxicity threshold entail close attention to water chemistry.

2 PROJECTS, TECHNOLOGIES, AND RESULTS

Four project case studies will demonstrate the concepts previously discussed to control HAB: (1) Nutrient denial via deep oxygen injection plus ferric to quench internal loading; Buoyancy disruption by destratification aeration; (3) Geochemical augmentation with aluminum salts. Two of the projects provide rigorous comparison of aeration versus oxygenation.

2.1 NUTRIENT DENIAL

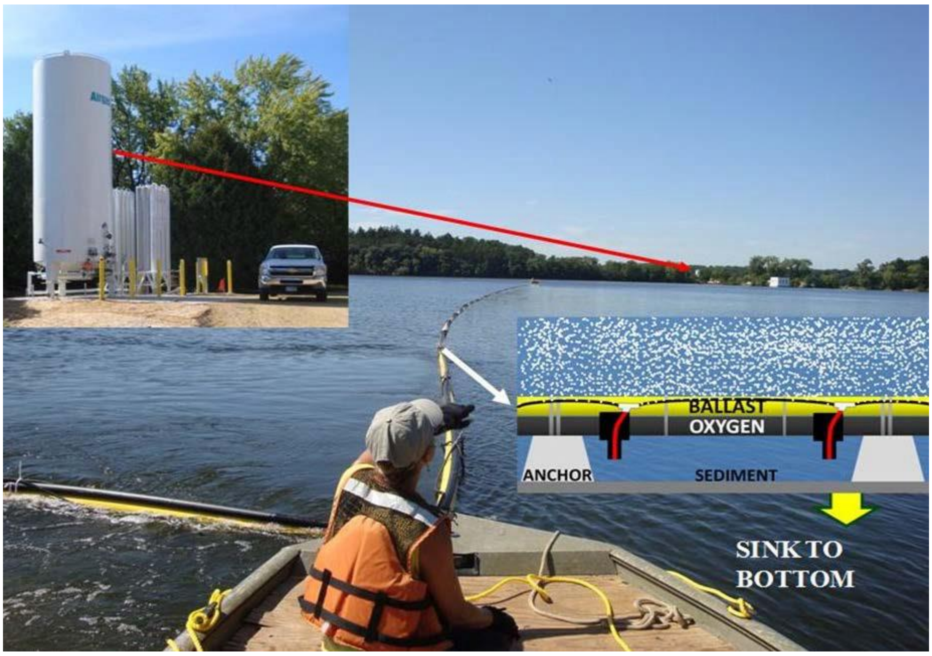
Saint Paul Regional Water Services in Minnesota USA pumps 170 MLD (45 MGD) from the Mississippi River through a chain of four lakes before treatment. Two lakes, Pleasant and Vadnais have stable thermal stratification in the summer (depth 16 m). Each was equipped with partial-lift hypolimnetic aeration (HA) systems (Walker et al. 1989) which were later replaced by hypolimnetic oxygenation (HO) using pure O_2 sparged in linear diffusers (Austin et al., 2019). Ferric chloride has been injected at the

Mississippi pump station, directly into the terminal lake (Vadnais), and into a local tributary stream that flows to Vadnais. The nominal ferric iron concentration in the pipeline or in plumes is 0.5 mg/L.

HA did not relieve hypolimnetic hypoxia in Pleasant (median DO 1.7 mg/L) or Vadnais (median DO 2.7 mg/L). In contrast, median HO values were aerobic (Pleasant 4.2 mg/L, Vadnais 8.5 mg/L).

Aeration substantially lowered hypolimnetic total Mn and Fe concentration in Pleasant (Figure 1) and Vadnais (Austin et al. 2019). Oxygenation lowered hypolimnetic total Mn and Fe concentrations yet more, Mn by 75% in Pleasant and 93% in Vadnais, and Fe by 84% in Pleasant and 81% in Vadnais. Substantially lower Mn and Fe concentrations cannot be attributed to increased oxygen flux rates with the oxygenation system. The HA systems were rated at 1200 kg O_2/d, mean oxygen delivery was 1775 kg/d in Pleasant and 970 kg/d in Vadnais. Superiority of HO performance can be attributed to much lower induced currents in HO compared to HA systems.

Hypolimnetic TP concentrations were inversely proportional to total iron concentrations (Figure 2) (Austin et al. 2019). HA substantially lowered median hypolimnetic TP from baseline, 0.552 mg/L to 0.146 mg/L in Pleasant, and 0.533 mg/L to 0.045 mg/L in Vadnais. HO median hypolimnetic TP was significantly lower at 0.053 mg/L in Pleasant, 0.030 mg/L in Vadnais. Lower hypolimnetic TP corresponded to lower surface TP



(Figure 3) (Austin et al. 2019).

Figure 1. Montage of pure oxygen linear diffusers. Diffusers are manufactured on site from butt-welded HDPE pipe, soaker hose, fittings, and cast concrete weights. The ballast pipe is charged with air or O_2 gas, as convenient, allowing the diffuser to be floated over the deepest water. Flooding of the ballast pipe places diffusers on the bottom. Gas supply is liquid oxygen (LOx) on shore. Vaporization pressure of LOx drives diffuser

gas flux.

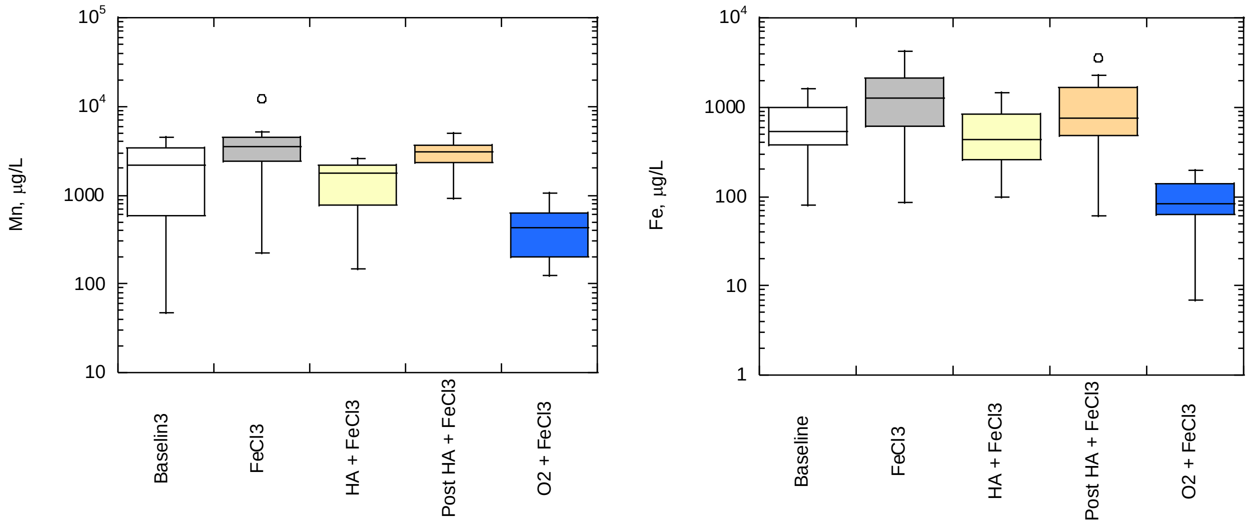


Figure 2. Hypolimnetic Mn and Fe by project phase. Baseline 1984-1986, $FeCl_3$ injection at Mississippi River pump station 1988-2018, hypolimnetic aeration (HA) 1995-2006, HA defunct 2007-2013, hypolimnetic oxygenation 2014-2018.

Chlorophyll-*a* from surface samples was reduced in successively greater increments by injection of $FeCl_3$, HA, and HO in Pleasant which receives water from the Mississippi after residence time of a few hours in a small upstream lake (Figure 4). In Pleasant, there is both a reduction in median surface Chl-*a* and variability. In Vadnais, which is the terminal reservoir and from which raw water supplies are withdrawn, the effect of HO was to reduce variability of Chl-*a*, but there was no significant difference in median values between HA and HO.

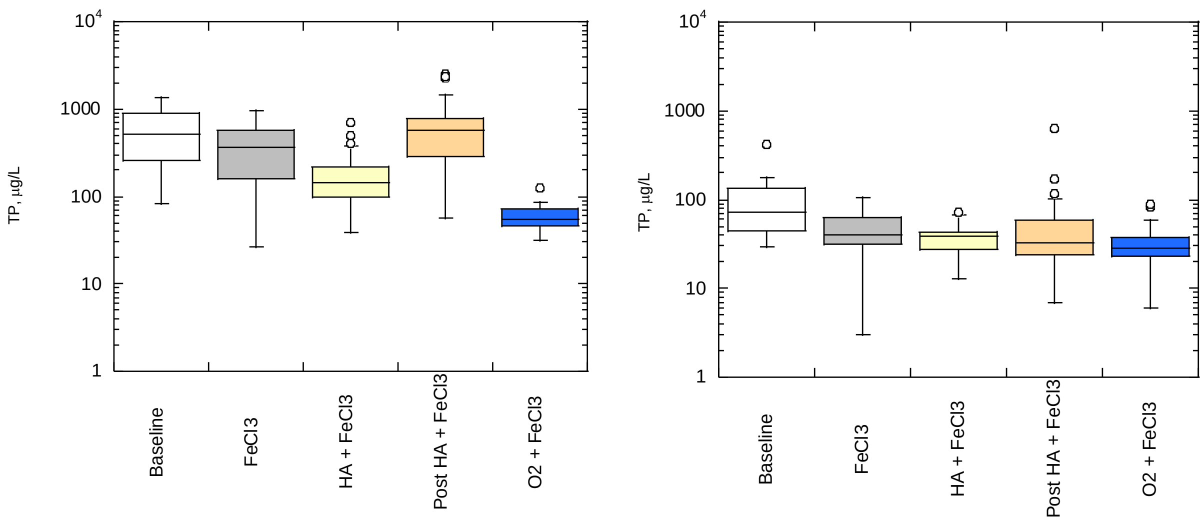


Figure 3. Hypolimnetic (left) and surface (right) TP. See Figure 1 for project phases.

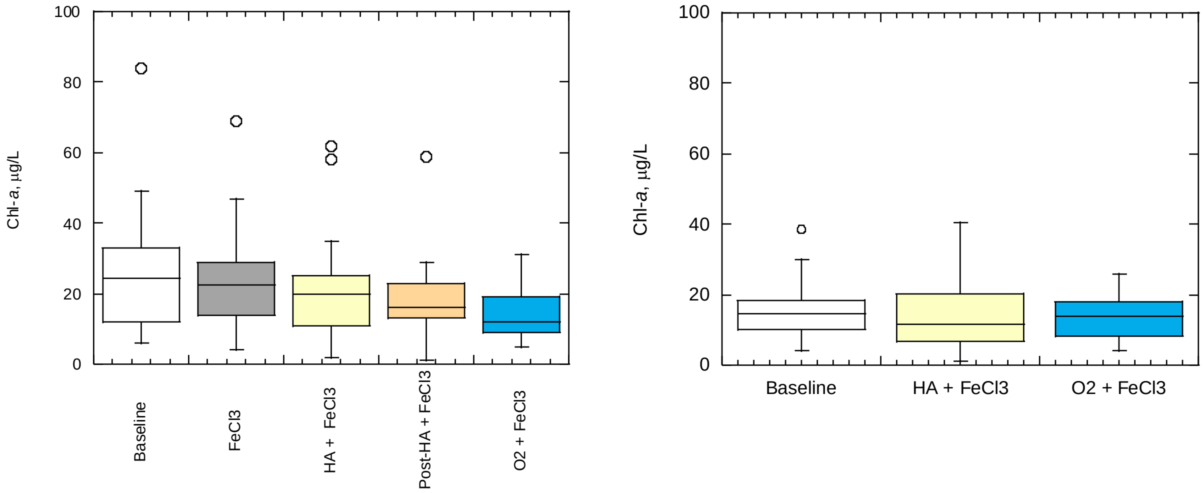


Figure 4. Chlorophyll-a in Pleasant (left) and Vadnais (right). See Figure 1 for Pleasant project phases. Vadnais project phases: baseline 1984-86, HA + $FeCl_3$ 1987-2010, O_2 + $FeCl_3$ 2012-2018.

2.2 BUOYANCY DISRUPTION

C.W. Bill Young Regional Reservoir is in Lithia, Florida which is east of Tampa. It is a sidestream reservoir storing water pumped from the Tampa Bypass Canal, Hillsborough and Alafia rivers. At full stage the storage capacity of 63,000,000 m^3, water surface area of 450 ha, and a maximum depth of 30 m. After reconstruction, the reservoir refilled in 2015. It is equipped with a hypolimnetic aeration (HA) system and a destratification aeration system. The former is used at full stage water elevation and the later when the reservoir about half full. Only one system can operate at a time.

Per the hydrodynamic controls on cyanobacteria blooms proposed by Huisman et al (2004), the destratification aeration system was also intended to operate should an incipient cyanobacteria bloom be identified. To this end an EXO2 (Xylem) continuous vertical profiler was installed October 2015. Every two hours the profiler travels through the water column. At one-meter intervals it monitors for pH, temperature, dissolved oxygen, conductivity, turbidity, chlorophyll-*a*, fluorescence dissolved organic matter, and phycocyanin. The latter is a pigment unique to cyanobacteria.

Thermal stratification of the reservoir began in March 2016 while the HA system was operating (Figure 5). The HA system could not prevent onset of hypoxic conditions by early May. An exponential increase in cyanobacteria pigments was observed to begin in late April 2016 (Figure 6), triggering a switch from the HA system to destratification on May 6th (Figure 5). Compete destratification to the diffuser elevation took approximately

one week but was at least 80% complete in three days. Destruction of the incipient cyanobacteria bloom followed destratification closely with accumulation of cyanobacteria trapped at the reservoir bottom observed at the end of the first week of destratification (Figure 6).

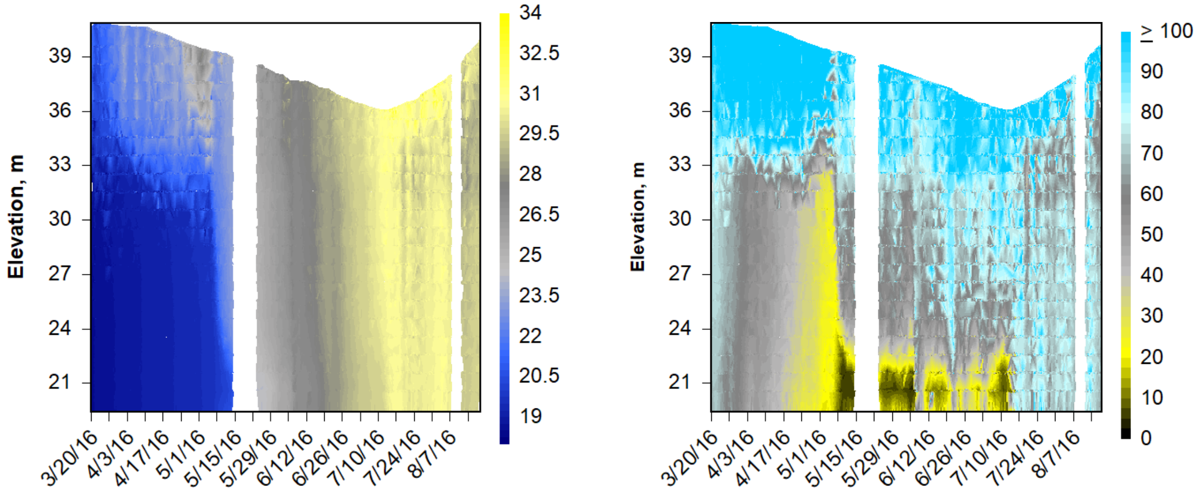


Figure 5. Temperature (left) and percent DO saturation (right) isopleths for C.W. Bill Young Reservoir. Vertical white columns are periods of no data.

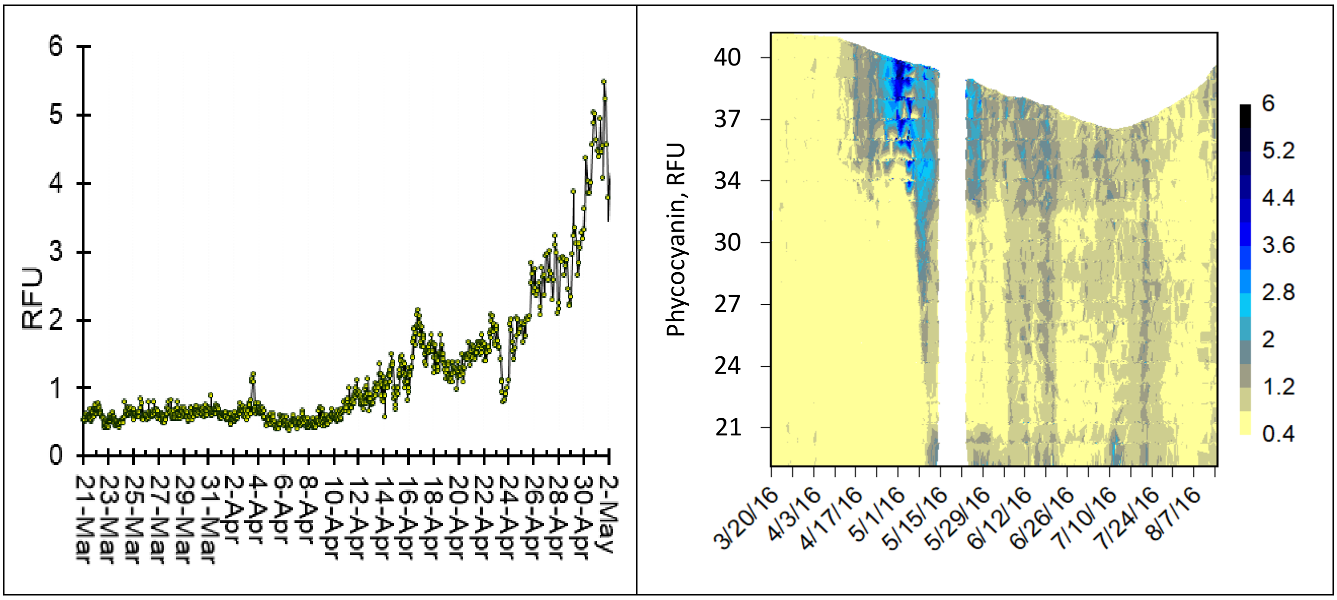


Figure 6. Phycocyanin surface values (left) and isopleth (right). Units are reference formazan unit, which is a calibration standard requiring calibration to biological units (e.g. cells/ml). Thus, units are a relative measure of phycocyanin. Note that the aeration diffuser is located approximately at elevation 20 m (above mean sea level).

2.3 GEOCHEMICAL AUGMENTATION

Phosphorus is almost always the principle limiting nutrient to algae freshwater (Wetzel, 2001). Fate of phosphorus, whether internal or external is ultimately a question of geochemistry or biogeochemistry. Either phosphorus binds irreversibly to minerals in the sediments or it is exported to outflows. From engineering perspective, forcing irreversible binding of phosphorus to iron, aluminium or lanthanum greatly simplifies deeply complex dynamics of fate and transport of phosphorus within reservoirs.

As seen previously, ferric iron is effective as quenching internal phosphorus loading. Without strong thermal stratification and strong controls over DO at the sediment surface, however, iron is unlikely to be effective. It may even stimulate increased algae/cyanobacteria growth in iron-deficient water.

Engineering practice removes phosphorus from water via coagulation clarifiers. However, given that the stoichiometric mass ratio of aluminium to phosphate-P is 0.87:1 the concentration of aluminium to remove phosphate-P from reservoir would be very low compared to the aluminium dose needed to coagulate water as argued below.

Reservoir water of 100 µg/L total phosphorus (TP) is highly eutrophic, of poor quality as a raw water source. Lowering reservoir TP to near or less than 40 µg/L would drastically reduce potential for HAB. The nominal Al concentration to lower TP from 100 µg/L to 40 µg/L is only 52 µg/L. By comparison, an alum dose of approximately 40 mg/L would be reasonable conceptual estimate of dose required for flocculation prior to calculations or jar testing. That dose corresponds to an Al concentration of about 3.6 mg/L at 9% Al by mass in $Al(SO_4)_3$. This comparison is simplistic because there are competing reactions for Al in water, especially form dissolved organic carbon (DOC). However, an Al dose of 520 µg/L, which ten times the nominal dose in this example, the Al concentration would be still be soluble.

Flocculation is needed in a clarifier, but not in a reservoir. Even a small water supply reservoir will have a hydraulic residence time (HRT) of weeks. Clarifiers typically have an HRT on the order of an hour, often less. A reservoir is a large geochemical reactor that allows slow reactions and slow sedimentation rates to scavenge phosphate from water provided the geochemistry favours those reactions. Common practice with aluminium salts in lakes and reservoirs is to use flocculating doses (Cooke et al. 2005) but use of soluble doses strategies will render some of this practice obsolete.

Geochemical augmentation is defined here as soluble doses of phosphate binding minerals at concentrations less than chronic toxicity. With iron, the task is simple as the total iron concentration toxicity threshold is 1.0 mg/L in the USA (US EPA, 1988). Chronic toxicity thresholds for lanthanum are currently in development, but the evidence suggests a continuous exposure concentration of 4 µg/L (Herrmann et al., 2016). The US EPA (2018) has promulgated an aluminium toxicity model in which pH, hardness, and DOC set the continuous concentration criterion (CCC). For example, at a pH of 7.5, hardness of 70 mg/L (as $CaCO_3$), DOC of 3 mg/L the CCC is 1300 µg/L. The model is sensitive to pH. For the same values, but pH of 6.5 the CCC is 440 µg/L. Aluminium is clearly more suited to continuous dosing to scavenge phosphate from water than is La. Because Al retains its bond with phosphate under anoxic, circumneutral conditions it may be more widely applicable than ferric iron, which is limited to projects with rigorous controls on sediment anoxia.

Geochemical augmentation is an emerging technology using of aluminium salts, but in an early stage of development. Osgood (2012) demonstrated lowering of stormwater pond TP from 170 µg/L into the 40 to 60 µg/L range using a 10:1 Al:TP mass dosing strategy. The project began in 1997 and continues to achieve this performance as of this writing (Osgood personal communication). Moore et al (Moore et al., 2009) report decreasing TP concentrations with alum addition dispersed in soluble doses in Newman Lake, Washington State, USA. Osgood's study was repeated in a Kansas, USA lake fed by stormwater (Austin et al., 2017) and in Lake Bowen, Spartanburg County, South Carolina, USA (Jacobs unpublished data) with results from both these projects summarized below.

Veteran's Lake is in Great Bend, Kansas. It has an area of 5.6 ha and a maximum depth of 2.4 m. It is a former sand quarry fed by urban stormwater and is a sump lake that discharges through groundwater seepage. A destratification aeration system installed by the City prior to 2010 prevents thermal stratification, but in no way succeeded in controlling HAB. Veteran's Lake was on the Kansas Department of Environmental Health cyanotoxin warning list from 2010-2015. In August 2014, there was a massive summer fish kill, apparently from cyanotoxins because the destratification system was operating.

A monitoring program began August 2014 using biweekly grab samples for TP and manual vertical profiles using a multi-probe: DO, pH, temperature, conductivity, phycocyanin, Chl-*a*, oxidation reduction potential.

Alum was metered into diffuser bubble plumes from 1.0 m^3 capacity bulk containers at the lake shore (Figure 7). The initial dosing period was late April to early July 2015. Over 81 days 40 m^3 of liquid alum solution (48.5% alum) was metered into Veteran's Lake (89,000 m^3). Another 20 m^3 was metered in April-May 2016.

Prior to dosing in 2015, the lake TP concentration was 210 µg/L (Figure 8). Over the 2015 dosing period the average TP concentration decreased by 1.85 µg/L/d, reaching 60 µg/L (Figure 8). The cyanobacteria bloom collapsed during the alum dosing period. In August 2015, the lake was removed from the state cyanobacteria warning list for the first time in five years. Dosing in 2016 maintained low TP and low phycocyanin concentrations. Failure to dose thereafter allowed return of the status quo ante water quality conditions.

There is no aluminium data for this study. Assuming a DOC of 5 mg/L, a known hardness of 160 mg/L, and an average lake pH of approximately 8.5 the CCC would be 1600 µg/L. The nominal daily Al dose was approximately 300 µg/L. It is unlikely that CCC was exceeded during the dosing periods.

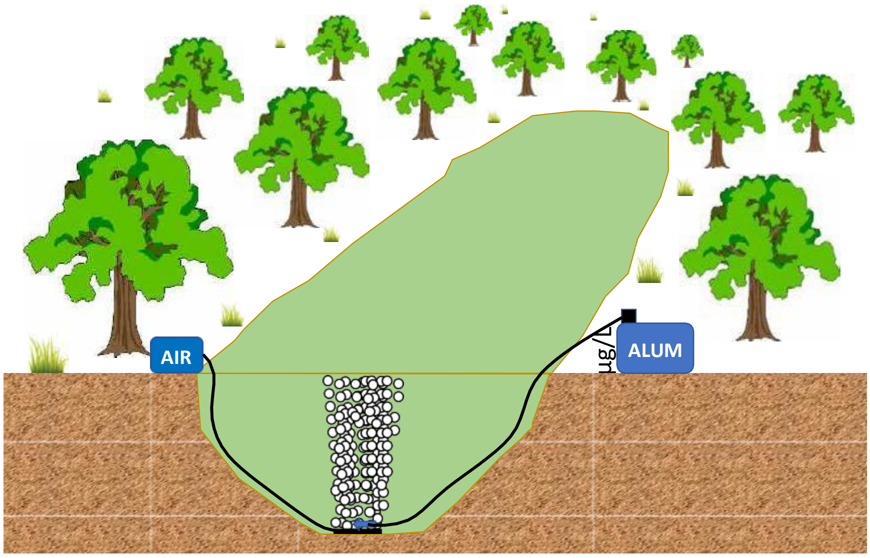


Figure 7. Schematic of Veteran's Lake alum dosing.

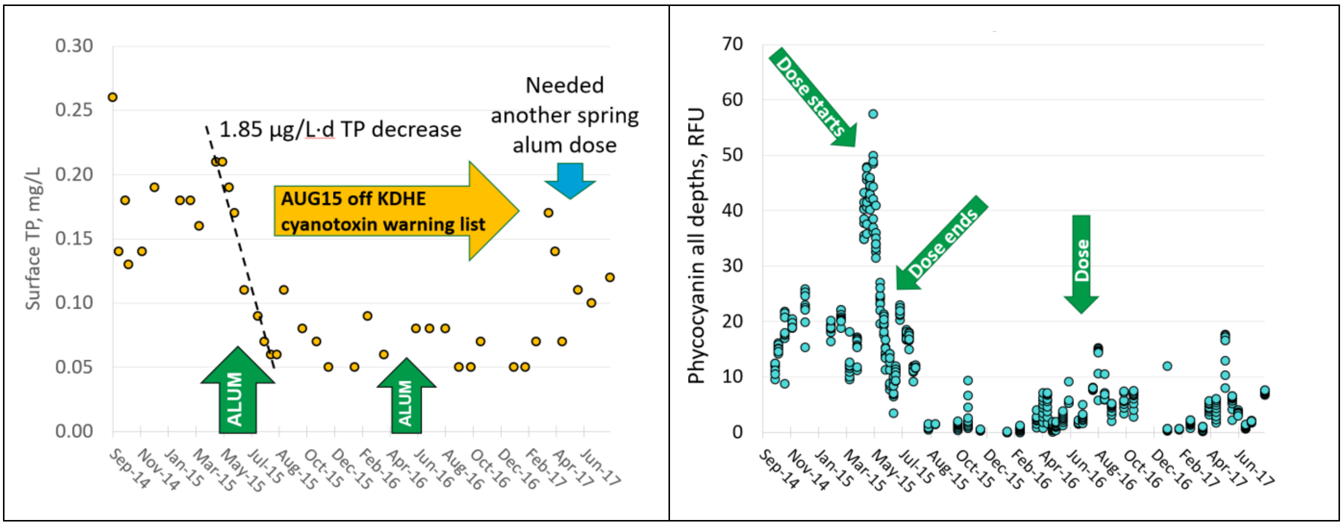


Figure 8. Veteran's Lake total phosphorus (left) and phycocyanin (right). Alum doses were April-July 2015 and April-May 2016.

Lake Bowen is owned and operated by Spartanburg Water, Spartanburg County, South Carolina USA. It has an area of 621 ha. The maximum depth is 12.5 m and average depth 4.5 m. Most of the lake is not stably stratified. Shallow water cyanobacteria blooms have developed, possibly from cyanobacteria recruitment from the sediment surface and nutrient release from intermittently anoxic surficial sediments.

Deteriorating water quality, including sustained cyanobacteria blooms, required remediation of Lake Bowen. A linear diffuser hypolimnetic oxygenation system with ferric chloride injection was installed in the hypolimnion in the region near the dam front in in 2016 (Figure 9). An aluminium chlorohydrate (ACH) injection system was installed in three locations in Lake Bowen to scavenge phosphate from surface waters. Over half of the lake is shallow water and the lake is heavily loaded with phosphorus from the watershed. Consequently, ACH geochemical augmentation system was needed to attenuate the externa phosphorus load.

The ACH system began dosing in January 2017. There was an immediate reduction in surface water total phosphorus, lowering mean TP from 32 µg/L to 24 µg/L (Figure 10). Of greater importance than the mean TP, however, is the rate of exceedance 40 µg/L or greater, which is the threshold for eutrophic conditions, from 4.5 days per month to 0.5 days per month. Chronic toxicity (continuous concentration criterion – CCC) is about 1500 µg/L based on pH, DOC (estimated), and hardness. Per US EPA criterion, the CCC is based on a four-day rolling average of total aluminium concentration which was not exceeded (Figure 11).

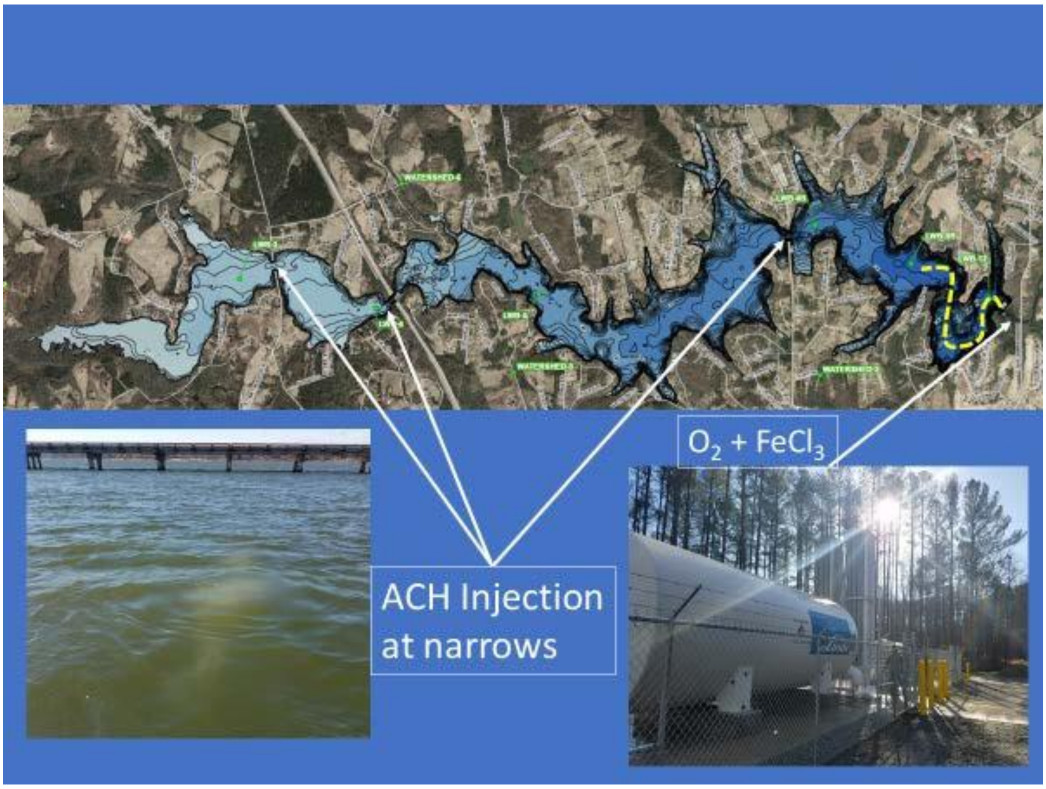


Figure 9. Lake Bowen. Aluminium chlorohydrate injection (ACH) is at narrows into bubble plumes (lower left). Dosing stations are permanent infrastructure. Pure oxygen and ferric chloride is injected into the hypolimnion in the region of the dam front. The yellow dashed lined is the approximate location of the linear diffuser. The LOx tank and ferric chloride dosing station is just downstream of the dam.

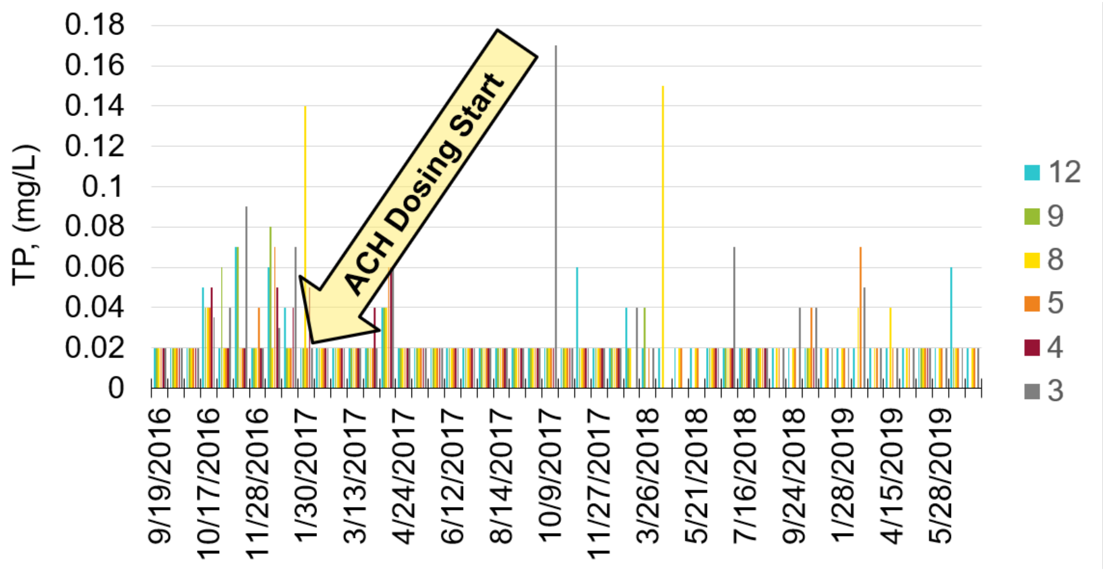


Figure 10. Lake Bowen TP. Colours correspond to sampling stations. Lowest numbers are far upstream in Lake Bowen, Station 12 is at the dam front. The TP method detection limit (MDL) was 0.04 mg/L. Values flagged as below the MDL were taken as 0.02 mg/L.

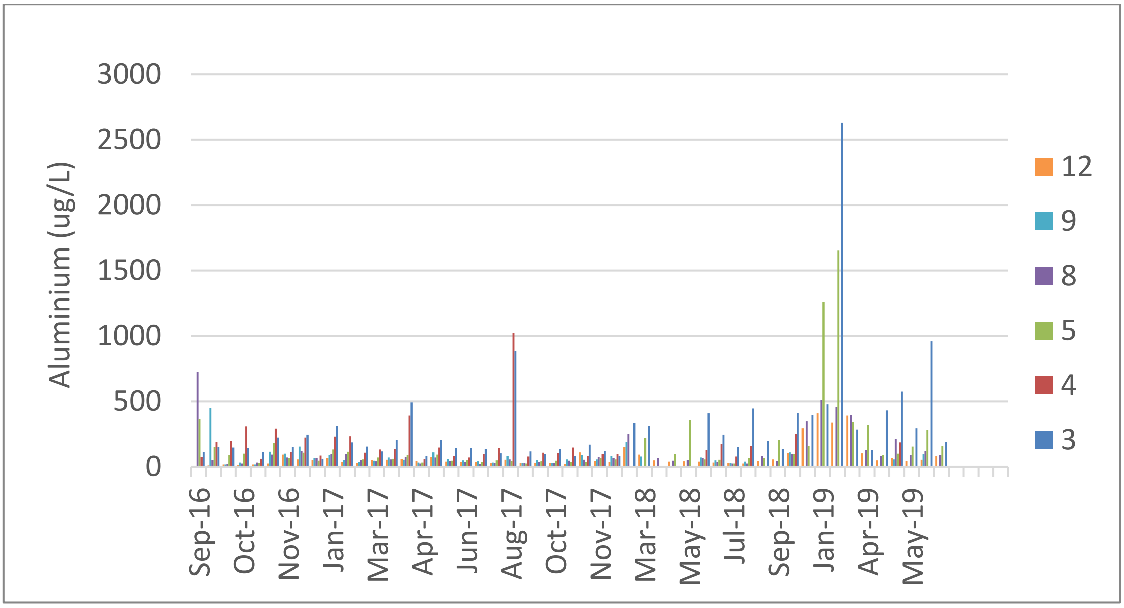


Figure 11. Lake Bowen Aluminium. Colours correspond to sampling stations. The chronic toxicity threshold is near 1500 µg/L.

3 DISCUSSION

The ecotechnologies described herein emphasize control of HAB by changing reservoir biogeochemistry. Internal nutrient loading is a critical and ubiquitous contributor to poor water quality natural and constructed basins worldwide.

Where anoxia strongly influences internal loading relief destruction of anoxia is necessary to improve water quality. The eutrophic Swiss Lake Sempachersee is often cited as a failure of oxygenation because the internal loading did not decrease despite 15 years of oxygenation (Gächter and Müller, 2003). In this lake, there was enough sulphur to scavenge the iron, sequestering it in sediments and rendering unavailable to bind phosphate. The case studies cited above included geochemical augmentation with iron because of early identification of iron deficiency to bind phosphate (Walker et al., 1989). Sufficient conditions for quenching anoxia-driven internal loading entail attention to geochemical requirements of phosphate sequestration in sediments.

When sufficient conditions are met, the Minnesota case study demonstrates complete quenching of sediment internal phosphorus loading. Internal nutrient loading, however, is a more complex phenomenon than phosphate release from anoxic sediments alone (Nürnberg, 2009).

Internal nutrient loading from thermocline is potentially significant. On a seasonal basis there can be an oxygen sag in the thermocline as algae and organic particles perch on a density gradient at the top of the hypolimnion (Figure 12). Migrating cyanobacteria can readily access nutrients released to the thermocline. While there may be sufficient ferric iron at the sediment surface to sequester hypolimnetic phosphate, the metalimnion may be iron-deficient or even in early stages of anoxia. Although no algae blooms have yet been noted caused by an oxygen sag in the thermocline, the possibility should not be dismissed, and merits consideration by design engineers.

Results from Veteran's Lake and Lake Bowen indicate that geochemical augmentation with aluminium salts is effective at controlling total phosphorus and cyanobacteria blooms in shallow basins. Creation of a phosphate scavenging geochemistry in water helps manage a broad spectrum of internal and external nutrient loading inputs that are unrelated to anoxia.

Geochemical augmentation with aluminium salts is constrained by chronic toxicity, but aluminium concentrations allowed EPA criteria enable substantial removal of total phosphorus from the water column. A key consideration here is the pattern of TP enrichment. Lowing mean TP is an improvement to water quality. Reducing the incidence of excessive TP concentration may be more meaningful to utilities than mean or median TP. Spikes in TP concentration fuel blooms if sustained. The project evidence indicates that geochemical augmentation erodes TP concentration at rates on the order of 1 or 2 μg/L/d. In many basins this constant suppression of TP concentrations will reduce the duration of TP spikes, which will tend to limit excessive algae production.

Choice of aluminium salts is important. Aluminium sulphate (alum) will raise the sulphate concentration in the water, but aluminium chlorohydrate (ACH) or poly-aluminium chloride (PAC) will not. Because sulphate reduction will stimulate production of methylmercury from sediments (Watras et al., 1995), and is a fundamental driver of internal nutrient loading, adding sulphate may be inadvisable. Other aluminium salts eliminate this potential adverse effect of geochemical augmentation.

The destruction of the incipient cyanobacteria bloom in C.W. Bill Young reservoir was a decisive demonstration of the hydrodynamic control theory formulated by Huisman et al Huisman et al. (2004). The fundamental lesson here is not, however, that destratification aeration is an effective means of suppressing cyanobacteria blooms. It failed completely in Veteran's Lake because it is too shallow. If a lake or reservoir is deep enough, destratification will destroy or prevent cyanobacteria blooms. A depth of about 20 m is apparently deep enough. The reader is invited to study the theory devised by Huisman et al to determine if a given reservoir is deep enough and has enough wind mixing to be a candidate for this technology. A key concept here is that destratification does not mix the reservoir. Rather, it creates isothermal conditions that facilitate wind-driven turnover (Cooke et al., 2005).

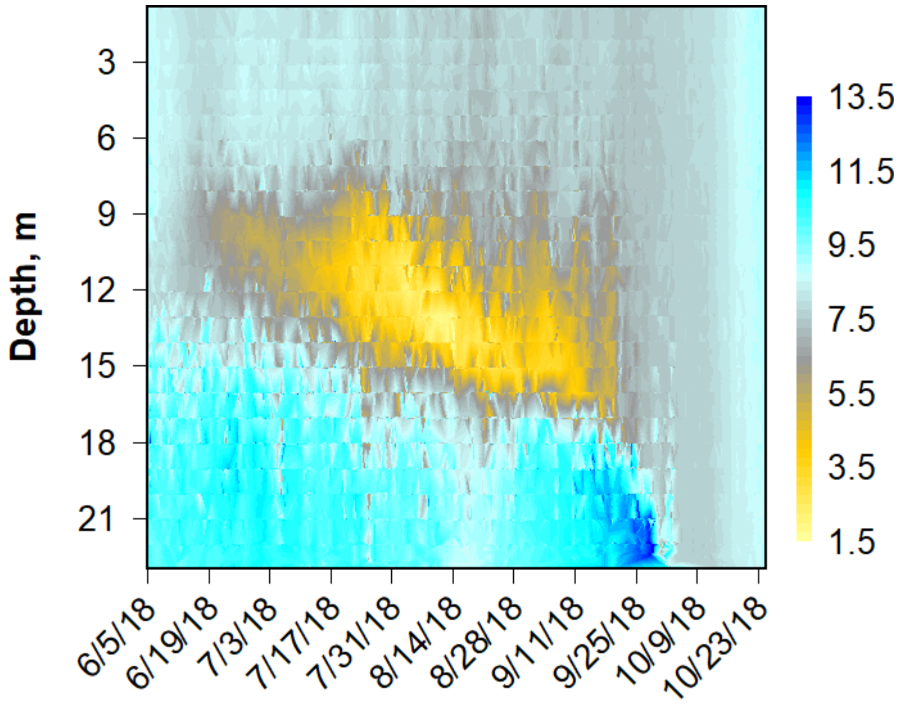


Figure 12. Dissolved oxygen (mg/L) isopleth for confidential client with linear diffuser pure oxygen injection into the hypolimnion, USA.

4 CONCLUSIONS

Lessons learned from projects demonstrate that HAB can be controlled changing reservoir geochemistry. The reservoir should be considered as the first unit process in provision of drinking water. Engineering methods can exert a high degree of control over internal nutrient dynamics and thereby reduce the frequency and duration of HAB.

Control starts with geochemistry. Anoxia should not be tolerated within the reservoir. Injection of oxygen is a straightforward means of destroying anoxia and is demonstrably superior to aeration of any type. Manufacturing diffusers at the reservoir edge and supplying them with pure oxygen is clearly less capital and operationally intensive than major treatment plant upgrades. Addition of aluminium or iron salts expand the potential for geochemistry management beyond destruction of anoxia.

Destratification aeration remains a candidate technology in some cases for managing cyanobacteria blooms. The caveat here is that sophisticated hydrodynamic investigation of the potential for this method should be mandatory. Many reservoirs with either be too shallow for this method or deep enough to be ideal suited for pure oxygen injection.

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