The effects of hypolimnetic oxygenation on the chemical, physical, and biological properties of a shallow drinking water reservoir

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ABSTRACT

Hypolimnetic anoxia can result in higher internal phosphorus (P) loads from the sediments to the water column, thereby increasing nutrient availability, making preventing anoxia a major goal for lake managers to improve water quality. Side-stream saturation (SSS), a type of hypolimnetic oxygenation system, has been developed to maintain oxygenated conditions at the sediments by withdrawing oxygen-depleted water from the hypolimnion to an on-site facility and injecting it with oxygen under high pressure before returning it to the hypolimnion. While this technique has been studied in select water bodies, to date it has not been successfully deployed in a shallow lake. This study investigated the effects of an SSS system deployed at Falling Creek Reservoir, a shallow drinking water reservoir located in Vinton, Virginia, USA. Specifically, we examined the effects of the SSS system on several chemical, physical, and biological response variables to ascertain the short-term impacts of hypolimnetic oxygenation on reservoir water quality. We found that the SSS system was successful in increasing dissolved oxygen concentrations in the reservoir hypolimnion without weakening stratification, warming the sediments, or increasing turbidity; however, we were unable to detect any short-term effects of SSS operation on P concentrations, P loading, pH, chlorophyll *a*, or algal density. Interestingly, we also observed an increase in oxygen demand in response to SSS operation, which must be taken into account when deploying oxygenation systems in the future. Continued monitoring is necessary to more completely assess the long-term impacts of SSS operation on water quality at Falling Creek Reservoir.

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1.0 INTRODUCTION

Eutrophication and Water Quality in Freshwater Lakes and Reservoirs

The deteriorating condition of freshwater resources is one of the most critical challenges impacting human and ecosystem health worldwide (Gleick, 2003; NRC, 2004). With most freshwater contained in glaciers or otherwise inaccessible, lakes and reservoirs represent the greatest source of available freshwater on the planet (Gleick and Palaniappan, 2010). Freshwater resources are critical in providing water for drinking, irrigation, industrial processes, and other ecosystem services (Millennium Ecosystem Assessment, 2005). Additionally, rising water demand and treatment costs have resulted in a critical effort to improve source water quality in lakes and reservoirs.

The eutrophication of freshwater sources has been a concern for decades (Hasler, 1969; Schindler, 1974; Schindler and Fee, 1974; Vollenweider, 1968; Vollenweider et al., 1974), and reducing the loading of nutrients to lakes and reservoirs is a key step for improving water quality (Moss, 1996; Smith et al., 1999; Welch and Jacoby, 2001). The limiting nutrient for productivity in these temperate lakes is typically phosphorus (P) (Schindler, 1977; Smith, 1983), but nitrogen (N) limitation also plays a role in some cases (e.g., Conley et al., 2009; Paerl, 2009; Paerl et al., 2011). After P enters the lake or reservoir, it tends to accumulate in the sediments, establishing a pool of potentially-mobile P (Larsen et al., 1981; Rydin, 2000; Sondergaard et al., 2001). Once a large stock of P has accumulated in the sediments (Carey and Rydin, 2011), thermal stratification can result in dissolved oxygen (DO) depletion and reducing conditions that favor the release of P from sediment iron (Fe) complexes (Cooke et al., 2005). Due to the feedback of P release from lake and reservoir sediments, internal loading has been shown to have a persistent impact on P concentrations even after external loading controls are adopted (Burger et al., 2008; Larsen et al., 1981; Marsden, 1989; Sondergaard et al., 1999; Welch and Cooke, 2005). Therefore, in freshwater lakes and reservoirs that have experienced extensive P deposition, the reduction of internal P loading from sediments is a crucial measure for controlling eutrophication.

In particular, controlling internal P loading in lakes and reservoirs can play a critical role in preventing cyanobacterial blooms (Paerl and Huisman, 2008; Brookes and Carey, 2011). In addition to factors such as increased temperature (Davis et al., 2009; Elliott, 2010; Liu et al., 2011; Paerl and Paul, 2012; Robarts and Zohary, 1987), light availability (Castenholz and Garcia-Pichel, 2012; Huisman et al., 1999; Richardson et al., 1983; Tilzer, 1987), water column stability (Lindenschmidt and Chorus, 1998; Steinberg and Hartmann, 1988; Walsby et al., 1997; Zhang and Prepas, 1996), and decreased zooplankton herbivory (Low et al., 2010; Sarnelle, 1993; Sommer and Lewandowska, 2011), increased nutrient availability is well-known to promote cyanobacterial proliferation in lakes and reservoirs (Cunha and Calijuri, 2011; Heisler et al., 2008). Cyanobacteria benefit as nutrient loading increases concentrations of growth-limiting nutrients. Certain cyanobacteria are capable of producing toxins that can lead to adverse health effects in humans and animals, causing concerns about exposure to contaminated drinking water sources during blooms (Carmichael, 1994; Falconer, 1999; WHO, 1999). Such growth can also lead to unpleasant taste and odor problems (Silvey et al., 1972; Smith et al., 2002). Controlling cyanobacterial blooms is an important part of the management strategies for freshwater lakes and reservoirs and ultimately results in additional costs incurred by water utilities. Bloom events are increasing in frequency and intensity worldwide (Hobson et al., 2012) and are expected to become even more widespread due expected changes in climate (Carey et al., 2012; Elliott, 2012; Paerl and Paul, 2012). As a result of this growing challenge, an increasing emphasis is being placed on mitigating the effects of eutrophication through the use of internal nutrient control.

The Potential of Hypolimnetic Oxygenation to Control Internal Loading

A variety of strategies exist for controlling internal nutrient loading in lakes and reservoirs. These include techniques such as nitrate addition (Ripl, 1976), sediment removal, sediment capping, chemical precipitation, hypolimnetic withdrawal, and hypolimnetic oxygenation (Cooke et al., 2005). While potential negative consequences are associated with each of these methods (Cooke et al., 2005), hypolimnetic oxygenation compares favorably in that it involves little physical disturbance to reservoir sediments, avoids reliance on the addition of chemical precipitates, and can treat poor-quality hypolimnetic water onsite rather than discharging it to downstream regions. Many hypolimnetic oxygenation systems have been deployed worldwide, and extensive reviews have been conducted summarizing results and case studies (Beutel and Horne, 1999; Cooke et al., 2005, Singleton and Little, 2006).

The purpose of hypolimnetic oxygenation is to maintain an oxygenated environment at the bottom of the lake or reservoir, thereby suppressing the release of nutrients, particularly P, from the sediments (Cooke et al., 2005). This process aims to raise the oxygen content of the hypolimnion without destratifying the overlying water column (Ashley, 1985; Lorenzen and Fast, 1977; McQueen and Lean, 1986). Such mixing could result in the entrainment of P that was isolated in the hypolimnion into the epilimnion, resulting in elevated P concentrations in the photic zone and increased nutrient availability. Furthermore, destratification would expand the water volume over which oxygen must be delivered to maintain desired DO concentrations, leading to inefficiency and added costs. Hypolimnetic oxygenation also aims to avoid heating of the hypolimnion, which could further destabilize thermal stratification and also pose a problem for benthic, cold-water organisms (Beutel and Horne, 1999).

The earliest documented application of this lake management technique used mechanical agitation in a splash basin to aerate water pumped from the hypolimnion (Mercier and Perret, 1949). More recently, common systems include deep water oxygen injection systems such as bubble-plume diffusers, and submerged down-flow bubble contact chambers such as the Speece Cone (Beutel and Horne, 1999; Singleton and Little, 2006). Bubble-plume diffuser systems have been successful in raising DO concentrations (Gachter and Wehrli, 1998; James et al., 1986; Mobley and Brock, 1995; Prepas and Burke, 1997). Deployment of Speece Cone chambers has also resulted in oxygenated hypolimnia in Camanche Reservoir, CA (Horne, 1995) and Newman Lake, WA (Thomas et al., 1994). In several cases, hypolimnetic oxygenation was a major factor in reducing internal P loading, thereby decreasing P concentrations (Gemza, 1997; Jung et al., 1998; Prepas and Burke, 1997; Liboriussen et al., 2009; Moore and Christensen, 2009) and reducing the severity of cyanobacterial blooms (Moore et al., 2012; Webb et al., 1997).

Ultimately, while these systems have been successful in elevating hypolimnetic DO and improving water quality, several associated drawbacks have been documented. In lakes and reservoirs that are subject to substantial external loading, the effects of hypolimnetic oxygenation on internal cycling may be masked (Gachter, 1987; Garell, 1977; Liboriussen et al., 2009). Hypolimnetic oxygenation may also induce increases in sediment and water column oxygen uptake, thereby accelerating hypolimnetic oxygen depletion and diminishing overall performance (Bryant et al., 2011; Gantzer et al., 2009). Heating of the hypolimnion occurred at Camanche Reservoir, CA, with hypolimnetic temperature increasing from 13.5 to 15 °C (Beutel and Horne, 1999). An additional limitation of these systems is that they may not be as effective in shallower lakes and reservoirs that lack sufficient depth to ensure that bubbles of injected oxygen dissolve in the hypolimnion or that thermal stratification is not disrupted by system operation (Cooke et al., 2005).

For implementing hypolimnetic oxygenation in shallower lakes and reservoirs, a technique known as side-stream saturation (SSS) represents an alternative to systems described above (Beutel and Horne, 1999). This technique involves withdrawing hypolimnetic water from the lake, injecting it with concentrated oxygen gas under high pressure, and then returning the oxygenated water to the hypolimnion (Singleton and Little, 2006). An early SSS application occurred at Ottoville Quarry, Ohio, resulting in elevated DO and maintenance of thermal stratification (Fast et al., 1975). SSS systems in Lake Bard, California (Debroux et al., 2012) and Lake Serraia, Italy (Toffolon et al., 2012) were successful in raising hypolimnetic DO and reducing water column P, although operation resulted in substantial heating and premature overturn in Lake Serraia (Toffolon et al.,2012). SSS systems have also been deployed for improving water quality in rivers, including the Canning River, Australia (Boardman, 1998) and the Tombigee River, Alabama (Speece, 1996). These cases, which represent the extent of documented SSS deployment to date, illustrate the application of SSS either to river systems or to lakes and reservoirs exceeding 18 m depth. To date, there have been no documented cases of successful SSS deployment in shallow lakes less than 10 m deep. Given that most lakes and reservoirs worldwide are small and shallow (Downing et al., 2006; Scheffer, 2004), it is a major goal to investigate if SSS application can improve water quality in these ecosystems.

In September 2012, an SSS system was installed in shallow (maximum depth 9.3 m), eutrophic Falling Creek Reservoir (FCR), Vinton, VA, USA to control internal nutrient loading. Our primary research question was to determine if SSS could successfully improve water quality in this shallow drinking water reservoir. We investigated the effects of the SSS system on the chemical, physical, and biological properties of this reservoir. Specifically, we examined the sensitivity of several reservoir response variables to system activation and deactivation to ascertain the short-term impacts of hypolimnetic oxygenation on reservoir water quality.

2.0 MATERIALS AND METHODS

2.1 Study Site Description

Falling Creek Reservoir (Figure 1), located in Bedford County northeast of Roanoke, VA, is operated and maintained by the Western Virginia Water Authority (WVWA) and provides drinking water for local residents. The reservoir covers 8.5 hectares with a water volume of 322 ML and a maximum depth of 9.3 m. FCR has exhibited cyanobacterial blooms associated with taste and odor problems in the spring and summer during its recent monitoring history. Nutrient enrichment assays have indicated that phytoplankton in FCR are strongly limited by P (Urbanczyk et al., unpubl. data), resulting in the need to control internal P loading. From 2007 to 2012, the WVWA monitored temperature, pH, DO, turbidity, color, algae counts, Fe, and Mn monthly at 0.1 m, 1.6 m, 3.8 m, and 6.2 m depth at the deep hole of FCR, which provided a historical context for testing the SSS system.

Timeline

During this study, we monitored FCR for a full year (April 2012 – May 2013) to ascertain background reservoir conditions before the SSS system was activated. The SSS system was deployed in September 2012, but not turned on until May 2013. The system was activated on 14 May 2013 and operated continuously for 5 weeks with the exception of a 4-day power outage during 13-17 Jun 2013. We deactivated the system on 20 Jun 2013 and the system remained out of operation until 1 Aug 2013. We turned the system on and off sequentially to measure the sensitivity of the reservoir to the system operation and compare reservoir conditions when the system was operating and when it was not.

2.2 SSS System Description

The SSS system was designed and deployed in FCR (Figure 1) in September 2012 under the direction of Gantzer Water Resources Engineering, LLC and Mobley Engineering, Inc. The system included withdrawal piping to remove water from the hypolimnion and distribution piping to return oxygenated water back to the hypolimnion. The entire piping system was positioned 0.5 m from the reservoir bottom. The distribution piping was 76 m in length and followed a traditional linear diffuser configuration with evenly-spaced nozzles to promote the uniform distribution of oxygenated water.

Unlike systems that aim to raise DO concentrations by injecting air or concentrated oxygen bubbles directly into the deep hypolimnion, the SSS system withdraws hypolimnetic water to an on-shore chamber where it is injected with concentrated oxygen gas and allowed to approach saturated levels. The oxygen gas dissolves into the water under high pressure after which the oxygenated water is returned to the hypolimnion. The process aims to provide greater oxygen transfer efficiency than would normally be achieved given the shallow depth of the hypolimnion in FCR. FCR has 90% of its storage volume at less than 5 m depth with a maximum depth of 9.3 m, making it substantially shallower than other lakes and reservoirs in which SSS has been studied. Ultimately, assessing the effects of the SSS system in this reservoir has significant intellectual merit because this technique could be applied to other lakes and reservoirs of similar size to address problems associated with hypolimnetic oxygen depletion.

2.3 Field Data Collection

2.3.1 Baseline Reservoir Conditions Prior to SSS Activation

Prior to the activation of the SSS system in May 2013, we monitored the reservoir weekly during the preliminary data collection period (June 2012 to May 2013), except for October 2012 to April 2013, for which monitoring was monthly. On every sample date, we collected temperature, DO, chlorophyll, and turbidity profiles at 7 sampling sites in a transect across the reservoir (Figure 1). Sampling sites were named in ascending order from the shallowest end of the reservoir (FCR10) to the deepest (FCR50).

We measured all variables following standard methods. Temperature, DO, chlorophyll, and turbidity profiles were measured using a Sea-Bird Electronics SBE-19plus SEACAT Conductivity, Temperature, and Depth (CTD) Profiler. Total phosphorus (TP) concentrations were measured along a depth profile (0.1 m, 0.8 m, 1.6 m, 2.8 m, 3.8 m, 5.0 m, 6.2 m, and 8.0 m) at the deep hole as well as in water entering the reservoir. A rectangular weir constructed by the WVWA with notch dimensions of 29.2 cm height and 110 cm width measured flow entering the reservoir. An INW AquiStar PT2X pressure sensor mounted within the weir box continuously recorded water levels every 15 minutes, which we used to calculate the flow through the weir. We also collected surface grab samples at site FCR50 every two weeks for phytoplankton enumeration.

2.3.2 SSS Operation Phase

The SSS system was activated on 14 May 2013 for 5 weeks. We collected CTD profiles at all reservoir sites 5 days prior to system activation and in the morning immediately preceding activation. We collected additional CTD profiles at all sampling sites 1 day following activation, 2 days following activation, and 7 days following activation. After the first week of operation, CTD profiles continued to be collected at least once per week. All other variables monitored during the preliminary phase were sampled weekly throughout the operation phase. The SSS system operated continuously at an oxygen addition rate of 27 kg/d.

2.3.3 SSS Deactivation Phase

The SSS system was deactivated on 20 Jun 2013. At the time of deactivation, the system had been in continuous operation for 5 weeks except for a 4-day power outage that occurred during 13-17 Jun 2013. Data collection procedures were as described above, with the exception of substituting a YSI ProODO handheld DO meter for the CTD for DO profile measurements after 3 Jul 2013. We re-activated the SSS system on 1 Aug 2013 once the system had been out of operation for 6 weeks.

2.3.4 Laboratory Analyses

We conducted all chemistry and phytoplankton analyses following standard methods at the WVWA laboratory. Phytoplankton enumeration was conducted according to Standard Method 10200-F (APHA, 2012) using a Fisher Scientific Digital Micromaster Premier compound microscope. We measured TP concentrations spectrophotometrically using the ascorbic acid method described in EPA Method 365.3 (EPA, 1989).

2.3.5. Statistical Analyses

We calculated the Schmidt stability, an index of the strength of the reservoir stratification (Idso, 1973), using Lake Analyzer (Read et al., 2011) in Matlab (R2012a 7.14.0.739) with temperature profiles and wind speed data measured at the Roanoke Regional Airport (ROA). The Roanoke Airport is located 20 km away from FCR. Schmidt stability (in J/m^2) indicates the degree to which waters from the hypolimnion could be mixed with the epilimnion and thus was useful for assessing the effect of SSS system operation on thermal stratification.

We established a mass balance for hypolimnetic TP as follows:

$$V * \left(\frac{dC}{dt} \right) = J_P A \qquad (1)$$

where V is the hypolimnion volume (m^3), C is the measured concentration of TP averaged over the 3 sampling depths (5.0 m, 6.2 m, 8.0 m) located within the hypolimnion (kg/m^3), and dC/dt is the change in TP concentration over the time interval between measurements ($kg/m^3/d$). Using this overall TP balance with a hypolimnion sediment surface area, A (m^2), we estimated the internal P loading from reservoir sediments, designated J_p ($kg/m^2/d$). We then used this term to compare internal P loading at different times during the experimental period and to ultimately assess the effect of the SSS system on internal P loading.

We also constructed an oxygen mass balance for the reservoir hypolimnion as follows:

$$V * \left(\frac{dC}{dt} \right) = M_{SSS} - D \qquad (2)$$

where V is the volume (m^3) of the hypolimnion, C is the volume-weighted oxygen concentration measured by profile (kg/m^3), and dC/dt is the change in oxygen concentration over the time interval between measurements ($kg/m^3/d$). During SSS operation, the oxygen mass added to the hypolimnion (kg/d) was measured and designated M_{SSS} in Eq. 2. By quantifying each of these terms in the oxygen mass balance, we estimated oxygen demand, D (kg/d). We then used this oxygen demand value to calculate the areal hypolimnetic oxygen demand (AHOD), the sediment oxygen demand (SOD), and the volumetric hypolimnetic oxygen demand (VHOD). We calculated AHOD by dividing the oxygen demand by the hypolimnion surface area. We calculated SOD as the oxygen demand divided by the surface area of the hypolimnetic sediments, and we calculated VHOD as the oxygen demand divided by the volume of the hypolimnion. These parameters were of interest because it was necessary for the SSS system to add sufficient oxygen to the hypolimnion to overcome the oxygen demand and increase oxygen concentrations. We could also observe any increases in oxygen demand, particularly SOD, induced by the SSS system (Ashley, 1983; Beutel, 2003; Moore et al., 1996).

3.0 RESULTS AND DISCUSSION

3.1 Dissolved Oxygen

Baseline reservoir conditions in 2012 in the absence of hypolimnetic oxygenation revealed substantial DO depletion extending even to the epilimnion (Figure 2). In early 2013, DO was uniformly distributed throughout the water column (Figure 2). DO concentrations steadily declined at an average rate of 0.13 mg/L/d prior to 14 May 2013 (Figure 4), resulting in concentrations < 6 mg/L in the hypolimnion immediately before SSS activation.

SSS activation resulted in an immediate increase in DO (Figure 2), with hypolimnetic DO increasing at a rate of up to 0.75 mg/L/d during the first 2 days of operation (Figure 4). Although DO concentrations continued to increase during the operation phase, we observed that the rate of increase was much lower after the first week of operation, averaging 0.16 mg/L/d after 21 May 2013. These results show that while reservoir DO responded quickly to the activation of oxygenation, a slower, constant rate of increase was achieved after about one week.

SSS operation resulted in a well-mixed hypolimnion, with uniform DO concentrations between 4.7 m and 9.0 m (Figure 2). We observed that SSS operation resulted in elevated DO even in the bottom 0.5 m of the reservoir, indicating that oxygen was added both above and below the diffuser. A sharply defined metalimnetic region between 3-4 m depth with lower DO persisted throughout the operation phase, presumably due to increased oxygen consumption at the steep density gradient. The occurrence of a metalimnetic DO minimum despite an oxygenated hypolimnion has been well documented (Bengtsson and Gelin, 1975; Garell et al., 1977; Walker et al., 1989; Steinberg and Arzet, 1984; McQueen and Lean, 1986) and could restrict fish movement (Cooke et al., 2005) and prevent complete mineralization of settling organic matter (Gachter, 1987).

A comparison of the spatial DO distribution showed that during operation, a well-mixed hypolimnion with increased DO was observed several hundred meters upstream from the SSS system in the reservoir, illustrating the large spatial extent of oxygenation (Figure 2). After SSS deactivation, the bulk hypolimnion in the deepest site of the reservoir near the SSS system was more resistant to DO depletion, with DO decreasing sharply near the sediments and occurring much more slowly in the upper hypolimnion. By comparison, in shallower regions without a well-developed hypolimnion, DO depletion extended throughout the water column 2 weeks after system deactivation.

During the deactivation phase, DO was observed to decrease within 5 days (Figure 2). The previously uniform structure of DO in the hypolimnion began to break down, with the greatest DO depletion occurring in the bottom meter of the hypolimnion near the sediments. DO depletion rates during the deactivation phase remained consistent, averaging 0.18 mg/L/d. Compared to the reservoir's initial fast response to SSS activation (a maximum rate of DO accumulation of 0.75 mg/L/d within 2 days), the response to SSS deactivation was slower (a maximum rate of 0.37 mg/L/d). However, we did observe a shift from DO accumulation to DO depletion within 5 days following deactivation (Figure 4).

The 4-day power outage in the middle of the operation phase and the shut-off of the SSS system during the deactivation phase indicated that the reservoir oxygen concentrations were very responsive to the system (Figure 3). Prior to power loss during 13-17 Jun 2013, the rate of change in DO concentrations was positive, indicating oxygen accumulation in the hypolimnion. While we did not collect oxygen profiles during the power outage, observed rates remained negative on 18 Jun 2013, less than 24 hours after the system was re-instated, indicating that the power outage resulted in a transition to oxygen depletion. SSS operation resumed on 17 June until 20 June, at which time rates of DO change were positive, indicating a quick reestablishment of DO accumulation in response to system re-activation. DO depletion occurred once again following SSS deactivation on 20 June (Figure 4). These results illustrate the high short-term sensitivity of reservoir hypolimnetic DO concentrations in response to SSS operation. Given this sensitivity, reservoir management plans must account for and anticipate an immediate decrease in DO in the event the SSS system is taken out of operation.

These results demonstrated that the SSS system was successful in oxygenating the reservoir hypolimnion. SSS operation delivering 27 kg/d of oxygen was able to increase and maintain elevated DO in the hypolimnion compared to sampling periods without operation. These results compared favorably with those of other SSS systems deployed in different ecosystems, which utilized oxygen addition rates between 14 and 380 kg/d (Debroux et al., 2012; Fast et al., 1975). An SSS system in Ottoville Quarry, Ohio (max depth = 18 m) was observed to increase hypolimnetic DO from 0 to 8 mg/L after 2 months of oxygenation at a rate of 14 kg/d, yielding a DO accumulation rate of 0.13 mg/L/d (Fast et al., 1975). SSS operation in Lake Bard, California (max depth = 27.4 m) with an oxygen addition rate of 380 kg/d resulted in a DO accumulation rate of 0.29 mg/L/d, with DO depletion returning at a rate of 0.10 mg/L/d following deactivation (Debroux et al., 2012). Despite the success of these SSS systems, they were deployed in lakes 2-3 times deeper than FCR, further emphasizing the importance of this study in improving our understanding of the ability of SSS to successfully oxygenate the hypolimnia of shallow lakes and reservoirs such as FCR.

3.2 Oxygen Demand

We observed increases in SOD, AHOD, and VHOD in response to SSS operation based on mass balance estimations (Figure 5). Overall patterns in these response variables between sampling phases were similar, given that each variable differed only by the factor over which the oxygen demand was distributed. AHOD values were essentially identical to SOD, explained by the very shallow hypolimnion depth relative to the hypolimnetic and sediment surface areas.

During the preliminary phase prior to activation, mean SOD was estimated to be 0.25 $g/m^2/d$. However, during the majority of the operation phase, mean SOD was estimated to be 0.93 $g/m^2/d$. During the deactivation phase, mean SOD values fell to 0.35 $g/m^2/d$. These results represented an increase in SOD by a factor of 3.7 during the operation phase, indicating that SSS operation increased oxygen uptake by the reservoir sediments. This increase falls within the range of induced SOD factors reported in the literature, which ranges from 1.5 to 4.0 (Cooke et al., 2005). Shallow lakes such as FCR are expected to experience induced SOD factors at the high end of this range due to the larger fraction of oxygen demand exerted at the sediments compared to the overlying shallow water column (Beutel, 2003).

Induced oxygen demand is a crucial issue for the design of hypolimnetic oxygenation systems. Failure to account for increased oxygen uptake can result in the inability of the system to achieve increases in hypolimnetic DO. The increased oxygen demand during SSS operation likely contributed to the observed slowing of DO increase rates following the first week of operation (Figure 4). Nonetheless, given the observed ability of the SSS system to raise hypolimnetic DO concentrations from 6 to 12 mg/L at a rate of 0.16 mg/L/d during the operation phase, the SSS system design appears to be adequate in its capacity to overcome the corresponding increases in oxygen demand.

3.3 Total Phosphorus

Hypolimnetic TP concentrations did not exhibit a clear response to SSS operation (Figure 6). We observed that the stock of hypolimnetic TP accounted for a much smaller fraction of the overall TP mass stock within the water column. This finding was not surprising given that the hypolimnion only represents 12% of the total reservoir volume. Internal P loading (J_P) estimates from mass balance calculations (Figure 7) indicated that J_P fluctuated between net release and net uptake during each phase of the experimental period. As with the reservoir hypolimnetic TP stock, there was no clear observed change in internal P loading in response to SSS operation.

Although one of the recognized potential benefits of hypolimnetic oxygenation is a reduction of internal P loading and a subsequent decrease in TP concentrations (Beutel and Horne, 1999; Cooke et al., 2005; Prepas and Burke, 1997), such effects are not always observed. For example, Gachter and Wehrli (1998) found that although hypolimnetic oxygenation successfully maintained DO levels above 3 mg/L in the study site, the effects on internal P loading remained unclear. Other studies have also found little reduction (Ravera, 1990) or even short-term increases in TP (Thomas et al., 1994). Ultimately, the marginal effect on hypolimnetic TP concentrations resulting from hypolimnetic oxygenation may be due partly to a lack of redox sensitive iron complexes available to bind P (Lean et al., 1986). Furthermore, persistent external loading may prevent the observance of decreases in TP during oxygenation. For example, we observed increases in both epilimnetic and total water column TP during the deactivation phase (Figure 6). During the sampling period, FCR received substantial external P loads due to several large precipitation events (Figure 7), which may have contributed to the observed increases in epilimnetic and total TP. Thus, in the absence of external nutrient loading controls, the SSS system may not be as successful in controlling P concentrations.

3.4 pH

Baseline reservoir conditions in 2012 revealed that reservoir pH remained near-neutral during much of the sampling period, with slightly higher pH near the surface (Figure 8). One notable exception occurred in late May to early June 2012, with pH increasing above 8.5. This increase may be attributable to an observed increase in algal growth (Figure 12), which has been shown to increase pH as acidic carbon dioxide is consumed during photosynthesis (Moss, 1973; Seitzinger, 1991).

In comparison, pH during 2013 remained near-neutral throughout each experimental phase, with no clear response to SSS operation (Figure 8). We did observe elevated pH in late June 2013, which was also associated with a slight increase in algal cell density, although not to the same extent as in 2012 (Figure 12).

3.5 Thermal Stratification

SSS operation was not observed to decrease thermal stratification in FCR (Figure 9). During the 2013 preliminary phase, stratification was initially weak but strengthened progressively leading up to SSS activation. During the operation phase, temperature profiles showed stratification continuing to develop. We did not observe any heating of the hypolimnion attributable to SSS operation, as indicated by disruption or weakening of the thermal structure.

Thermocline strength, measured by Schmidt stability, supports this finding (Figure 9). Prior to SSS activation, Schmidt stability remained low, with values below 20 J/m^2. During the operation phase, Schmidt stability increased to about 40 J/m^2, and higher values continued to persist even after SSS deactivation. While the increase in Schmidt stability does correspond to the activation of the SSS system, it is more likely that this increase is a result of background seasonal development of stratification rather than a response to SSS operation. Nonetheless, the observation that stratification continued to strengthen during SSS operation is important and illustrates that the SSS system was able to successfully oxygenate the hypolimnion without excessive mixing of the water column. As discussed previously, a crucial design concern for hypolimnetic oxygenation systems is their potential to create turbulence and destratify the water column (Ashley, 1985; Lorenzen and Fast, 1977; McQueen and Lean, 1986), leading to a suite of water quality concerns. Results from this study indicate that the SSS system in FCR is appropriately sized not only to deliver adequate oxygen to the hypolimnion, but also to maintain thermal structure in the overlying water.

Temperature profiles from the 2012 sampling period revealed a need to consider that year-to-year variations in stratification can influence the susceptibility of the water column to mixing. In comparison to 2013, warming in 2012 developed earlier and extended much deeper into the water column, resulting in an overall weaker thermal structure. We were unable to compare Schmidt stability from the two years because the temperature profile measurements in 2012 only extended to 6.2 m. Regardless, these observed visual differences in stratification indicate that the impact of SSS operation on thermal structure could differ from year to year and should continue to be monitored.

3.6 Turbidity

We did not observe a response in FCR turbidity to SSS operation (Figure 10). In fact, hypolimnetic turbidity remained low during the operation phase, an encouraging result revealing the ability of the system to operate effectively without causing much physical disturbance of the underlying sediments. However, we did observe increased turbidity levels during the deactivation phase, beginning in early July 2013 in the epilimnion. This increase may be associated with the substantial external loading entering the epilimnion that occurred during the deactivation phase, as discussed previously (Figure 7). Particulate matter brought to the epilimnion during these loading events appeared to settle downward as the deactivation phase progressed, resulting in increased turbidity in the hypolimnion.

Baseline data indicated that turbidity in 2012 was fairly low and uniform in the water column before mid-June. Afterward, we observed increased turbidity centered around 6 m depth (Figure 10). While increased turbidity was observed in July during both 2012 and 2013, the depth at which these increases occurred differed. This difference may be due to differences in thermal stratification, with warmer temperatures extending much deeper into the water column in 2012 relative to 2013 (Figure 9). Water entering the reservoir from external tributaries had a mean temperature of 25.0 °C in 2012 and 17.7 °C in 2013, indicating that it may have penetrated deeper into the water column in 2012, while being restricted to the epilimnion in 2013.

3.7 Phytoplankton Biomass

We did not observe a response in chlorophyll *a* concentrations resulting from SSS operation (Figure 11). However, chlorophyll concentrations did increase towards the end of the operation phase and during the deactivation phase, likely in response to increasing temperatures as the season progressed. The greater extent of increased chlorophyll within the epilimnion during the deactivation phase may be the result of the dual effects of phytoplankton growth near the surface as well as biomass accumulation along the strong metalimnetic density gradient. We observed that chlorophyll concentrations during the deactivation phase were higher in comparison to 2012 (Figure 11) even though 2013 water temperatures were cooler (Figure 9).

These differences highlight the influence that other reservoir conditions may have in the observed year-to-year variations in chlorophyll concentrations.

The spatial distribution of chlorophyll within the reservoir (Figure 11) further reinforces our finding that although chlorophyll concentrations did not respond directly to SSS operation, they continued to increase during the 2013 sampling period, particularly toward the end of the deactivation phase. Chlorophyll concentrations not only increased over a greater depth range as the season progressed, but the increased chlorophyll was observed further away from the SSS system.

In addition to chlorophyll profiles, algal cell density did not exhibit a response to SSS operation (Figure 12). Algal counts remained low throughout the 2013 preliminary phase and appeared to increase during the rest of the sampling period through both the operation and deactivation phases. Similar to chlorophyll, cell density reached a maximum at the end of the sampling period in 2013. Data from 2012 show a notable spike in cell density around 1 Jun 2012. This spike was likely associated with a significant bloom event and may have contributed to an observed increase in pH (Figure 8). As with several of the other response variables, year-to-year variations in reservoir environmental conditions likely had a significant influence on observations of both chlorophyll and algal cell density and must be considered in long-term assessments of the effects of SSS operation.

4.0 CONCLUSIONS

This study examined the effects of the SSS hypolimnetic oxygenation system on several chemical, physical, and biological response variables in FCR. The SSS system had a substantial effect on hypolimnetic DO, resulting in an increase in DO from less than 6 mg/L to over 12 mg/L at a rate of 0.16 mg/L/d over several weeks despite an observed induced oxygen demand. We also showed that FCR was highly sensitive to the ceasing and restarting of oxygenation, as illustrated during a brief power outage and during activation and deactivation transitions.

Other response variables were less affected by the SSS system. Hypolimnetic TP concentration and internal P loading were not observed to change in response to oxygenation. Similarly, pH, chlorophyll, and algal cell counts did not appear to be directly affected by SSS operation. Turbidity remained low during operation, illustrating the system's ability to operate without physically disrupting the sediments and increasing hypolimnetic turbidity. Thermal stratification was also unaffected by SSS operation, indicating the system's capacity to oxygenate the hypolimnion without destratifying the water column or warming the sediments.

Overall, this study illustrated the success of the SSS system in its primary purpose of hypolimnetic oxygenation at FCR. While further monitoring is needed to fully assess the long-term effects of SSS oxygenation on parameters of interest, including TP and phytoplankton growth, the system shows substantial promise as a means for maintaining oxygenated conditions in FCR and other shallow freshwater sources.

REFERENCES

American Public Health Association, 2012. Standard methods for examination of water and wastewater. 22nd edition. E.W. Rice, R.B. Baird, A.D. Eaton, L.S. Clesceri, eds.

Ashley, K.I., 1983. Hypolimnetic aeration of a naturally eutrophic lake: physical and chemical effects. Can. J. Fish. Aquat. Sci. 40, 1343-1359.

Ashley, K.I., 1985. Hypolimnetic aeration: practical design and application. Water Research 19 (6), 735-740.

Bengtsson, L., Gelin, C., 1975. Artificial aeration and suction dredging methods for controlling water quality. Proceedings on Water Quality Symposium, 313-342.

Beutel, M.W., 2003. Hypolimnetic anoxia and sediment oxygen demand in California drinking water reservoirs. Lake and Reservoir Management 19 (3), 208-221.

Beutel, M.W., Horne, A.J., 1999. A review of the effects of hypolimnetic oxygenation on lake and reservoir water quality. Lake and Reservoir Management 15 (4), 285-297.

Boardman, B., 1998. Draft report on the 1997/1998 oxygenation trial at Bacon Street on the Canning River. Australian Water and River Commission. 46 p.

Brookes, J.D., Carey, C.C., 2011. Resilience to blooms. Science 334, 46-47.

Bryant, L.D., Gantzer, P.A., Little, J.C., 2011. Increased sediment oxygen uptake caused by oxygenation-induced hypolimnetic mixing. Water Research 45 (12), 3692-3703.

Burger, D.F., Hamilton, D.P., Pilditch, C.A., 2008. Modelling the relative importance of internal and external nutrient loads on water column nutrient concentrations and phytoplankton biomass in a shallow polymictic lake. Ecological Modelling 211, 411-423.

Carey, C.C., Ibelings, B.W., Hoffmann, E.P., Hamilton, D.P., Brookes, J.D., 2012. Eco-physiological adaptations that favour freshwater cyanobacteria in a changing climate. Water Research 46 (5), 1394-1407.

Carey, C.C., Rydin, E., 2011. Lake trophic status can be determined by the depth distribution of sediment phosphorus. Limnol. Oceanogr. 56 (6), 2051-2063.

Carmichael, W.W., 1994. The toxins of cyanobacteria. Scientific American 270 (1), 78-86.

Castenholz, R.W., Garcia-Pichel, F., 2012. Cyanobacerial responses to UV radiation. In: Ecology of Cyanobacteria II: Their Diversity in Space and Time, B.A. Whitton, ed., Springer Netherlands, 481-499.

Conley, D.J., Paerl, H.W., Howarth, R.W., Boesch, D.F., Seitzinger, S.P., Havens, K.E., Lancelot, C., Likens, G.E., 2009. Controlling eutrophication: nitrogen and phosphorus. Science 323, 1014-1015.

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

Cunha, D.G.F., Calijuri, M.D., 2011. Limiting factors for phytoplankton growth in subtropical reservoirs: the effect of light and nutrient availability in different longitudinal compartments. Lake and Reservoir Management 27, 162-172.

Davis, T.W., Berry, D.L., Boyer, G.L., Gobler, C.J., 2009. The effects of temperature and nutrients in the growth and dynamics of toxic and non-toxic strains of *Microcystis* during cyanobacteria blooms. Harmful Algae 8, 715-725.

Debroux, J., Beutel, M.W., Thompson, C.M., Mulligan, S., 2012. Designing and testing of a novel hypolimnetic system to improve water quality in Lake Bard, California. Lake and Reservoir Management 28 (3), 245-254.

Downing, J.A., Prairie, Y.T., Cole, J.J., Duarte, C.M., Tranvik, L.J., Striegl, R.G., McDowell, W.H., Kortelainen, P., Caraco, N.F., Melack, J.M., Middelburg, J.J., 2006. The global abundance and size distribution of lakes, ponds, and impoundments. Limnol. Oceanogr. 51 (5), 2388-2397.

Elliott, J.A., 2010. The seasonal sensitivity of cyanobacteria and other phytoplankton to changes in flushing rate and water temperature. Global Change Biology 16, 864-876.

Elliott, J.A., 2012. Is the future blue-green? A review of the current model predictions of how Climate change could affect pelagic freshwater cyanobacteria. Water Research 46 (5), 1364-1371.

EPA, 1989. Methods for Chemical Analyses of Water and Wastewater. Method 365.3. EPA Manual 600/4—79020.

Falconer, I.R., 1999. An overview of problems caused by toxic blue-green algae (cyanobacteria) in drinking and recreational water. Environmental Toxicology 14 (1), 5-12.

Fast, A.W., Overholtz, W.J., Tubb, R.A., 1975. Hypolimnetic oxygenation using liquid oxygen. Water Resources Research 11 (2), 294-299.

Gachter, R., 1987. Lake restoration: why oxygenation and artificial mixing cannon substitute for a decrease in the external P loading. Schweiz. Z. Hydrol. 49, 170-185.

Gachter, R., Wehrli, B., 1998. Ten years of artificial mixing and oxygenation: no effect on the internal phosphorus loading of two eutrophic lakes. Environ. Sci. Technol. 32, 3659-3665.

Gantzer, P.A., Bryant, L.D., Little, J.C., 2009. Effect of hypolimnetic oxygenation on oxygen depletion rates in two water-supply reservoirs. Water Research 43 (6), 1700-1710.

Garrell, M.H., Confer, J.C., Kirschner, D., Fast, A.W., 1977. Effects of hypolimnetic aeration on nitrogen and phosphorus in a eutrophic lake. Water Resources Research 13 (2), 343-347.

Gemza, A.R., 1997. Water quality improvements during hypolimnetic oxygenation in two Ontario lakes. Water Quality Research Journal of Canada 32, 365-390.

Gleick, P. H., 2003. Global freshwater resources: Soft-path solutions for the 21st century. Science 302, 1524-1528.

Gleick, P.H., Palaniappan, M., 2010. Peak water limits to freshwater withdrawal and use. Proceedings of the National Academy of Sciences of the United States of America 107 (25), 11155-11162.

Hasler, A.D., 1969. Cultural eutrophication is reversible. Bioscience 19 (5), 425-431.

Heisler, J., Glibert, P.M., Burkholder, J.M., Anderson, D.M., Cochlan, W., Dennison, W.C., Dortch, Q., Gobler, C.J., Heil, C.A., Humphries, E., Lewitus, A., Magnien, R., Marshall, H.G., Sellner, K., Stockwell, D.A., Stoecker, D.K., Suddleson, M., 2008. Eutrophication and harmful algal blooms: a scientific consensus. Harmful Algae 8, 3-13.

Hobson, P., Dickson, S., Burch, M., Thorne, O., Tsymbal, L., House, J., Brookes, J., Chang, D., Kao, S., Lin, T., Bierlein, K., Little, J., 2012. Alternative and innovative methods for source water management of algae and cyanobacteria. Water Research Foundation.

Horne, A.J., 1995. The 1993-94 Camanche Reservoir oxygenation experiment report. Report to EBMUD, Oakland, CA. 88 p.

Huisman, J., Jonker, R.R., Zonneveld, C., Weissing, F.J., 1999. Competition for light between phytoplankton species: experimental tests of mechanistic theory. Ecology 80 (1), 211-222.

Idso, S.B., 1973. On the concept of lake stability. Limnol. Oceanogr. 18 (4), 681-683.

James, W.F., et al. 1986. Water quality studies: Richard B. Russell and Clark Hill Lakes; Second annual interim report. Miscellaneous Paper EL-86-12, US Army Engineer Waterways Experiment Station, Vicksburg, Miss. 199 p.

Jung, R., Sanders, J.O., Lai, H.H., 1998. Improving water quality through lake oxygenation at Camanche Reservoir. Presentation at the Cal. Lake Manage. Soc., Corte Madera. September 1998.

Larsen, D.P., Schults, D.W., Malueg, K.W., 1981. Summer internal phosphorus supplies in Shagawa Lake, Minnesota. Limnology and Oceanography 26 (4), 740-753.

Lean, D.R.S., McQueen, D.J., Story, V.A., 1986. Phosphate transport during hypolimnetic aeration. Arch. Hydrobiol. 108, 269-280.

Liboriussen, L., Sondergaard, M., Jeppesen, E., Thorsgaard, I., Grunfeld, S., Jakobsen, T.S., Hansen, K., 2009. Effects of hypolimnetic oxygenation on water quality: results from five Danish lakes. Hydrobiologia 625, 157-172.

Lindenschmidt, K., Chorus, I., 1998. The effect of water column mixing on phytoplankton succession, diversity and similarity. Journal of Plankton Research 20 (10), 1927-1951.

Liu, X., Lu, X., Chen, Y., 2011. The effects of temperature and nutrient ratios on *Microcystis* blooms in Lake Taihu, China: an 11-year investigation. Harmful Algae 10, 337-343.

Lorenzen, M.W., Fast, A.W., 1977. A guide to aeration/circulation techniques for lake management. Ecol. Res. Ser. USEPA-600/3-77-004.

Low, E.W., Clews, E., Todd, P.A., Tai, Y.C., Ng, P.K.L., 2010. Top-down control of phytoplankton by zooplankton in tropical reservoirs in Singapore? The Raffles Bulletin of Zoology 58 (2), 311-322.

Marsden, M.W., 1989. Lake restoration by reducing external phosphorus loading: the influence of sediment phosphorus release. Freshwater Biology 21, 139-162.

McQueen, D.J., Lean, D.R.S., 1986. Hypolimnetic aeration: an overview. Water Pollut. Res. J. Can. 21, 205-217.

Mercier, P., Perret, J., 1949. Aeration station of Lake Bret. Schweiz. Ver. Gas. Wasserfach. Monatsbull. 29, 25-30.

Millennium Ecosystem Assessment, 2005. Ecosystems and human well-being: synthesis. Island Press, Washington, DC.

Mobley, M.H., Brock, W.G., 1995. Widespread oxygen bubbles to improve reservoir releases. Lake and Reservoir Management 11 (3), 231-234.

Moore, B.C., Chen, P.H., Funk, W.H., Yonge, D., 1996. A model for predicting lake sediment oxygen demand following hypolimnetic aeration. Wat. Resour. Bull. 32, 1-9.

Moore, B.C., Christensen, D., 2009. Newman Lake restoration: a case study. Part I. Chemical and biological responses to phosphorus control. Lake and Reservoir Management 25 (4), 337-350.

Moore, B.C., Cross, B.K., Beutel, M., Dent, S., Preece, E., Swanson, M., 2012. Newman Lake restoration: a case study. Part III. Hypolimnetic oxygenation. Lake and Reservoir Management 28 (4), 311-327.

Moss, B., 1973. The influence of environmental factors on the distribution of freshwater algae: an experimental study: II. The role of pH and the carbon dioxide-bicarbonate system. Journal of Ecology 61 (1), 157-177.

Moss, B., 1996. A land awash with nutrients – the problem of eutrophication. Society of Chemical Industry.

NRC (National Research Council), 2004. Confronting the nation's water problems: the role of research. Washington, D.C., National Academies Press.

Paerl, H.W., 2009. Controlling eutrophication along the freshwater-marine continuum: dual nutrient (N and P) reductions are essential. Estuaries and Coasts 32, 593-601.

Paerl, H.W., Huisman, J., 2008. Blooms like it hot. Science 320, 57-58.

Paerl, H.W., Paul, V.J., 2012. Climate change: links to global expansion of harmful cyanobacteria. Water Research 46 (5), 1349-1363.

Paerl, H.W., Xu, H., McCarthy, M.J., Zhu, G., Qin, B., Li, Y., Gardner, W.S., 2011. Controlling harmful cyanobacterial blooms in a hyper-eutrophic lake (Lake Taihu, China): the need for a dual nutrient (N & P) management strategy. Water Research 45, 1973-1983.

Prepas, E.E., Burke, J.M., 1997. Effects of hypolimnetic oxygenation on water quality in Amisk Lake, Alberta, a deep, eutrophic lake with high internal phosphorus loading rates. Can. J. Fish. Aquat. Sci. 54, 2111-2120.

Ravera, O., 1990. The effects of hypolimnetic oxygenation in the shallow and eutrophic lake Comabbio (North Italy) studied by “enclosure.” Verh. Int. Ver. Theor. Angew. Limnol. 24, 184-195.

Read, J.S., Hamilton, D.P., Jones, I.D., Muraoka, K., Winslow, L.A., Kroiss, R., Wu, C.H., Gaiser, E., 2011. Derivation of lake mixing and stratification indices from high-resolution lake buoy data. Environmental Modelling and Software 26, 1325-1336.

Richardson, K., Beardall, J., Raven, J.A., 1983. Adaptation of unicellular algae to irradiance: an analysis of strategies. New Phytologist 93 (2), 157-191.

Ripl, W., 1976. Biochemical oxidation of polluted lake sediment with nitrate – a new lake restoration method. Ambio 5 (3), 132-135.

Robarts, R.D., Zohary, T., 1987. Temperature effects on photosynthetic capacity, respiration, and growth rates of bloom-forming cyanobacteria. New Zealand Journal of Marine and Freshwater Research 21 (3), 391-399.

Rydin, E., 2000. Potentially mobile phosphorus in Lake Erken sediment. Water Research 34 (7), 2037-2042.

Sarnelle, O., 1993. Herbivore effects on phytoplankton succession in a eutrophic lake. Ecological Monographs 63 (2), 129-149.

Scheffer, M., 2004. Ecology of shallow lakes. Kluwer Academic Publishers, The Netherlands.

Schindler, D.W., 1974. Eutrophication and recovery in experimental lakes: implications for lake management. Science 184 (4139), 897-899.

Schindler, D.W., 1977. Evolution of phosphorus limitation in lakes. Science 195 (4275), 260-262.

Schindler, D.W., Fee, E.J., 1974. Experimental lakes area: whole-lake experiments in eutrophication. J. Fish. Res. Board Can. 31, 937-953.

Seitzinger, S.P., 1991. The effect of pH on the release of phosphorus from Potomac Estuary sediments: implications for blue-green algal blooms. Estuarine, Coastal and Shelf Science 33, 409-418.

Silvey, J.K., Henley, D.E., Wyatt, J.T., 1972. Planktonic blue-green algae: growth and odor-production studies. Journal (American Water Works Association) 64 (1), 35-39.

Singleton, V.L., Little, J.C., 2006. Designing hypolimnetic aeration and oxygenation systems – a review. Environmental Science and Technology 40, 7512-7520.

Smith, V.H., 1983. Low nitrogen to phosphorus ratios favor dominance by blue-green algae in lake phytoplankton. Science 221 (4611), 669-671.

Smith, V.H., Sieber-Denlinger, J., deNoyelles, F., Campbell, S., Pan, S., Randtke, S.J., Blain, G.T., Strasser, V.A., 2002. Managing taste and odor problems in a eutrophic drinking water reservoir. Lake and Reservoir Management 18 (4), 319-323.

Smith, V.H., Tilman, G.D., Nekola, J.C., 1999. Eutrophication: impacts of excess nutrient inputs on freshwater, marine, and terrestrial ecosystems. Environmental Pollution 100, 179-196.

Sommer, U., Lewandowska, A., 2011. Climate change and the phytoplankton spring bloom: warming and overwintering zooplankton have similar effects on phytoplankton. Global Change Biology 17, 154-162.

Sondergaard, M., Jensen, J.P., Jeppesen, E., 1999. Internal phosphorus loading in shallow Danish lakes. Hydrobiologia 408/409, 145-152.

Sondergaard, M., Jensen, J.P., Jeppesen, E., 2001. Retention and internal loading of phosphorus in shallow, eutrophic lakes. The Scientific World 1, 427-442.

Speece, R.E., 1996. Oxygen supplementation by u-tube to the Tombigee River. Wat. Sci. Tech. 43, 83-90.

Steinberg, C., Arzet, K., 1984. Impact of hypolimnetic aeration on abiotic and biotic conditions in a small kettle lake. Environmental Technology Letters 5, 151-162.

Steinberg, C.E., Hartmann, H.M., 1988. Planktonic bloom-forming cyanobacteria and the eutrophication of lakes and rivers. Freshwater Biology 20, 279-287.

Thomas, J.A., Funk, W.H., Moore, B.C., Budd, W.W., 1994. Short term changes in Newman Lake following hypolimnetic oxygenation with the Speece Cone. Lake and Reservoir Management 9 (1), 111-113.

Tilzer, M.M., 1987. Light-dependence of phytosyntesis and growth in cyanobacteria: implications for their dominance in eutrophic lakes. New Zealand Journal of Marine and Freshwater Research 21 (3), 401-412.

Toffolon, M., Ragazzi, M., Righetti, M., Teodoru, C.R., Tubino, M., Defrancesco, C., Pozzi, S., 2012. Effects of artificial hypolimnetic oxygenation in a shallow lake. Part 1: Phenomenological description and management. Journal of Environmental Management 114, 520-529.

Vollenweider, R.A., 1968. Scientific fundamentals of the eutrophication of lakes and flowing waters, with particular reference to nitrogen and phosphorus as factors in eutrophication. Organ. Econ. Coop. Dev. (Paris) Tech. Rep. DAS/CSI/68.27., 182.

Vollenweider, R.A., Munawar, M., Stadelmann, P., 1974. A comparative review of phytoplankton and primary production in the Laurentian Great Lakes. J. Fish. Res. Board Can. 31, 739-762.

Walker, W.W. Jr., Westerberg, C.E., Schuler, D.J., Bode, J.A., 1989. Design and evaluation of eutrophication control measures for the St. Paul water supply. Lake Reserv. Manag. 5, 71-83.

Walsby, A.E., Hayes, P.K., Boje, R., Stal, L.J., 1997. The selective advantage of buoyancy provided by gas vesicles for planktonic cyanobacteria in the Baltic Sea. New Phytologist 136 (3),407-417.

Webb, D.J., Robarts, R.D., Prepas, E.E., 1997. Influence of extended water column mixing during the first 2 years of hypolimnetic oxygenation on the phytoplankton community of Amisk Lake, Alberta. Can. J. Fish. Aquat. Sci. 54, 2133-2145.

Welch, E.B., Cooke, G.D., 2005. Internal phosphorus loading in shallow lakes: importance and control. Lake and Reservoir Management 21 (2), 209-217.

Welch, E.B., Jacoby, J.M., 2001. On determining the principle source of phosphorus causing summer algal blooms in western Washington lakes. Lake and Reservoir Management 17 (1), 55-65.

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

Zhang, Y., Prepas, E.E., 1996. Regulation of the dominance of planktonic diatoms and cyanobacteria in four eutrophic hardwater lakes by nutrients, water column stability, and temperature. Can. J. Fish. Aquat. Sci. 53, 621-633.

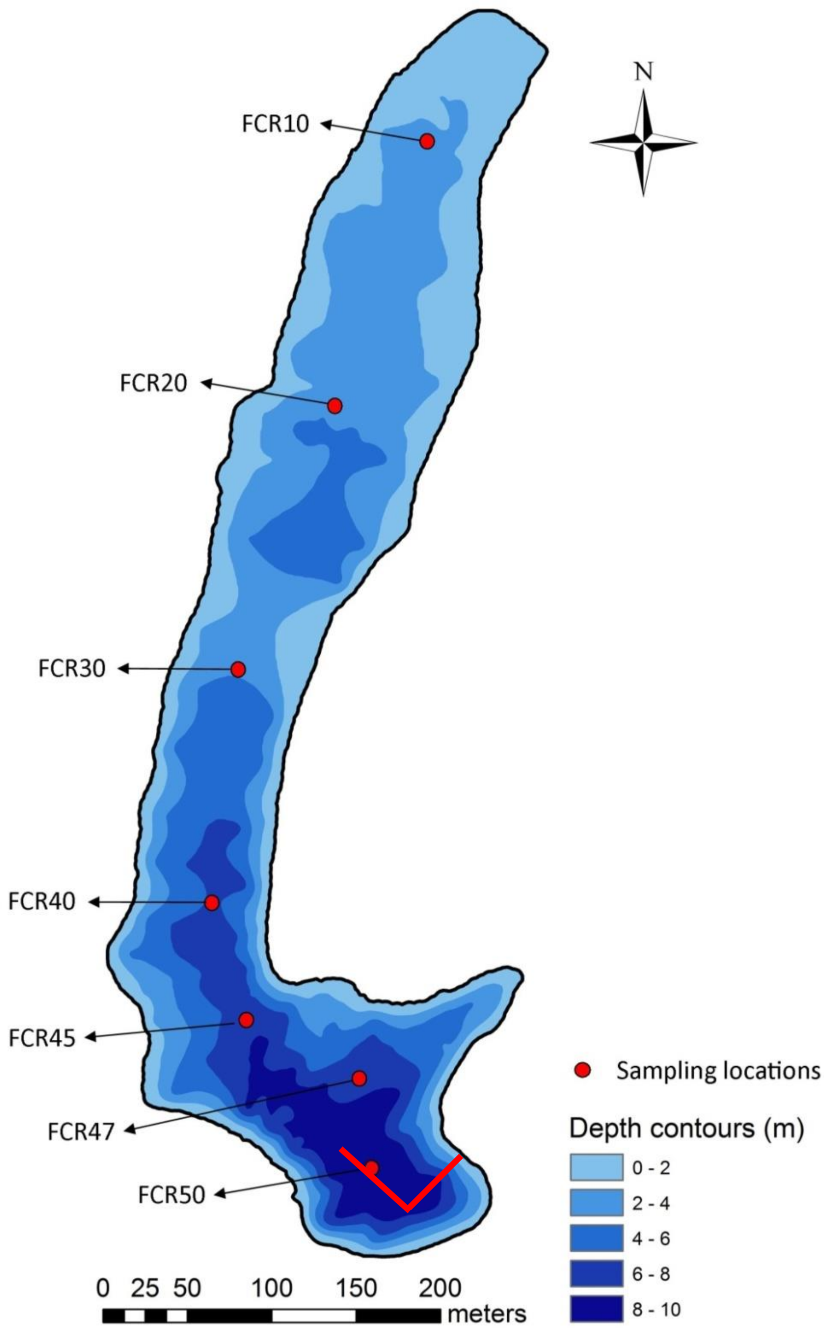

Figure 1. Map of Falling Creek Reservoir (FCR) bathymetry showing locations of sampling sites. Layout of side-stream saturation (SSS) system distribution and withdrawal piping shown with red lines at the deepest part of the reservoir.

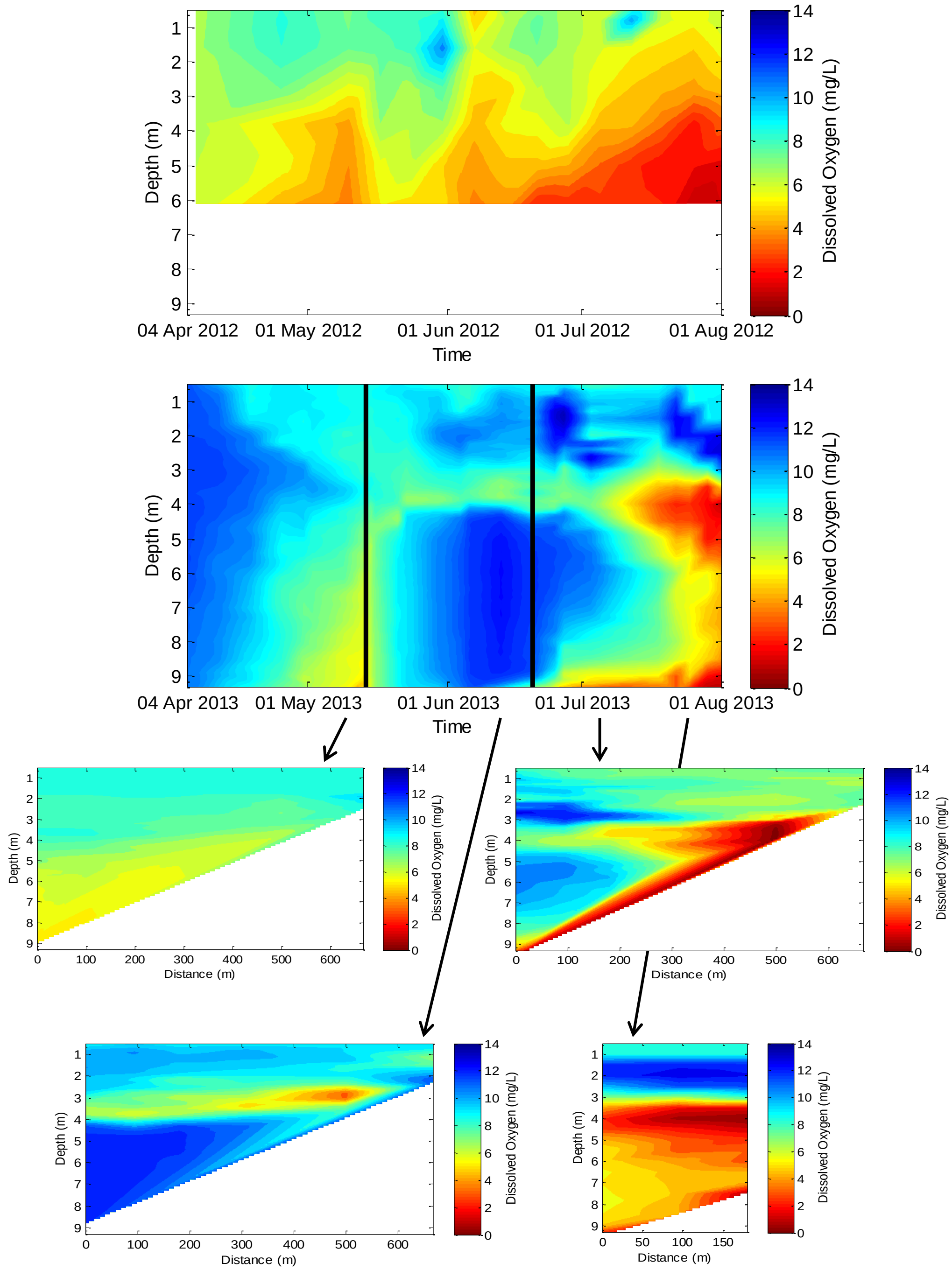

Figure 2. Dissolved oxygen (DO, in mg/L) profiles from site FCR50 during 2012 (top panel) and 2013 (second panel) sampling periods. SSS activation and deactivation are marked by black vertical lines. Additional panels include DO profiles plotted with distance from the SSS system on the following dates: 14 May 2013, 13 Jun 2013, 03 Jul 2013, and 25 Jul 2013. Profiles from 2012 extend only to 6.2 m since this was the maximum depth of sampling collection during this period. Profiles from 25 Jul 2013 were only collected up to 180 m from the SSS system.

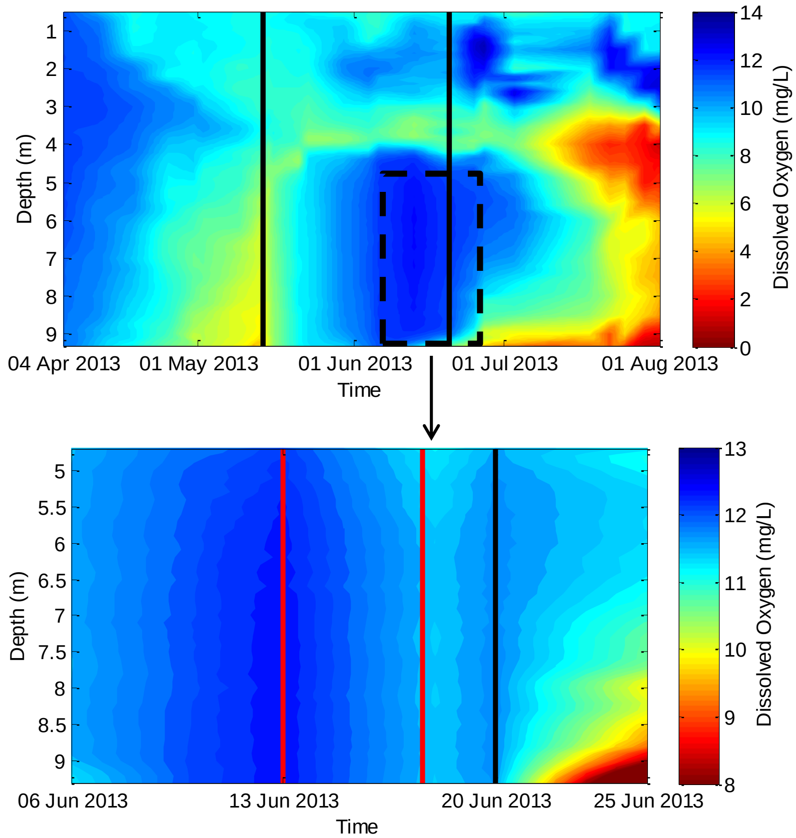

Figure 3. DO profiles from site FCR50 during the 2013 sampling period (top panel) and during a 4-day power outage (bottom panel). SSS activation and deactivation are marked by black vertical lines. Power loss and reestablishment are marked by red vertical lines. Profiles are displayed only over the hypolimnion in the bottom panel (identified by dotted box in top panel). The range of color contours is also adjusted to better illustrate subtle changes in DO during the power outage.

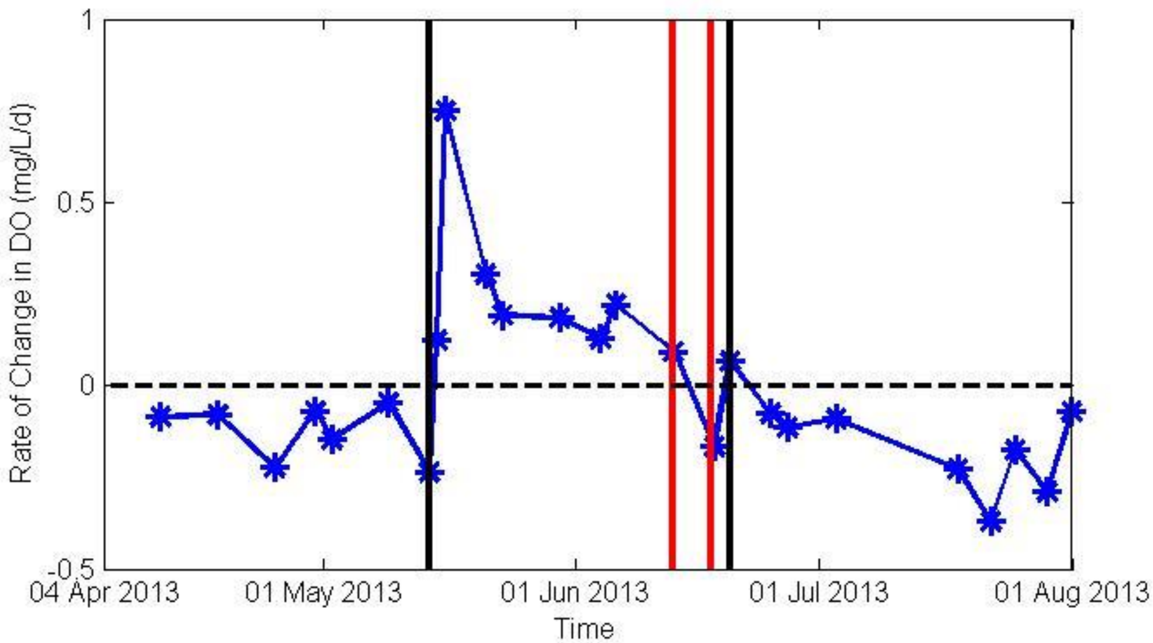

Figure 4. Rates of change in hypolimnetic DO (in mg/L/d) during the 2013 sampling period. SSS activation and deactivation are marked by black vertical lines. Power loss and reestablishment are marked by red vertical lines. Positive rates (increasing DO) and negative rates (decreasing DO) are separated by the horizontal dotted line.

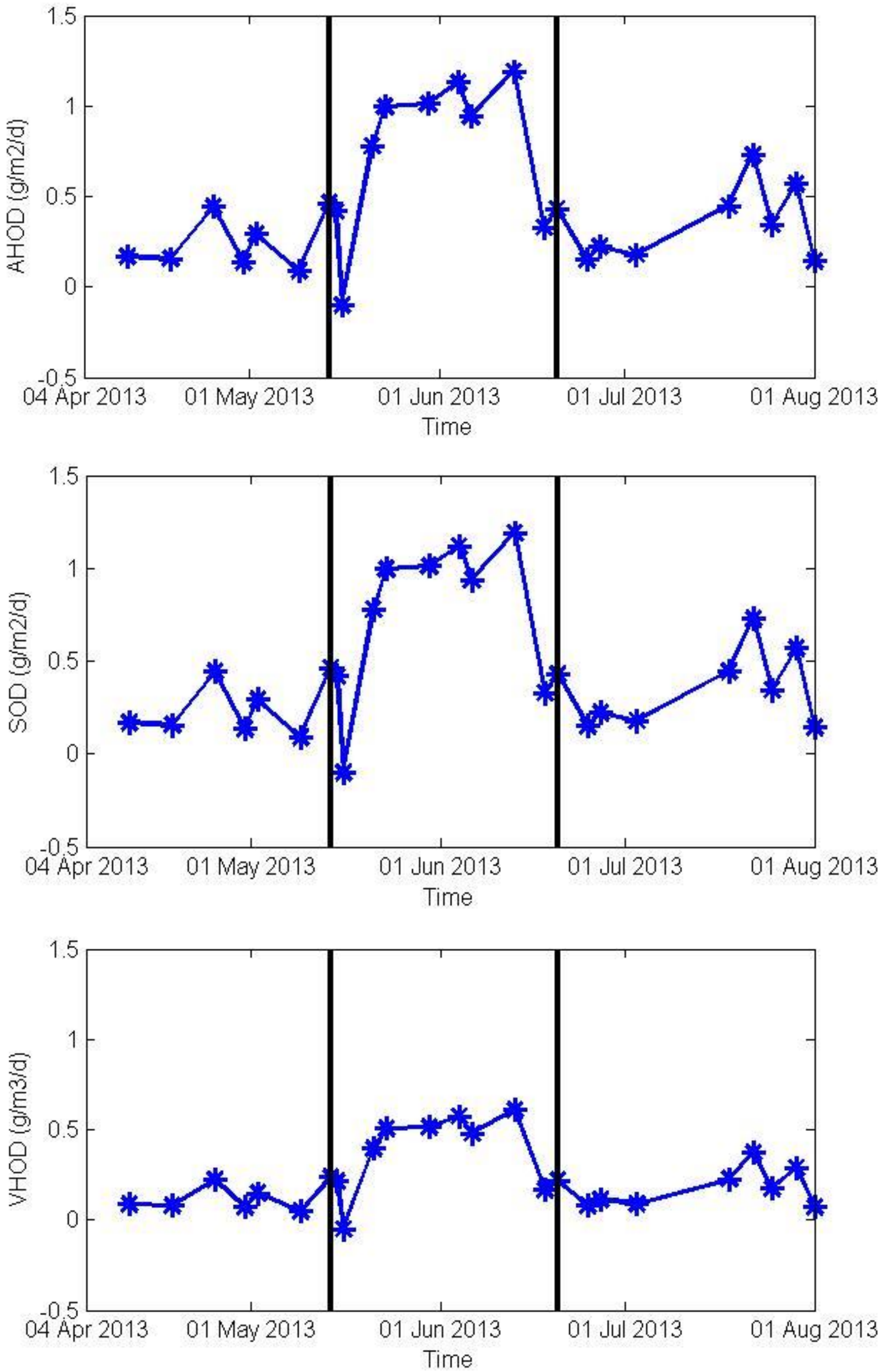

Figure 5. Areal hypolimnetic oxygen demand (AHOD, in $g/m^2/d$ (top panel)), sediment oxygen demand (SOD, in $g/m^2/d$ (middle panel)), and volumetric hypolimnetic oxygen demand (VHOD, in $g/m^3/d$ (bottom panel)) estimations during the 2013 sampling period. SSS activation and deactivation are marked by black vertical lines.

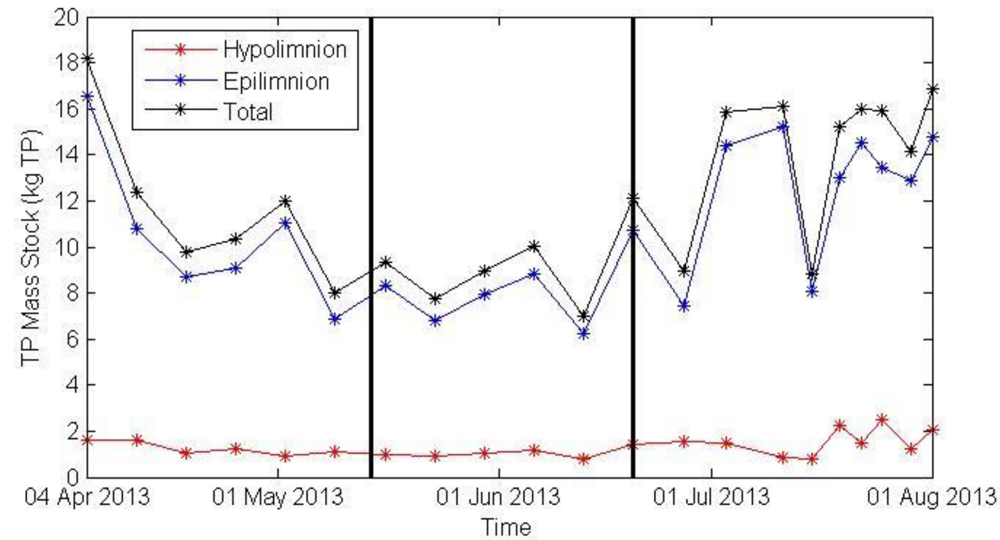

Figure 6. Total phosphorus stock (in kg TP) measured during the 2013 sampling period for the hypolimnion, epilimnion, and total reservoir volume. TP stock for the hypolimnion represents the average over the lower 3 sampling depths located in the hypolimnion. TP stock for the epilimnion represents the average over the upper 5 sampling depths. Total TP stock represents the sum of hypolimnetic and epilimnetic TP content. SSS activation and deactivation are marked by black vertical lines.

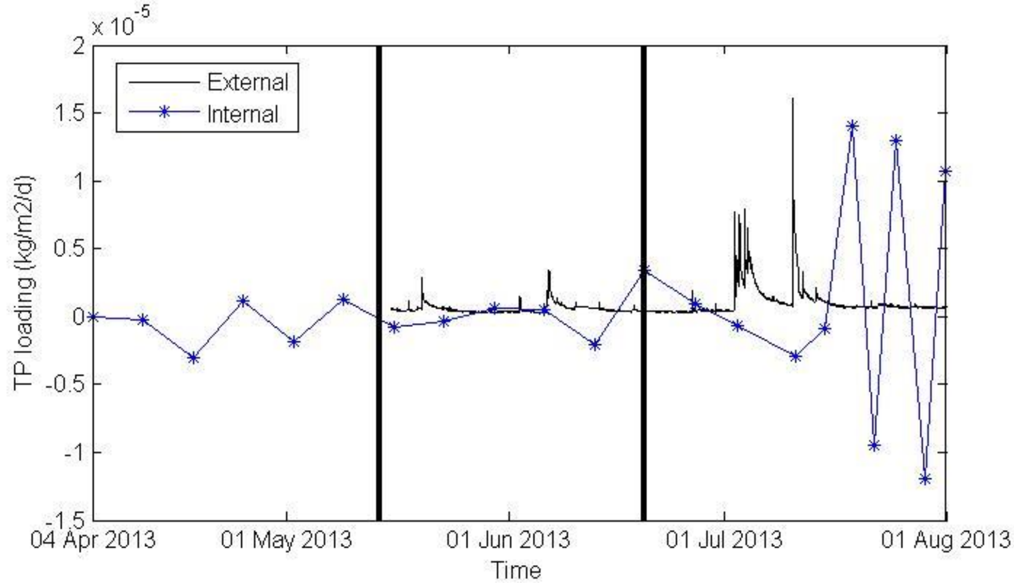

Figure 7. External and internal P loading (in $kg/m^2/d$) to FCR during the 2013 sampling period. External loading measurements commenced on 15 May 2013 when the weir and pressure sensor began to continuously monitor flow. External loading was calculated over 15-minute intervals while internal loading was calculated weekly. SSS activation and deactivation are marked by black vertical lines.

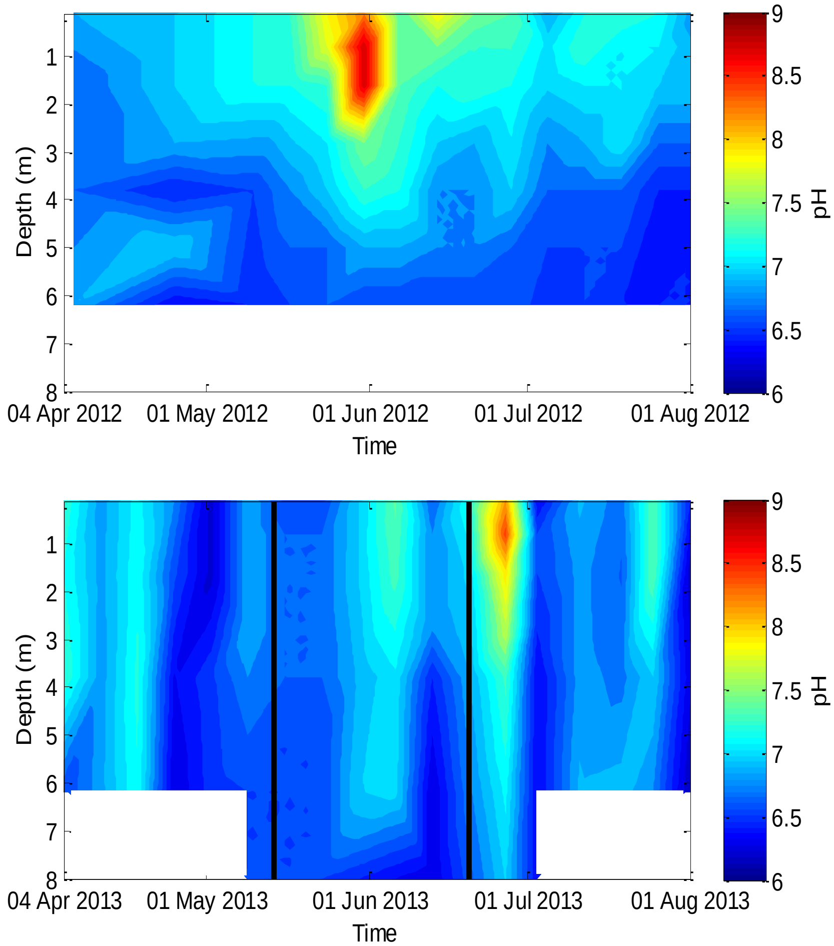

Figure 8. Profiles of pH from depth samples at site FCR50 during the 2012 (top panel) and 2013 (bottom panel) sampling periods. 2012 profiles and some 2013 profiles extended only to 6.2 m depth. SSS activation and deactivation are marked by black vertical lines.

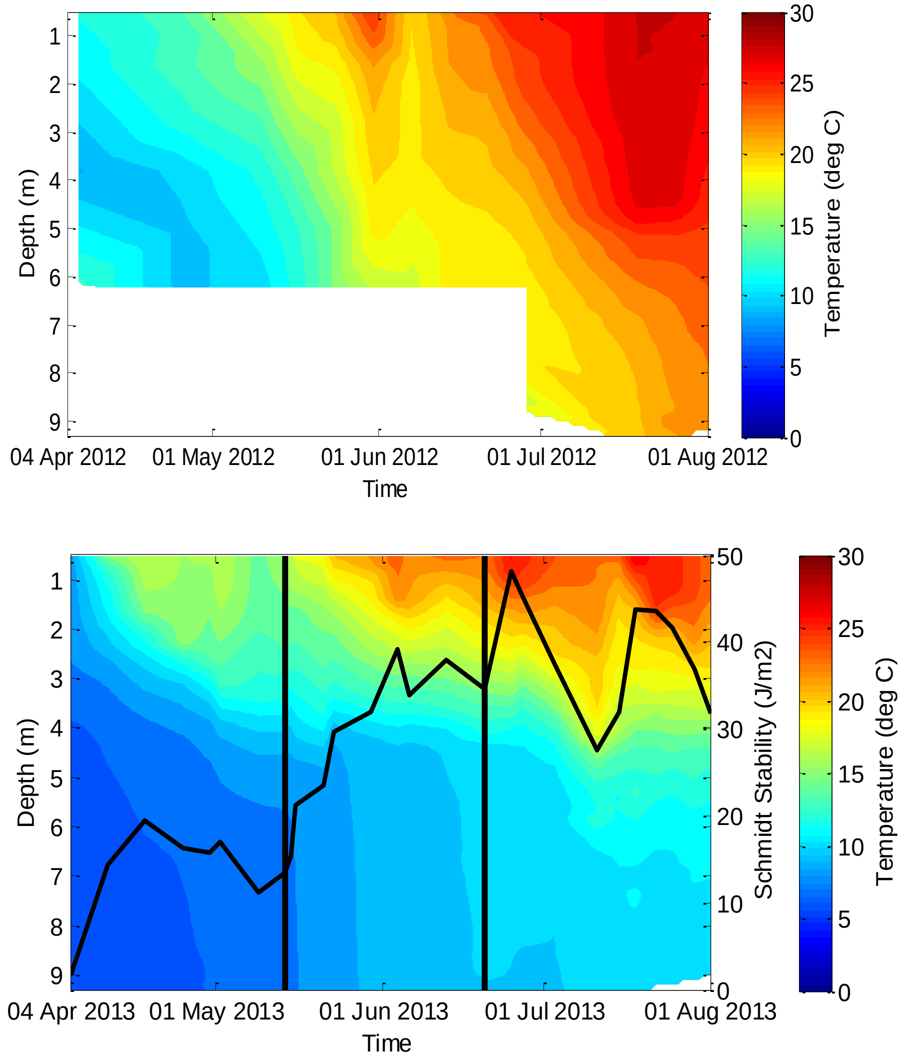

Figure 9. Temperature profiles (in °C) from site FCR50 during the 2012 (top panel) and 2013 (bottom panel) sampling periods. SSS activation and deactivation are marked by black vertical lines. Schmidt stability (in J/m^2) is plotted as a time series for 2013. Temperature profiles for 2012 were initially collected by depth samples up to 6.2 m. After 28 Jun 2012, profiles extended to the bottom depth of the reservoir.

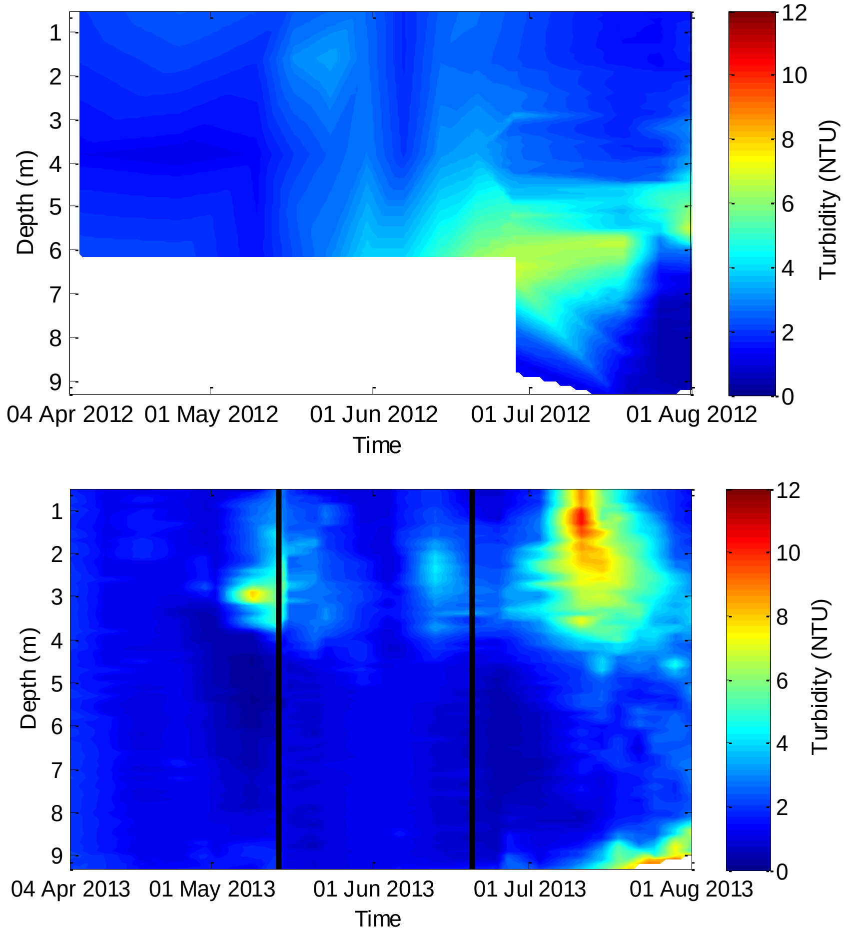

Figure 10. Turbidity (in NTU) profiles from site FCR50 during the 2012 (top panel) and 2013 (bottom panel) sampling periods. SSS activation and deactivation are marked by black vertical lines. Turbidity profiles for 2012 were initially collected by depth samples up to 6.2 m. After 28 Jun 2012, profiles extended to the bottom depth of the reservoir.

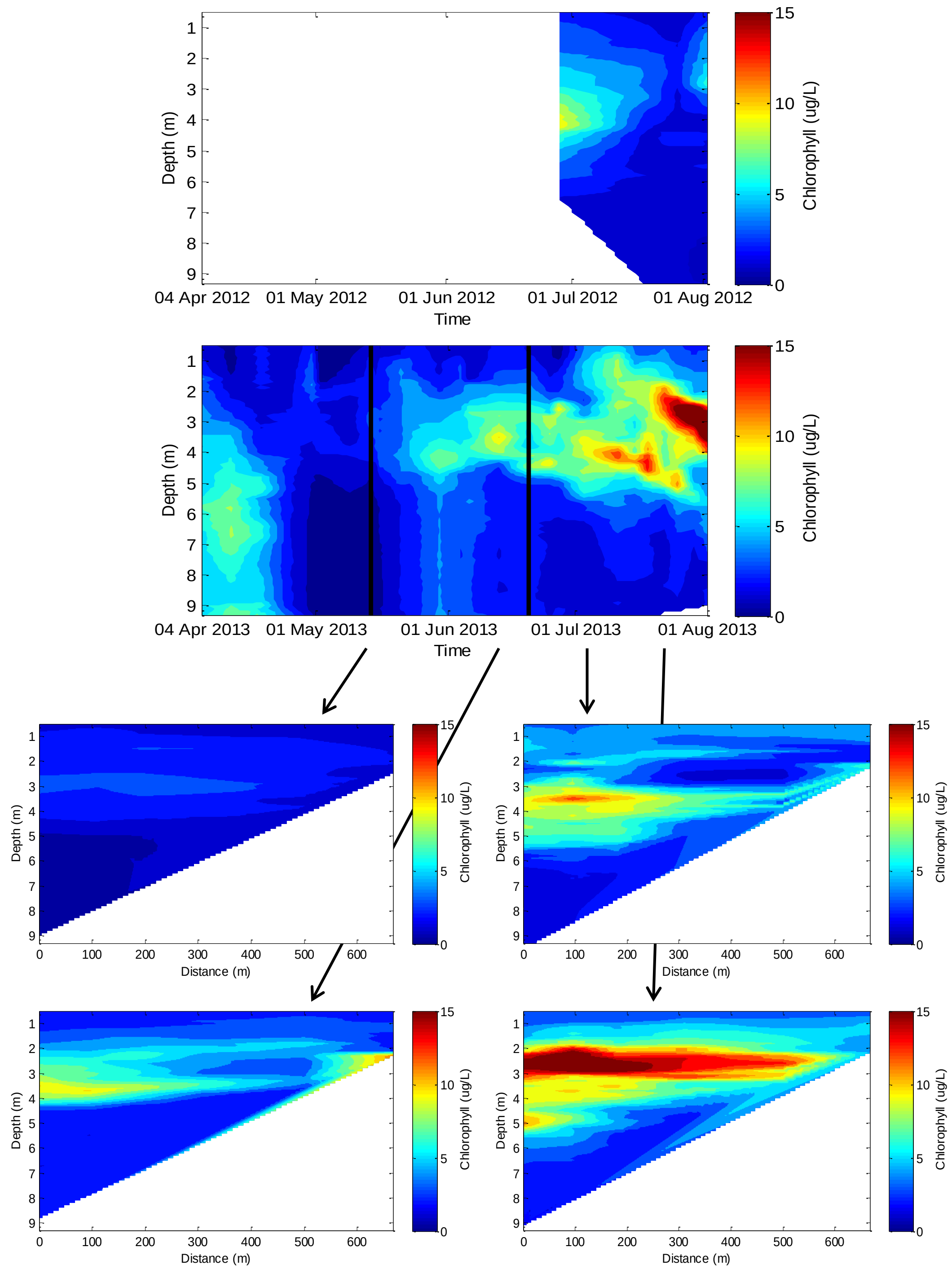

Figure 11. Chlorophyll *a* (in µg/L) profiles from site FCR50 during the 2012 (top panel) and 2013 (second panel) sampling periods. SSS activation and deactivation are marked by black vertical lines. Chlorophyll profile collection in 2012 did not begin until 28 Jun 2012. Additional panels include chlorophyll profiles plotted with distance from the SSS system on the following dates: 14 May 2013, 13 Jun 2013, 03 Jul 2013, and 25 Jul 2013.

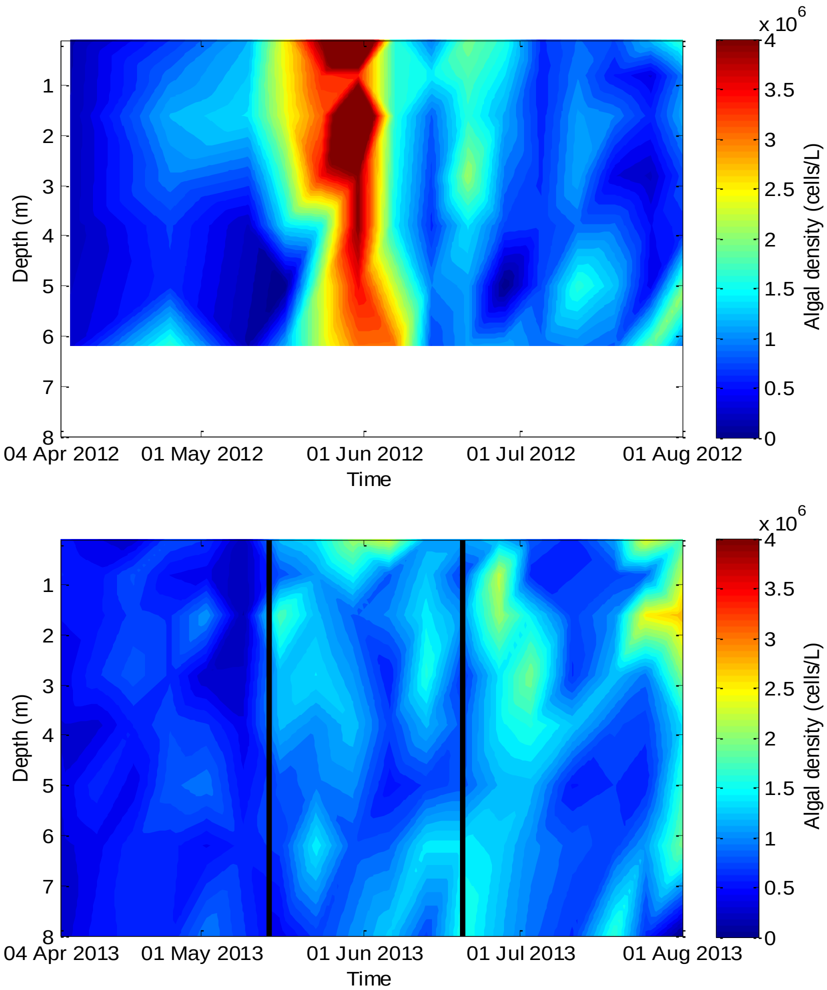

Figure 12. Algal count density (in cells/L) profiles from depth sample data during the 2012 (top panel) and 2013 (bottom panel) sampling periods. SSS activation and deactivation are marked by black vertical lines. Depth samples from 2012 extended only to 6.2 m depth, while 2013 depth samples extended to 8.0 m depth.