

DISSERTATION

RESTORATION OF A SMALL, SHALLOW, EUTROPHIC LAKE BY SUBMERGED
AERATION AND COMPARISON WITH A SIMILAR LAKE

Submitted by

Xiaoju Zhang

Department of Civil and Environmental Engineering

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Doctoral Committee:

Advisor: Larry Roesner

Sybil Sharvelle
Kenneth Carlson
Kimberly Catton
John Stednick

ABSTRACT

RESTORATION OF A SMALL, SHALLOW, EUTROPHIC LAKE BY SUBMERGED AERATION AND COMPARISON WITH A SIMILAR LAKE

A submerged aeration system was introduced to Fossil Creek Lake, a small scale, shallow, eutrophic, very hard water lake, to increase hypolimnetic dissolved oxygen levels, prevent thermal stratification and improve water quality. There has been a long-standing concentration of research effort on aeration systems in deep water lakes, but very little attention has been paid to urban shallow lakes. This research describes the investigation of physical, chemical and biological parameters of the lake before and after aeration. The aeration system gradually and permanently improved the water quality of the entire lake and eliminated many undesirable lake conditions. With the destratification caused by the aeration system, the bottom water dissolved oxygen concentrations increased significantly ($p < 0.05$) from less than 1 mg L^{-1} to above 4 mg L^{-1} . Furthermore, the vertical variations of many water quality parameters have been significantly reduced with the complete mixing of the entire lake, which are: pH, specific conductance, nutrients, chlorophyll a, alkalinity, total suspended solids, sulfate, and hardness. However, limiting nutrient, trophic status and water clarity of the lake were barely affected by the aeration system. Multiple years of continuous aeration on this lake may be necessary to achieve a lower trophic condition. In addition, this research also discusses sources of extremely high sulfate and sulfide levels of Fossil Creek Lake and provides aeration startup recommendations for lakes with high sulfide concentrations.

In this study, the water quality parameters, trophic status and nutrient mass balance of Fossil Creek Lake were compared with Sheldon Lake from April to December in 2010 and 2011, a

turbid, phytoplankton-dominated lake, which is restored by submerged aeration after bottom sediment removal. Monthly mean values of temperature and dissolved oxygen, as well as monthly mean total alkalinity, total inorganic carbon, total organic carbon, orthophosphate and total nitrogen concentrations showed similar distributions in the two lakes. However, the total suspended solids and chlorophyll a concentrations were significantly ($p < 0.05$) higher in Sheldon Lake than those in Fossil Creek Lake. In addition, the Secchi disk transparency of Sheldon Lake rarely exceeded 0.5 m, which was lower than Fossil Creek Lake (> 1 m).

Models relating water clarity to monitored water quality parameters over a period of 2 years are developed in both lakes. In Fossil Creek Lake, the water clarity is not only significantly related with inorganic carbon, nitrogen and orthophosphate, but also hardness, alkalinity, total suspended solids and chlorophyll a concentrations. But in Sheldon Lake, the lake clarity is only significantly related to two parameters, which are total suspended solids and chlorophyll a concentrations.

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CHAPTER 1

LITERATURE REVIEW

1.1 Urban shallow lake characteristics

Urban lakes are usually shallow, highly artificial and often hypertrophic and associated with many human activities (Birch and McCaskie, 1999). A "shallow lake" is usually defined as a permanent standing body of water that is sufficiently shallow to allow light penetration to the bottom sediments to potentially support photosynthesis of aquatic plants over the entire bottom (Wetzel, 2001). Turbidity from abiotic or biotic sources may prevent light from reaching the sediments, but the lakes or ponds are sufficiently shallow for this potential condition to occur.

Shallow lakes tend to accumulate terrestrial organic matter and nutrients. Being of relative small volume, the loading of nutrients per unit volume can be high (Wetzel, 2001). The shallowness makes the lake more vulnerable to variations in weather and nutrient inputs from runoff. Moreover, nutrients in shallow lakes may be significantly increased by sediment resuspension, which makes the eutrophication process more rapid and drastic (Schallenberg and Burns 2004). As a result, the shallow lakes are less responsive to external nutrient loading reductions (Sondergaard et al. 2005a), due to the prevalence of internal loading (Wilander and Persson 2001). All this may have contributed to the fact that the eutrophication control in heavily degraded shallow lakes appear to be less successful than for deep lakes.

Urban shallow lakes provide many different demands, such as fisheries, drinking water, irrigation, boating, recreation and aesthetic landscaping, which make them more vulnerable to changes in water quality than rural, natural lakes. The water quality deterioration manifests itself in changes

in the chemical, physical and biological characteristics including (i) decreased water transparency; (ii) increased pH; (iii) hypolimnetic deoxygenation and the accompanying production of hydrogen sulfide (H_2S) and methane (CH_4); and (iv) release of nitrogen (N), phosphorus (P) and iron (Fe) from lake bottom sediments during periods of anoxia (Kalf 2002, Oberholster, et al., 2007).

Shallow eutrophic lakes are typically dominated by non nitrogen-fixing cyanobacteria such as *Oscillatoria*, whereas deep lakes typically stratify and are generally dominated by nitrogen-fixing cyanobacteria (Havens et al. 2003). Relative to deep lakes, shallow lakes may have higher chlorophyll a or phytoplankton biomass at high total phosphorus concentrations (i.e., $> 100 \mu\text{g L}^{-1}$) (Sondergaard et al. 2005). Effects of macrophytes are generally greater in shallow lakes (Sondergaard et al. 2005). Due to the presence of macrophytes, effects of grazing and trophic cascades are hypothesized to be lower in shallow lake ecosystems.

1.2 Lake stratification and circulation

Lake thermal stratification results naturally due to the change in water's density with temperature (Brönmark et al., 2004). This profile changes from one season to the next and creates a cyclical pattern that is repeated from year to year (Figure 1-1).

In spring, as the ice cover break up, the lake surface water temperature will be around 4°C . Given the similarity in density throughout the water mass there will be little resistance to mixing and only a small amount of wind energy will be needed to mix the whole water column. Thus, the entire water mass will circulate, a process driven by wind energy and temperature induced convection currents, which is called spring lake overturn.

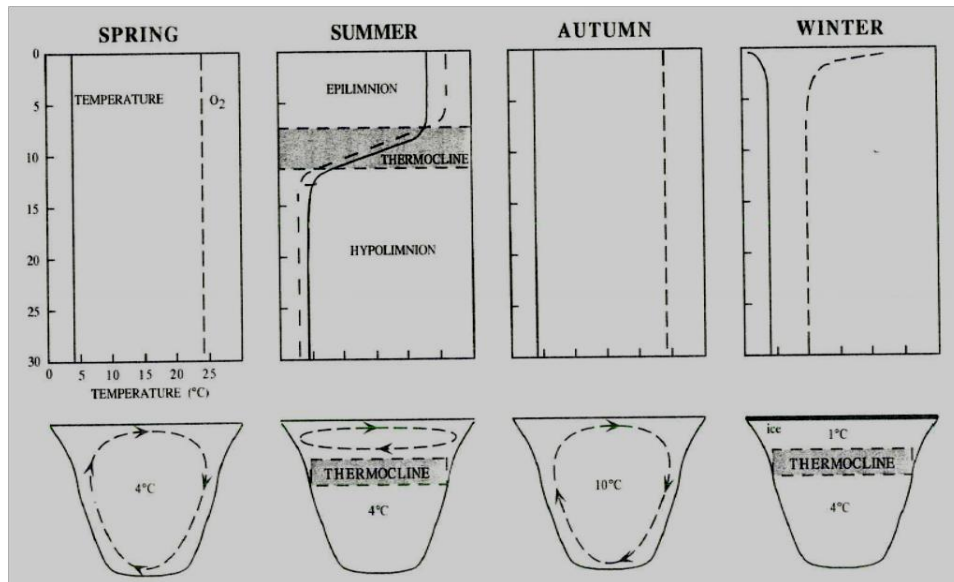


Figure 1-1 Seasonal circulation of water, as well as depth gradients of temperature (solid line) and oxygen (dashed line) in a temperature lake (from Brönmark and Hansson, 2004)

As spring proceeds, the surface water will be heated rapidly. There will be several degrees of difference in temperature between warm top layers (epilimnion) and cooler bottom layers (hypolimnion). The stratum between the two layers, characterized by a steep thermal gradient, is called the thermocline. Stratification restricts water circulation between the epilimnion and hypolimnion, and prevents bottom layers from contacting the atmosphere. Without contact to atmospheric, the hypolimnion, lacking enough light for photosynthesis to occur, is quickly depleted of dissolved oxygen as biological respiration processes consume the remaining dissolved oxygen rapidly. When water rich with nutrients becomes hypoxic, sulfur reducing bacteria will reduce sulfate to hydrogen sulfide gas, creating the "rotten egg" odor often associated with stagnant water bodies or wastewater. Some of the sulfur in the H_2S may combine with iron to form pyrite FeS_2 . Hydrogen sulfide is produced by anaerobic bacteria, and increases in the hypolimnion as long as the bacteria have enough nutrients and low dissolved oxygen (DO).

As autumn approaches and temperatures decrease, the epilimnion will have a net loss of heat, resulting in a decreasing temperature difference between epilimnion and hypolimnion. When the density differences are small enough, the wind energy will cause the whole water column to re-circulate, which is called the autumn turnover.

In winter, when the surface water is cooled below 4 °C, an inverse stratification will establish where the surface water has the lowest temperature. However, the density difference between these layers will be easily disrupted by wind-induced water movements. This pattern will continue until surface ice forms eliminating further wind energy impacts.

1.3 Lake eutrophic characteristics

1.3.1 Lake trophic status

Trophic state is the integrated expression of the nutritional status of a water body (Huber et al., 1982), and can be classified into three major categories, which are oligotrophic, mesotrophic, and eutrophic. The definition of eutrophication adopted by the Organization for Economic Co-Operation and Development (OECD) refers to the changes in water chemical properties triggered by accumulation of excessive nutrients such as nitrogen and phosphorus. It is a joint byproduct of light, heat, and hydrodynamics resulting from a series of biological, chemical, and physical processes (OECD 1982 and Le, C., 2010). Eutrophication is the process of increased productivity of a lake as it ages. Eutrophic lakes are prone to low DO levels in their hypolimnia, as opposed to oligotrophic lakes, which feature a near uniform distribution of oxygen in all water layers. Figure 1-2 shows some common characteristics of eutrophic and oligotrophic lakes, including their pH, temperature, carbon dioxide, and DO profiles.

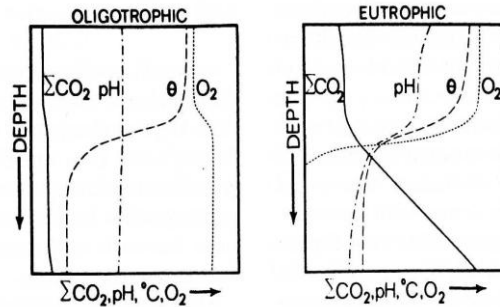


Figure 1-2 Generalized vertical distribution of dissolved oxygen, total CO₂ (ΣCO₂), pH, and temperature (θ =temperature) in stratified lakes of low and high nutrient loading (from Wetzel, 2001)

It is widely accepted that typical trophic parameters are dissolved oxygen, primary productivity, total phosphorus, total nitrogen, chlorophyll-a, transparency and organic matter in sediments (Leach and Herron 1992). Several of these parameters are combined to develop composite indices. To emphasize the continuum nature of a biomass-based trophic state, Carlson (1977) used a numeric scale, dividing the range of algal biomass based on a doubling of Secchi depth, a variable that is affected by algal density. Because it is directly related to lake phosphorus concentration, the Carlson Index was developed for phosphorous limited lakes and reservoirs (Carlson 1977). Work by Kratzer and Brezonik (1981) allows the index to be predicted from nitrogen concentrations as well. Their index could be used especially if nitrogen, rather than phosphorus, is limiting. The Carlson's and Kratzer and Brezonik trophic status index were estimated based on Equations 1-1 to 1-4. Index values and corresponding total phosphorus, chlorophyll-a and transparency are shown graphically in Figure 1-3.

$$\text{TSI (SD)} = 60 - 14.41 \ln \text{SD} \quad \text{Equation 1-1}$$

$$\text{TSI Chl-a)} = 9.81 \ln \text{Chl-a} + 30.6 \quad \text{Equation 1-2}$$

$$\text{TSI (TP)} = 14.42 \ln \text{TP} + 4.15 \quad \text{Equation 1-3}$$

$$\text{TSI (TN)} = 54.45 + 14.43 \ln \text{TN} \quad \text{Equation 1-4}$$

Where

SD: Secchi depth, in unit of m;

Chl-a: Chlorophyll a, in unit of $\mu\text{g L}^{-1}$;

TP: Total phosphorus, in unit of $\mu\text{g L}^{-1}$;
 TN: Total nitrogen, in unit of mg L^{-1} .

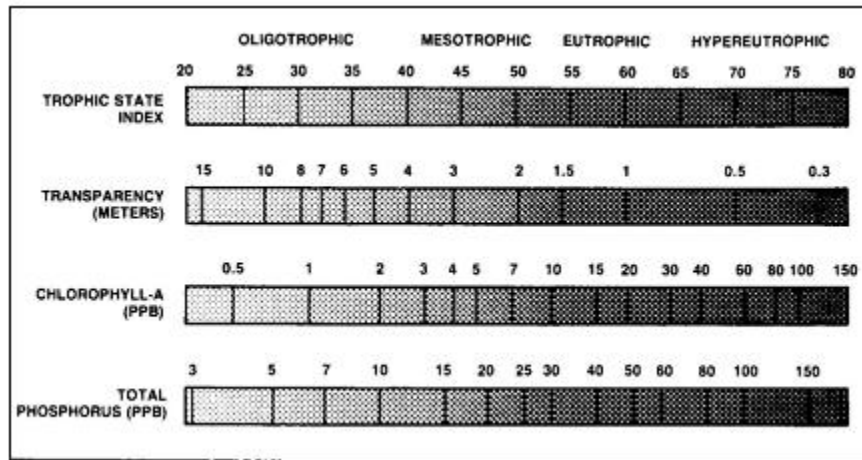


Figure 1-3 Carlson's Trophic Status Index related to transparency chlorophyll a and TP (Adapted from EPA, 1990)

Besides the Carson trophic state indices, other major trophic state classification systems include:

1. U.S. EPA National Eutrophication Survey (U.S. EPA, 1974), and the variables used were total phosphorus (TP), dissolved phosphorus, inorganic nitrogen, Secchi depth, chlorophyll a, and minimum dissolved oxygen (15-DO_{\min}) from more than 200 lakes. The evaluation criteria are summarized in Table 1-1.

2. Vollenweider Phosphorous Loading Index (1975), a model based upon a phosphorous loading concept. Both depth (H) and residence time (τ_w) are considered in relation to the loading of total phosphorus TP (L_p) ($\text{g/m}^2/\text{yr}$) (See Figure 1-4).

3. OECD index (Vollenweider and Kerekes, 1980), which used a statistical approach to quantify the ranges of several variables within each trophic designation. The evaluation criteria are summarized in Table 1-1. Besides the above mainly used models, Table 1-1 also summarizes some other trophic status classification models.

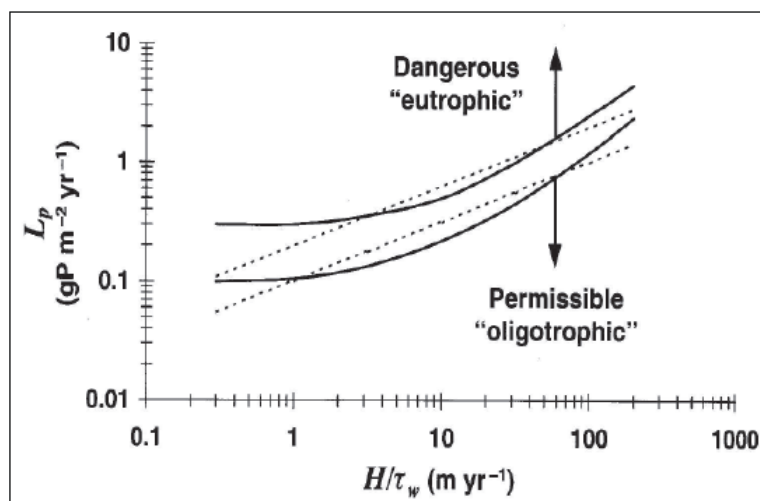


Figure 1-4 Vollenweider's 1975 loading plot (adapted by Chapra, 1997)

Table 1-1 Summary of selected trophic status classification schemes

Variable	Lake Trophic Status				Reference
	Olig	Mes	Eutr	Hypereu	
TP ($\mu\text{g L}^{-1}$)	<10	10-35	35-100	>100	OECD 1982
	4-10	10-20	35-100	>100	CCME 2004
	<10	10-20	>20		Thomann and Mueller 1987
	<5	10-30		>100	Chambers et al. 2001
			30-100	>100	Wetzel 2001
	<10	10-30		>100	Nürnberg 1996
	<10	10-20	>20		U.S. EPA 1974
Chlorophyll a ($\mu\text{g L}^{-1}$)	<2.5	2.5-8	8-25	>25	OECD 1982
	<4	4-10	>10		Thomann and Mueller 1987
	0.3-3	2-15	10-500		Wetzel 2001
	<3.5	3.5-9	9.1-25	>25	Nürnberg 1996
	<7	7-12	>12		U.S. EPA 1974
Secchi Depth (m)	>6	3-6	<1.5		OECD 1982
	>4	2-4	<2		Thomann and Mueller 1987
	>4	2-4	1-2.1	<1	Nürnberg 1996
	>3.7	3.7-2	<2		U.S. EPA 1974
TN ($\mu\text{g L}^{-1}$)	<350	350-650	651-1200	>1200	Nürnberg 1996
				500-15000	Wetzel 2001
Inorganic nitrogen ($\mu\text{g L}^{-1}$)			500-1500	>1500	Wetzel 2001
Organic nitrogen ($\mu\text{g L}^{-1}$)			700-1200	>1200	Wetzel 2001
Hypolimnetic oxygen saturation (%)	>80	10-80	<10		Thomann and Mueller 1987
TOC (mg L^{-1})	<1-3	<1-5	5-30		Wetzel 2001
DOC (mgC L^{-1})	2	3	10		Kalff 2002

Source: Revised from North/South Consultants Inc., 2006.

1.3.2 Limiting nutrients

As early as 1940, the C: N: P molar composition ratio of plankton was accepted to be 106:16:1 (Redfield et al. 1963); this ratio is now referred to as the Redfield ratio. The ratio of N: P is often used to assess nutrient limitation, which is based on the assumption that algal growth is proportional to the quantity of either one or the other nutrient in the water body (Ryding and Rast 1989). Total and bioavailable N: P ratios can provide useful information regarding nutrient limitation. Although there is widespread agreement that high ratios are indicative of P limitation and low ratios indicative of N limitation, there is no widespread agreement on the levels at which changes occur. Whether either nutrient is limiting, growth also depends on absolute concentrations and other factors affecting growth. Although the ratio of N: P changes during the course of the growing season, as a general approximation, the yearly average TN: TP mass ratios were usually examined to describe overall conditions (Ptacnik et al., 2010). Table 1-2 and 1-3 summarizes the indicators values of limiting nutrients to phytoplankton growth.

Table 1-2 Indicator values of nutrient deficiency for phytoplankton growth (Healey and Hendzel 1980 and Hecky et al., 1993). Composition ratios of C: N, N: P, C: P ($\mu\text{mol } \mu\text{mol}^{-1}$), C:Chl a ratios ($\mu\text{mol } \mu\text{g}^{-1}$), and physiological ratio of APA (alkaline phosphatase activity): Chl a ($\mu\text{mol } \mu\text{g}^{-1}$)

Ratio	Deficiency	Degree of nutrient deficiency		
		None	Moderate	Severe
C:N	N	<8.3	8.3-14.6	>14.6
N:P	P	<22		>22
C:P	P	<129	12.9-258	>258
C:Chl a	General	<4.2	4.2-8.3	>8.3
APA: Chl a	P	<0.003	0.003-0.005	>0.005

Table 1-3 Indicators values of N or P limitation to phytoplankton growth
Source: Kosten et al., 2009

Source	Ratio	N limited	P limited	No limitation	Reference
Sediment	TN:TP	<9	>23	9-23	Guildford et al. 2000 and Hecky et al. 1993
	TN:TP	<20	>38	20-38	Sakamoto 1966
Water	DIN:SRP	<13	>50	13-50	Morris and Lewis 1988, and Reynolds 1999

1.3.3 Nutrients management approach

The reduction of in-lake P has been generally considered the key to recovery of eutrophic lake. Would the limiting nutrient in the lake be the appropriate choice for reduction in the inflow to reduce the lake's trophic state? Lewis and Wurtsbaugh (2008) separated the question of which nutrient controls growth in the lake from which nutrient should be controlled in the input to manage lakes, and they concluded for management purposes the answer is P. Evidence from lake responses to manipulations of input nutrients also points to P as the nutrient to reduce trophic state. Algal biomass began to decline only after soluble reactive phosphorus (SRP) reached $<10 \mu\text{g L}^{-1}$ (Sas et al. 1989); therefore, controlling N would provide minimum benefit due to N fixation because N fixation operates to restore the N: P ratio to the Redfield level (Hutchinson 1970). Moreover, reducing the external N supply relative to P may actually promote dominance by N-fixing cyanobacteria (Schindler 1977, Nürnberg 2007, Schindler et al. 2008, Vrede et al. 2009). There may be situations where inflow N reduction would recover eutrophic lakes without resulting in increased dominance by cyanobacteria and N fixation (Schindler et al. 2008). Reduction of inflow N in such circumstances would probably be cost-effective only if background P were naturally high and uncontrollable. However, there is not any published case that demonstrates the effectiveness of N-only reduction, or for the necessity of N reduction in addition to P reduction (Welch, 2009). A lake can be restored to a stable clear-water state by reducing the phosphorus level to between 50 and 100 $\mu\text{g L}^{-1}$, and when the soluble reactive phosphorus (SRP) reached lower than 10 $\mu\text{g L}^{-1}$. The commonly used P reduction techniques include P precipitation and inactivation. Their purpose is to lower the lake's P content by removal of P from the water column (P precipitation) and by retarding P release from lake sediments (P inactivation).

1.4 Aqueous sulfide

1.4.1 Distribution of sulfur in freshwater lakes

Sulfate is the dominant form of dissolved sulfur in water. Nearly all assimilation of sulfur is as sulfate, but during decomposition of organic matter in anoxic condition, sulfur is released mainly as hydrogen sulfide. Ferrous released from sediments reacts vigorously with H_2S to form FeS , which is insoluble in a majority of natural waters. In an anaerobic hypolimnion, the water must be somewhat acidic in order for appreciable H_2S to accumulate. If the water is alkaline, H_2S will accumulate only after most of the Fe^{2+} has been precipitated as FeS . The removal of sulfide by the release of ferrous ions permits an increase in the migration of other metals from the sediments, such as Cu, Zn, and Pb, which form even more insoluble sulfides. In eutrophic lakes with highly organic sediments and low iron concentrations, a larger portion of the sedimentary iron is present as ferrous sulfides (Giblin et al., 1990). Furthermore, organic volatile sulfur compounds (OVS) of biogenic origins are found in quantities throughout aquatic environments sufficient to contribute to the global atmospheric sulfur cycle, including dimethyl sulfide (CH_3SH), by bacterial degradation of dimethylsulfoniopropionate in freshwater ecosystems, methanethiol (CH_3SH), dimethyl disulfide (CH_3SSCH_3), carbonyl sulfide (COS), and carbon disulfide (CS_2). In eutrophic lakes, most of the sulfur in sediment is organic. These lakes with high concentrations of organic S in the sediments are likely sites of much of OVS production.

The reduction of sulfate (SO_4^{2-}) to sulfide (S^{2-}), and oxidation of sulfide to sulfate play a significant role in the modification of conditions for mobilization of phosphate and numerous other nutrients. Most of the sulfur was stored as sulfate and sulfide in water, and as sulfide in sediments (Ingvorsen et al., 1981).

1.4.2 Aqueous sulfide property

Hydrogen sulfide is one of the principal components in the natural sulfur cycle. Its salts and closely related organic sulfides are significant environmental pollutants. Hydrogen sulfide and the mercaptans, CH_3SH and $\text{CH}_3\text{-CH}_2\text{SH}$, have unpleasant odors and very low odor threshold concentrations (Appendix A-1 and AppendixA-2) (Chen and Morris, 1972). According to the Environmental Protection Agency, a maximum acceptable level of H_2S for fish and aquatic life is 0.002 ppm.

Bacteria, fungi, and actinomycetes (a fungus-like bacteria) release hydrogen sulfide during the decomposition of sulfur containing proteins and by the direct reduction of sulfate (SO_4^{2-}). Hydrogen sulfide is also consumed by bacteria found in soil and water that oxidize hydrogen sulfide to elemental sulfur (Figure 1-5). Photosynthetic bacteria can oxidize hydrogen sulfide to sulfur and sulfate in the presence of light and the absence of oxygen (EPA 1993; WHO 1981). Degradation of hydrogen sulfide in the atmosphere can occur through oxidation by oxygen (O_2) and ozone (O_3) to give sulfur dioxide (SO_2), and ultimately, sulfate compounds. Sulfur dioxide and sulfates are eventually removed from the atmosphere through absorption by plants and soils or through precipitation (Hill 1973). Hydrogen sulfide in air can also react with photochemically generated hydroxyl radicals. Hydrogen sulfide has been identified in at least 35 of the 1,689 hazardous waste sites that have been proposed for inclusion on the EPA National Priorities List (NPL) (HazDat, 2006 and HHS, 2006). The frequency of these sites can be seen in Figure 1-6.

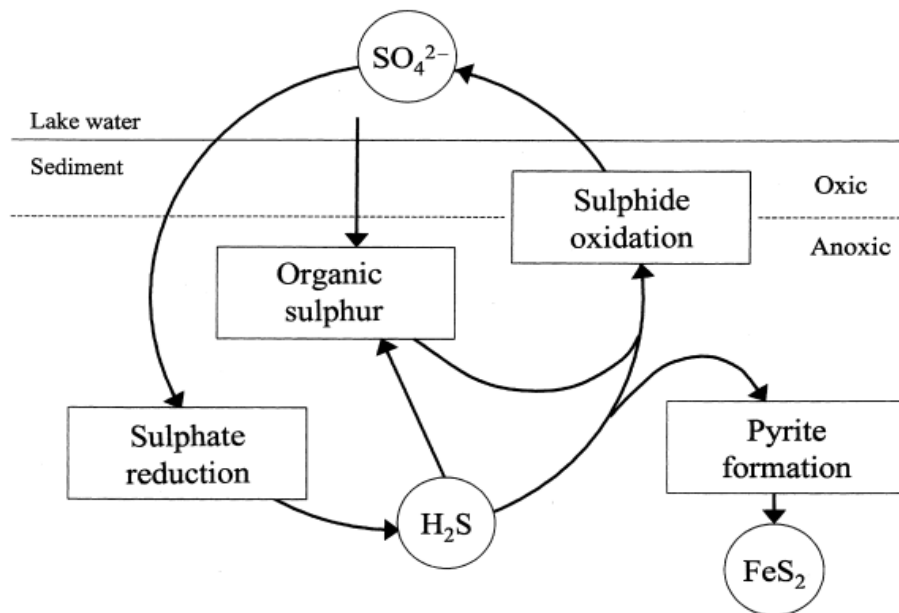


Figure 1-5 Schematic presentation of the sulphur cycle in freshwater sediments (modified from Jørgensen, 1988).
Source: Holmer and Storkholm, 2001

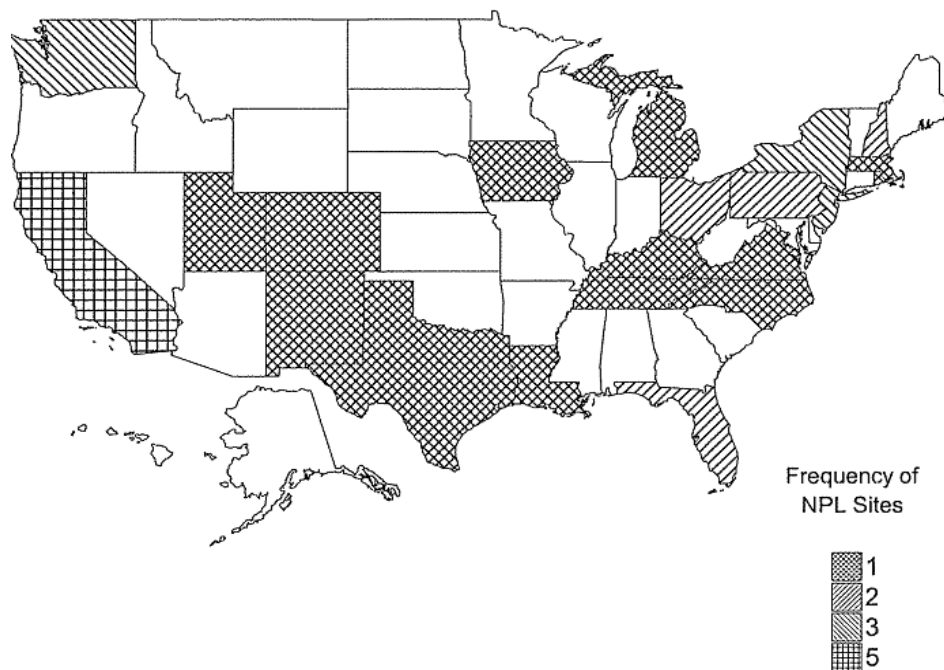
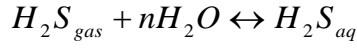


Figure 1-6 Frequency of EPA National Priorities List Sites with Hydrogen Sulfide Contamination
In aqueous solution, hydrogen sulfide is a weak acid, exhibiting two acid dissociation constants.
Derived from HazDat 2006

The first dissociation of hydrogen sulfide yields bisulfide ion (HS^-), and the second yields sulfide ion (S^{2-}), with pKa values for each of these dissociations of 7.04 and 11.96, respectively (O'Neil et al. 2001). The reactions are given below:



Hydrogen sulfide is most toxic at a pH lower than 6.5 when H_2S is the dominant species in aqueous systems. At a pH of 7.0, the ratio of the concentration of aqueous hydrogen sulfide to bisulfate ion is approximately 1-to-1. As the pH increases above 7.0, the ratio of the concentration of bisulfide ion to aqueous hydrogen sulfide increases. Only above pH 12 will the concentration of sulfide ion become significant, which is above 50% (Figure 1-7). In contrast to pH, at increased temperatures, very low levels of H_2S can be toxic. In July and August, sulfide concentrations, which are relatively harmless during the winter, can cause significant damage and mortalities of fish within a few minutes. Hydrogen sulfide can be exasperated in warm months, as rapid organic decomposition occurs during periods with low DO levels, higher water temperatures, and large pH fluctuations.

Hydrogen sulfide has been shown to sorb to various soils (Cihacek and Bremner 1993; Smith et al. 1973). Several species of soil, aquatic, and marine microorganisms oxidize hydrogen sulfide to elemental sulfur (Jørgensen 1982). Hydrogen sulfide can also form insoluble sulfide salts with various metals (i.e., copper, zinc, nickel, and iron) that may be present in soils (Pouliquen et al. 1989).

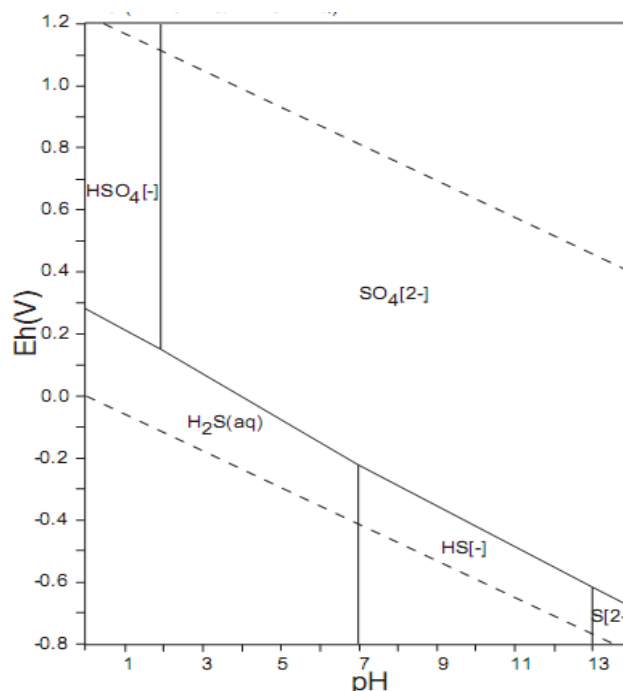


Figure 1-7 Eh-pH diagrams of the system S-O-H at 25 °C, 298.15K, 10^5 Pa

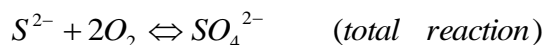
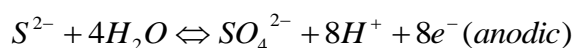
Accurate measurements of hydrogen sulfide water levels are usually complicated by the presence of other sulfide compounds. At $\text{pH} \geq 7$, hydrogen sulfide is significantly dissociated, and the exact source of sulfide would not necessarily be known. A method of determining sulfide concentration is by first transforming it to nondissociated hydrogen sulfide and then measuring the atomic absorption of the product.

1.4.3 Kinetics of oxidation of aqueous sulfide

Sulfur oxidizing bacteria are commonly differentiated into two groups: the chemosynthetic (colorless) sulfur-oxidizing bacteria, and the photosynthetic (colored) sulfur bacteria, anaerobes that can be divided conveniently into the green sulfur bacteria (Chlorobacteriaceae) and the purple sulfur bacteria (Thiorhodaceae). The purple sulfur bacteria require light energy for the oxidation of H_2S and other reduced sulfur compounds, especially thiosulfate, to sulfate in the photosynthetic reduction of CO_2 . Some of the purple sulfur bacteria are able to grow autotrophically, with thiosulfate as the electro donor. Many purple bacteria can utilize hydrogen

as the only electron acceptor, simultaneously with assimilatory sulfate reduction system. In anaerobic environments where iron is abundant, phototrophic sulfur bacteria can use FeS as an electron donor as well as H₂S. Thus, under conditions where light supports phototrophic sulfur bacteria in proximity to combined sources of reduced iron and sulfide, the sulfide is oxidized either directly or indirectly through FeS to sulfate at rates adequate to prevent iron sulfide from forming in the water column.

For proper control and minimizing of the obnoxious odors associated with H₂S and the mercaptans, oxygen oxidation is the mechanism by which sulfide is removed naturally from water. Hydrogen sulfide evaporates from surface waters and is not likely to persist in highly oxygenated waters (HHS, 2006). At low concentrations the sulfide can be eliminated by providing adequate aeration in order to decrease dead or stagnant areas in lake. Keeping ponds aerated will help decrease H₂S levels, but sulfide concentrations above 70-200mg L⁻¹ are recommended to be removed chemically, because of the inhibition of bacterial metabolism. Sulfide can be oxidized to any higher level. The half reaction of sulfide is as follows:



The rate of the oxidation reaction is affected by a number of factors like temperature, pH, sulfide ion concentration, dissolved oxygen concentration, neutral salts, bacterial action, and organic species. Several studies indicate that high rates of sulfide oxidation can accompany high rates of sulfate reduction. Sulfide oxidation can occur nearly as rapidly as sulfate reduction. Importantly,

the formation of organic sulfur in sediments consumes H^+ ions via sulfate reduction in acidified lakes.

1.5 Lake ecosystem

Lakes are complex ecosystems with many species of animals and plants interacting with each other and their environment. External factors such as sunshine, wind, air temperature and water inflows combine with internal forces such as evaporation rates, currents, nutrient release from sediments, nutrient uptake by algae, and plant-animal interactions to produce an intricate web of relationships. The interrelationships of the water chemical, physical and biological properties in many lakes vary seasonally throughout the year. An especially dynamic period for lakes occurs during the "growing season" of mid-spring through early autumn when lake-dwelling organisms increase their activities. By contrast, the lake waters cool temperatures and low light levels through the late fall through early spring keep biological activities at a minimum.

Submerged macrophytes are a natural and essential part of the lake, and play a key role in the functioning of shallow lake ecosystems. They provide refuge for herbivorous zooplankton (Timms and Moss, 1984), produce oxygen, enhance water clarity by reducing the resuspension of bottom sediment by wind and benthivorous fish (Meyer et al., 1990; Scheffer 1992), and suppress algal growth by competing for available nutrients. Waterfowl, shore birds and aquatic mammals use plants to forage on, and as nesting materials and cover. The loss of submerged macrophytes can cause a reduction in the diversity of keystone macroinvertebrate and zooplankton species, and lead to an alternative stable turbid state in shallow eutrophic lakes (Oberholster et al., 2007). Return of aquatic vegetation is imperative for a successful restoration of turbid lakes.

Phytoplankton components of eutrophic, shallow lakes have frequently undergone dynamic changes in composition and abundance with important consequences for lake functioning and stability. Seasonal succession of phytoplankton species in lakes reflects changes in physical and chemical characteristics of the water body. There are two main factors describe below that could induce the rapid transition to phytoplankton-dominated conditions where submerged macrophytes are suppressed or eliminated.

1. Competition for light. The development of phytoplankton, especially small algal and cyanobacterial species with a high capacity for light absorption, early in the season in nutrient-rich shallow waters is able to build up populations dense enough to attenuate light markedly and suppress submerged plant propagules growing.

2. Competition for nutrients. Small-celled algae may compete more effectively at low concentrations of available nitrogen in water and sediment. Due to photosynthesis, the pH may rise to values where carbon dioxide constitutes only a small proportion of the inorganic carbon available. Algae associated with high pH values (above about 8.4) either can use bicarbonate or can extract CO₂ at very low concentrations (Talling, 1976; and Shapiro 1997). The values of pH may be buffered at high alkalinities so that they do not rise extremely high, and bicarbonate is in plentiful supply for those plants that can use it. However, not all aquatic plants can use bicarbonate, and the speed of diffusion into bulky tissues is low. Perhaps more than half of the species found in high alkalinity waters can take up bicarbonate (Hutchinson, 1975; and Balls et al., 1989). However, there is no simple explanation of the switch from aquatic plant to phytoplankton dominance. All that is firmly established is a definite association with extreme eutrophication (Hartmann, 1977; Osborne and Moss, 1977 and Moss, 1979, 1980).

If a large proportion of small zooplanktivorous fish dominates the fish communities, the grazing pressure on phytoplankton by large cladoceran and copepod zooplankton is reduced. As a result, the grazing mortality of phytoplankton is reduced. Figure 1-8 presents a generalized model of a simple phytoplankton–zooplankton–planktivore–piscivore system and some possible outcomes due to trophic interactions. For example, intense stocking of piscivore can reduce planktivore biomass. Reduced planktivore biomass can result in a reduced feeding pressure on zooplankton, and therefore cascading to the phytoplankton level via increased grazer control by zooplankton. This in turn increases water clarity, and submerged macrophytes can grow. Another possible scenario might be that intense fishing on predation might allow planktivore biomass to increase. The result would be a decrease in zooplankton biomass which would result in an increase in phytoplankton abundance. Increased phytoplankton biomass may, in fact, have a direct impact on piscivores by lowered DO concentrations and loss of spawning habitat.

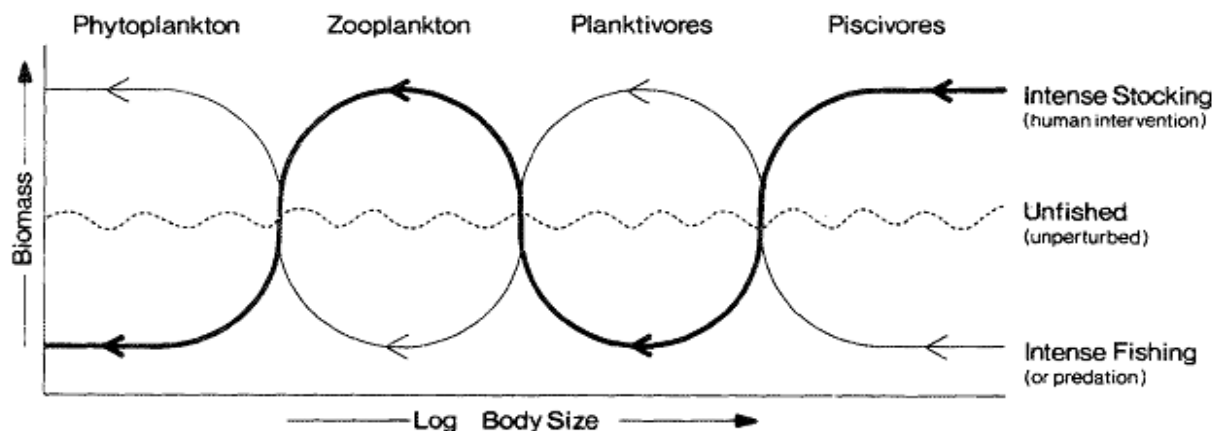


Figure 1-8 Generalized model of a simple phytoplankton, zooplankton, planktivore, piscivore system and some possible outcomes due to interrelationships (adapted from Christie et al. 1987b and Hartig et al., 1991)

1.6 Restoration data from other lakes

There has been a substantial amount of research on the restoration of lakes by destratified aeration, hypolimnetic aeration, nutrient control, hypolimnetic withdrawal, etc. This section summarized some water quality parameters before and after restoration

Table 1-4 Location, morphometry and restoration methods of lakes

References: 1. Qiu, D et al., 2001; 2. Lawson, R. and Anderson, M., 2007; 3. Jeppesen, E et al., 2007; 4. Jeppesen, E et al., 2007; 5. Oberholster, J. P. et al., 2007; 6. Cowell, B.C. et al., 1987; 7. Fast, A.W. et al., 1973; 8. Fast, A.W. et al., 1973; 9. Welch, E.B., 2009; 10. Denkenberger, J.S., 2007; 11. Denkenberger, J.S., 2007; 12. Labaugh, J.W., 1980; 13. Solter, R.A., et al., 1994; 14. Steinberg et al. 1982; 15. Sosiak and Trew 1996; Sosiak 2002.

ID	Lake	Location	Area, km ²	Mean depth, m	Restoration method
1	Shuiguohu	China		7.6	Restoration of macrophytes
2	Lake Elsinore	California	14.7	3.3	Axial flow pumps
3	Lake Arreso	Denmark	40	3.1	Nutrient control and biomanipulation
4	Lake Paranoa	Brazil	38	13	Nutrient control
5	Sheldon Lake	Colorado	0.07	2	Dredging and artificial mixing
6	Lake Brooker	Florida	0.11	4	Destratification aeration
7	Section Four Lake	Michigan	0.01		Destratificaiton aeration
8	Hemlock Lake	Michigan	0.02		Hypolimnetic aeration
9	Moses Lake	Washington	29	5.6	Nutrient control
10	Otisco Lake	New York	7.6	10.2	Nutrient control
11	Onondaga Lake	New York	11.7	12	Nutrient control
12	Spruce Knob Lake	Virginia	0.11	2.1	Destratificaiton aeration
13	Medical Lake	Washington	0.63		Hypolimnetic aeration
14	Obinger Lake	Germany	0.31	7	hypolimnetic withdrawal
15	Pine Lake	Canada	0.44	5	hypolimnetic withdrawal

Table 1-5 Water quality parameters before and after restoration

Lake ID	1		2		3		4		5	
Water quality parameters	before	after	before	after	before	after	before	after	before	after
TN (mg L ⁻¹)	2.47	1.66			2.85	2.15	0.08	0.04	1.81	2.21
Ammonia (mg L ⁻¹)	2.21	0.27								
Nitrite+Nitrate (mg L ⁻¹)	0.13	0.11							2.28	2.87
TP (mg L ⁻¹)	0.54	0.13			0.45	0.28	0.1	0.06		
RP (mg L ⁻¹)	0.28	0.04			0.15	0.08	0.08	0.05		
Secchi (m)	0.38	1.12	0.25	1.31	0.35	0.45			0.51	0.61
Chlorophyll a (µg L ⁻¹)					300	120			72	58

Table 1-5 continued Water quality parameters before and after restoration

Lake ID	6		7		8		9		10	
Water quality parameters	before	after	before	after	before	after	before	after	before	after
TN (mg L ⁻¹)	1.63	1.15								
Ammonia (mg L ⁻¹)	0.41	0.29								
Nitrite+Nitrate (mg L ⁻¹)	0.007	0.002					0.48	0.25		
TP (mg L ⁻¹)	0.24	0.23					0.15	0.02		
RP (mg L ⁻¹)	0.034	0.046					0.04	0.02		
Secchi (m)	0.91	1.06	5.5	3.7	11	8				
Chlorophyll a (µg L ⁻¹)	53.11	51.78					60	11	8.6	4.3
Sulfate	26.53	22.74								

Table 1-5 continued Water quality parameters before and after restoration

Lake ID	11		12		13		14		15	
Water quality parameters	before	after	before	after	before	after	before	after	before	after
TN (mg L ⁻¹)										
Ammonia (mg L ⁻¹)					0.5	0.4				
Nitrite+Nitrate (mg L ⁻¹)			0.15	0.075						
TP (mg L ⁻¹)	0.4	0.02	0.28	0.1	0.13	0.04	0.12	0.07	0.27	0.2
RP (mg L ⁻¹)			0.05	0.002						
Secchi (m)							1.2	1.3	2.2	2.5
Chlorophyll a (µg L ⁻¹)	21.6	18.8			5.13	5.87	34.5	33.3	14.5	14

1.7 Research objectives and approach

A lake is a unique body of water with physical, chemical, and biological characteristics, reflecting the characteristics of the surrounding watershed and the climate, as well as the shape and volume of the lake basin. Developing a management plan to protect and maintain healthy lake systems requires a thorough analysis of the lake impairment condition, trophic status, nutrient sources, transport routes, and ecosystem responses. In addition to such analysis, a methodology to aggregate scientific knowledge in such a way as to make it useful for management application is necessary. The combination of these two aspects, analysis and synthesis, has been seen as the distinctive feature of the restoration of shallow eutrophic lakes (Somlyódy, 1986).

Lake restoration is defined as the manipulation of a lake ecosystem to affect an in-lake improvement of degraded or undesirable conditions. The goal of lake restoration is to return, as far as is possible, the lake to its original conditions, including 1) healthy functioning of the lake ecosystems and aesthetically pleasing appearance; 2) no objectionable odor problem; 3) minimal algae blooms, for example minimal duration, frequency, and intensity of algae bloom.

Before starting the research experiments and based on both our preliminary results and results from other studies in the literature it has been hypothesized that:

1. Placing aerators in the deepest locations of the lake will result in complete mixing due to water movement “downhill”, and will eliminate thermal stratification and improve near bottom water dissolved oxygen concentrations.
2. Submerged aeration will gradually decrease lake nutrient concentration and may change the lake trophic status in Fossil Creek Lake.
3. Chlorophyll a concentrations in Fossil Creek Lake will be suppressed by submerged aeration.
4. Submerged aeration in Fossil Creek Lake and submerged aeration after dredging in Sheldon Lake may have different effects on nutrient levels, phytoplankton biomass and clarity.
5. The water clarity in both lakes is not only highly related to nitrogen, phosphorus and chlorophyll a, but also hardness and alkalinity.

The general objectives of this research are to study the effects of restoration methods on water quality, trophic dynamics, and aquatic plants, which in turn affect the functioning of urban shallow lakes. The approach followed is the combination of both controlled experiments on different scales (laboratory and outdoor field experiments) and comparative field studies with a reference lake under different restoration methods: artificial aeration versus artificial aeration after bottom sediment removal.

The project aims to address the following main points:

1. Determine how the submerged aeration system affects the lake water quality and trophic status in Fossil Creek Lake; identify the strengths and weaknesses of this method; and determine whether the aeration system causes potentially detrimental effects at the start of aeration and identify approaches to minimize and prevent startup problems.
2. Compare how two different lake restoration methods affect lake water quality, trophic status and nutrient mass balance in two similar size urban shallow lakes: bottom sediment removal and complete aeration in Sheldon Lake versus submerged aeration in Fossil Creek Lake.
3. Identify additional future lake management and restoration measures when the existing methods cannot reach the ideal nutrient control targets.
4. Identify the relationships between water quality parameters and water clarity in each lake; develop water clarity-water quality parameters models; and compare the water quality parameters which have significant relationships with water clarity in both lakes.

Based on results obtained from this study, one scientific manuscript has been submitted to a peer reviewed journals, and two papers are in preparation. Chapter 2 represents the first manuscript based on 4 years of water quality data monitored in Fossil Creek Lake before and during submerged aeration in 2006, 2009, 2010, and 2011. This manuscript has been submitted to *Lake and Reservoir Management Journal*. Chapter 3 provides a detailed comparison of water quality parameters obtained from Fossil Creek Lake and Sheldon Lake from 2010 and 2011. This chapter has been submitted to *Journal of Water Resource and Protection*. Chapter 4 presents research of water quality parameters that affect lake clarity in Fossil Creek Lake and Sheldon

Lake. This chapter is in preparation and for submission to *Lake and Reservoir Management Journal*. Chapter 5 includes the conclusions and suggests future work that can be undertaken as an extension of this study.

CHAPTER 2

ASSESSMENT OF WATER QUALITY IMPROVEMENTS OF A SMALL, SHALLOW EUTROPHIC LAKE BY SUBMERGED AERATION

2.1 Introduction

There has been a substantial amount of research on the restoration of lakes by aeration or oxygenation (Rybak, 1985; Cooke et al. 2005), a technique intended to eliminate thermal stratification and improve bottom oxygen levels. For small lakes, a high accumulation of organic matter and nutrients in the limited volume along with the prevalence of internal loading can accelerate the lake eutrophication process, which may account for the perception that the rehabilitation of heavily degraded small shallow lakes appears to be more difficult than that of deep lakes. Dunst et al. (1974) listed 123 lakes where aeration had been applied to improve water quality and oxygen conditions. Labaugh (1980) evaluated the aeration system in Spruce Knob (0.11 km^2), West Virginia, and concluded that total inorganic carbon, total alkalinity, nitrite, soluble reactive phosphorus, total phosphorus, pH and chlorophyll concentrations were reduced after aeration during the summer at lake bottom. But the trophic condition and the phytoplankton biomass was unaffected by aeration. The artificial mixing of two Oklahoma lakes: Arbuckle Lake (9.51 km^2) and Ham's Lake (0.41 km^2), decreased the ammonia and hydrogen sulfide and improved dissolved oxygen in the hypolimnion, while other water quality factors did not change. However, the water clarity of Ham's Lake was decreased by the aeration, and algal biomass was increased (Toetz, 1981). Oxygenation and aeration in Lake Wilcox (0.56 km^2) in Ontario, Canada, distributed numerous filaments throughout the water column and to the water surface (Nürnberg et al., 2003). The whole lake aeration in Lake Brooker, Florida (0.11 km^2), eliminated oxygen depletion, blue-green algal blooms, and low benthic diversity, but it did not change the trophic

state very much (Cowell, 1987). Similarly, the aeration of Lake Särkinen, Finland, had a positive effect, but less effect on reducing the nutrient load (Sandman et al., 1990). In Newman Lake (5.15 km²), whole-lake alum treatment followed by hypolimnetic oxygenation reduced total phosphorus concentrations, in contrast to pre-restoration years when frequent blue-green blooms dominated in summer. An early summer clear water phase following spring diatoms blooms has been apparent for the past 8 years after restoration (Moore and Christensen, 2009). In stratified, mesotrophic hardwater lakes, with phosphorus levels above 20 µg L⁻¹, blooms occurred in several cases after treating some lakes with whole-lake aeration and mixing, (Nürnberg et al, 2003). Engstrom and Wright (2002) evaluated long-term effects of aeration-induced circulation on sediment accumulation and composition in 10 aerated and non-aerated small shallow lakes (0.05-0.53 km²) in Minnesota and concluded that aeration-induced circulation does not enhance the removal of organic sediments. Ashley et.al (1987) tested the empirical hypolimnetic aeration sizing method by comparing St. Mary Lake (1.82 km²) with Glen Lake (0.16 km²) in British Columbia and provided recommendations to avoid undersizing future aeration installations. Miller and Mackay (2003) conducted experiments in lakes (0.007-0.51km²) of Alberta, Canada to optimize artificial aeration size and prevent winterkill; optimal sizing occurred with 0.15-0.23 kW ha⁻¹ for air injection and 0.06-0.1 kW ha⁻¹ for surface aeration. Ott and Koiv (2005) studied the ecological status and changes in the past eight decades in Lake Verevi in Estonia (0.13 km²), and they proposed restoring the lake by phosphorus precipitation and hypolimnetic aeration.

The research described above for shallow lakes with surface area smaller than 0.05 km² is quite limited, however. Despite many successful studies, there is a need for detailed case studies on a variety of lake types, geographic locations (Cowell, 1987), and long-term physical, chemical,

and biological responses to aeration (Jeppesen et al. 2005, Nürnberg 2007, Schindler et al. 2008, Moore and Christensen, 2009), which will be vitally important to advance lake water quality restoration and management (Osgood 2007). Moreover, there are many unexplained exceptions to findings in previous studies which may be the result of short duration experiments, lack of pre-aeration data, or lack of vertical and spatial distribution data of the entire lake. In an attempt to address some of these limitations, we have studied Fossil Creek Lake, treating it with submerged aeration and documenting the vertical and spatial distributions both before and after the aeration.

The objective was to conduct a detailed study of the effects of an aeration system on the water quality of Fossil Creek Lake (0.048 km²), and to determine the potential of aeration for management and restoration of small, shallow, eutrophic lakes. We first conducted a one and a half year analysis of physical, chemical and biological properties of the lake prior to installation of the aeration. We then performed a two and one half year analysis of the same parameters during aeration.

2.2 Description of the study area

Fossil Creek Lake, a very hard water lake located in south Fort Collins, Colorado (40 °30' 21.81" N, 105 °03' 49.66" W) is the remnant of the old Portner Reservoir, constructed more than 100 years ago. During construction of the park in the late 1990s, the reservoir remnant was divided into three connected lakes by constructing causeways between the East Lake and the Big Lake and between the Big Lake and the West Lake (Figure 2-1). The lakes are now maintained as an aesthetic and recreational fishing amenity to park visitors. The lakes, at an elevation of 1,516 meters, have a combined a surface area of 5.8 ha ; the West Lake is about 1.0 ha, the Big Lake is

approximately 4.2 ha, and the East Lake is approximately 0.6 ha. The annual average rainfall within the watershed of the lake is approximately 45.72 cm; however less than 20 ha of watershed drain to the lakes. The major water supply for the lake is a diversion pipe from Fossil Creek, which delivers an annual average flow of $0.02 \text{ m}^3 \text{ s}^{-1}$ from April to October each year, flowing through a free water surface wetland, and then entering the West Lake and Big Lake at the same time through a flow splitter. Cutting through Pierre shale, a sulfur rich marine deposit, Fossil Creek has high sulfate content at around 1000 mg L^{-1} , which is much higher than that of other lakes in North America (lower than 40 mg L^{-1}). In addition to the direct inflow from the wetland, the Big Lake receives overflow from the West Lake at two points shown in Figure 2-1. The East Lake is fed by overflow from the Big Lake.



Figure 2-1 Map of Fossil Creek Park Lake, CO

The Big Lake mean depth is 2 m and the maximum depth is 5.5 m, with total volume of around 68900 m^3 , and hydraulic detention time of 40 days. About 90% of the lake bottom is covered by submerged macrophytes and the lake has comparatively high transparency most times of a year except when the lake overturns. Water from the East Lake is used to irrigate lawns during

summer time, resulting in the shortest hydraulic residence time. The high sulfate content, the accumulation of 100 years of organic matter in bottom sediments coupled with summer and winter thermal stratification results in extremely low redox potentials in the bottom sediments and high concentrations of reduced sulfur (S^{2-}). The presence of H_2S (g), particularly during lake turnover events has resulted in numerous odor complaints from park visitors and residents whose homes abut the lake. In addition, fish kills occur during the late summer and through the winter as a result of low DO and high H_2S concentrations. The Big Lake shown in Figure 2-1 was monitored and aerated and refers to Fossil Creek Lake in this paper.

2.3 Methods

A submerged aeration system with continuous duty was installed in Fossil Creek Lake on August 22nd, 2009. This system is powered by two 248 W shore-installed air compressors, which were each set at 0.25 m³ per minute. Releasing air at the deepest locations, rather than uniformly distributed over the lake area may prove to be a very economical technique of removing high concentrations of hydrogen sulfide from a small lake (Lorenzen and Fast, 1977). Moreover, when placing the aerators in the deeper pockets of the lake, density differences will cause the heavier bottom water to “flow downhill” toward the aerators, increasing the effective circulation cells of the individual aerators. Each compressor was connected to three self-weighted rubber pressure pipes (0.5” I.D. with 0.25” side walls) extending to the diffusers on the bottom of the deepest locations of the lake. When compressed air is released through the diffuser at the lake bottom, air begins to migrate towards the surface, drawing oxygen-depleted water from the bottom, oxygenating it and transporting it to the surface. Meanwhile the aerator draws cooler water toward it, creating a permanent circulation pattern. The bubbles generated by this system can move 30 m³ of water from the bottom to the surface per hour. At this rate it is possible to

completely circulate the entire lake in around 67 days. Bathymetry of the lake and the location of the aerators are shown in Figure 2-2.

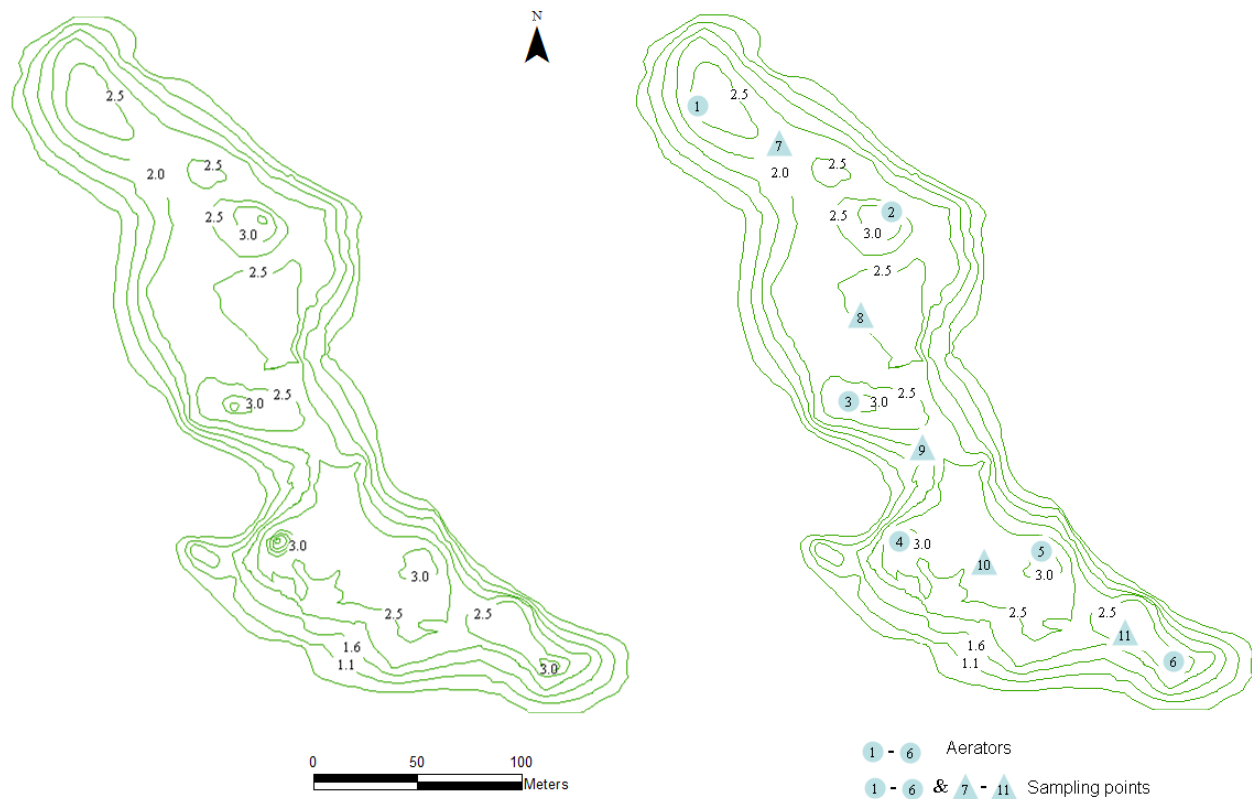


Figure 2-2 Bathymetric map of Fossil Creek Lake with aerators and sampling locations

Fossil Creek Lake sampling and analyses were conducted from May 9th to October 20th in 2006 on a biweekly basis to determine the pre-aeration conditions. Monitoring focused on four primary water quality parameters: temperature (T), dissolved oxygen (DO), pH, and specific conductance (SPC) in locations 1 through 6 in 0.3 m intervals from surface to bottom. These parameters were determined with a in field test model-YSI Professional Plus model. In the pre-aeration months of 2009 (May 4th -August 22nd), and the post-aeration years 2010 and 2011, the lake was sampled on a biweekly basis to monitor changes during aeration. Besides the same water quality parameters that were monitored in 2006, total organic carbon (TOC), total nitrogen (TN), total phosphorus (TP) and orthophosphate (RP), sulfide, sulfate, Secchi disk depth,

chlorophyll a, total suspended solids (TSS), alkalinity, and hardness were measured. For the post-aeration months in 2009 (August 24th to November 20th), only T, DO, pH, and SPC at stations 1 through 11 at surface (0.3 m depth), middle and bottom were monitored. Table 2-1 summarizes the sampling details in the past years.

Table 2-1 Sampling information before and after aeration

Aeration status	Year	No. of sampling station	In field parameter	Lab analysis parameter
Pre	2006	6	T, DO, pH, SPC	NA
Pre	05/2009-08/2009	11	T, DO, pH, SPC, Secchi depth, H ₂ S	TOC, TN, TP, RP, TSS, SO ₄ ²⁻ , H ₂ S, alkalinity and hardness
Post	08/2009-11/2009	11	T, DO, pH, SPC	NA
Post	2010	11	T, DO, pH, SPC, Secchi depth, H ₂ S	TOC, TN, TP, RP, TSS, SO ₄ ²⁻ , H ₂ S, alkalinity and hardness
Post	2011	11	T, DO, pH, SPC, Secchi depth, H ₂ S	TOC, TN, TP, RP, TSS, SO ₄ ²⁻ , H ₂ S, alkalinity and hardness

Water quality parameter for lab analyses, such as TOC, TN, RP, TP, sulfate, chlorophyll a, etc were sampled and composited across the 11 sampling sites at each depth (surface and bottom) separately. The location of the sampling stations is shown in the right hand side of Figure 2. All water samples were placed on ice for transport to the laboratory, and were preserved in the field following EPA approved protocols. Standard analytical methods (Eaton et al., 1998) were used for all analyses, and quality assurance samples (blanks, duplicate analyses, and standards) were analyzed at a rate of 10%.

Total phosphorus concentrations were determined using the acid persulfate digestion method; orthophosphate was measured using the ascorbic acid method; nitrate + nitrite was measured by the cadmium reduction method; ammonia was tested by the salicylate method; sulfate was determined by USEPA method 375.4; hydrogen sulfide was tested by the methylene blue method

in the laboratory and with the color chart/effervescence of H₂S method for the field test (Hach company hydrogen sulfide test kit, Model HS-C). A Hach DR/2500 Spectrophotometer, a UV absorption spectrophotometer, was used to determine the above parameters after colorimetric reactions. Hardness was determined by the EDTA titration method; alkalinity was determined by the phenolphthalein titration method; total suspended solids were determined by the gravimetric method. A Shimadzu TOC-VCS/CP and ASI-V auto sampler analyzer, a high-temperature combustion measurement method was used for the total organic carbon analysis; total nitrogen was measured by Shimadzu's TNM-1 model with combustion oxidation and chemiluminescence detection. Chlorophyll a determinations were made using the method of Arar and Collins (1997). Chlorophyll-containing phytoplankton in a measured volume of sample water was concentrated by filtering it at low vacuum through a glass fiber filter (Whatman GF/C). Pigments were extracted from the phytoplankton in 90% acetone by a tissue grinder and allowed to steep for a minimum of 2 hr to ensure thorough extraction of the chlorophyll a. The filter slurry was centrifuged at 675 g for 15 min (or at 1000 g for 5 min) to clarify the solution. An aliquot of the supernatant was transferred to a glass cuvette and fluorescence is measured before and after acidification to 0.003 N HCl with 0.1 N HCl. Most chemical data were assumed to be normally distributed. One way analysis of variance (ANOVA's) was used to evaluate the differences in water quality parameters between sample depths before and after aeration. All statistical tests were performed using Matlab 2009, and statistical significance was considered to be $p < 0.05$.

2.4 Results

2.4.1 Vertical distributions of physiochemical parameters before aeration

During the months before the aeration system was installed, the temperature, dissolved oxygen, pH and specific conductance showed significant vertical stratification ($p < 0.05$) from June to

August. All shallow stations (less than 3 m deep) showed no statistical differences in T, DO, pH and SPC to sampling Station 1, and Station 2 with depths greater than 3 meters showed results no statistical differences to Station 4; therefore only Stations 1 and 4 results are shown in this section. The depth-time diagram of temperature and dissolved oxygen in both shallow and deep sampling locations of Fossil Creek Lake in 2006 is presented in Figure 2-3. The results of sampling Station 1 with a maximum depth of 2.8 meters are shown in Figure 2-3(a) and (c), while Figure 2-3(b) and (d) show sampling results at Station 4, with a maximum depth of 5.5m, which is the deepest region of the lake. Summer stratification initiated in June, and lasted until August with maximum surface and bottom temperature differences of 4.3 °C at Station 1, and 7.3 °C at Station 4. Maximum epilimnion temperatures were observed in July, which were above 25 °C. Surface temperatures began to drop in August with surface mixing to 1.8 m at Station 1 and to 3.2 m at Station 4. Complete overturn occurred in September. In May and September, surface and bottom temperature differences were less than 1.5 °C in both Stations, which indicated the lake was relatively well mixed by spring and autumnal overturn. Winter stratification initiated from October, and the temperature in the hypolimnion became higher than the epilimnion in both the deep and shallow stations. Although the temperatures are still 9-11 °C in October, the apparent inverse stratification on 10/20 in must be due to chemical stratification, since 11 °C bottom water based only on temperature would be positively buoyant relative to 9 °C overlying water. The strong salinity gradient (Figure 2-4 (d)) plays a role in the overall resistance to mixing.

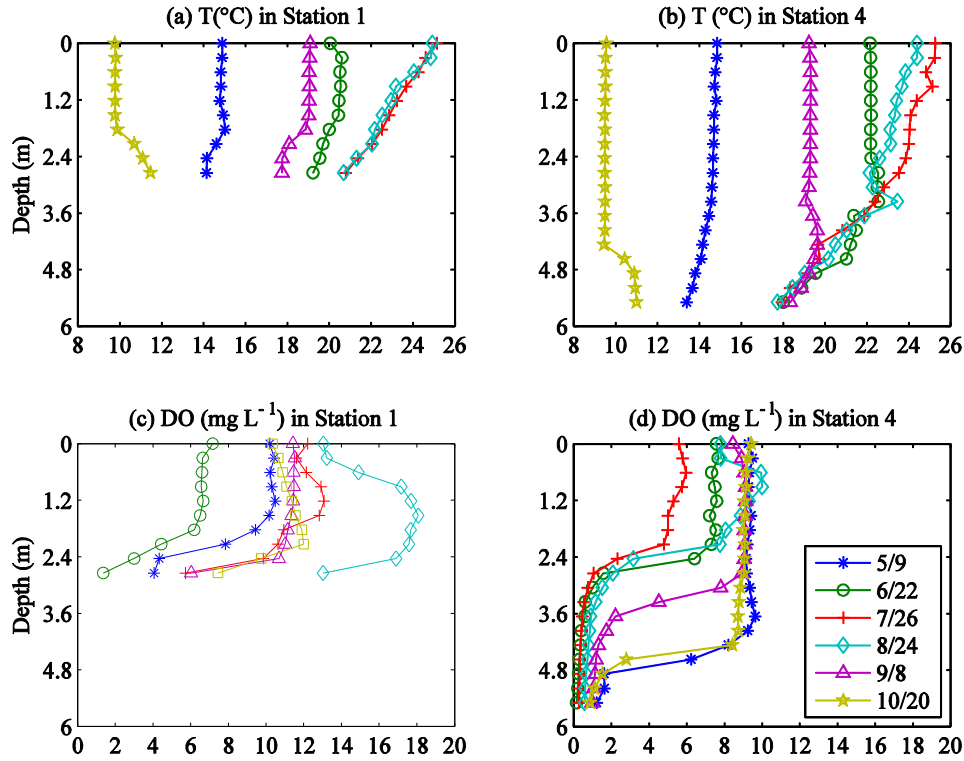


Figure 2-3 Vertical distributions of temperature and dissolved oxygen in Stations 1 and 4

The behavior of DO profiles at Stations 1 and 4 is similar to that of temperature as expected. At the water surface, DO ranged from 7.2 to 18.1 mg L⁻¹ (often above 100% saturation) in Station 1, a result attributed to oxygen production by photosynthesis of submerged macrophytes. Dissolved oxygen levels at the bottom of Station 1 were lower than 2 mg L⁻¹ in June, and above 4 mg L⁻¹ in the other months, probably due to wind mixing. In Station 4, both the surface and bottom DO concentrations were lower than the shallow region for each sampling event. Partly because of lack of macrophytes in the deep pockets, and partly because of the emission of hydrogen sulfide from the near bottom water, surface dissolved oxygen in Station 4 ranged from 5.6 to 10.0 mg L⁻¹, and concentrations near the bottom (5.5 m) were close to 0 mg L⁻¹ at all times of the year. During the period of summer stratification from June to August, dissolved oxygen levels in the deep pockets began to drop at 2.5 m below the surface and kept decreasing until full oxygen depletion near the bottom (5.5 m).

The maximum pH values observed in 2006 at Stations 1 and 4 were in the epilimnion during summer time (Figure 2-4(a) and Figure 2-4(b)). In the shallow region, in the months of May, June and August, pH values usually exceeded 8.7 to a depth of 2 m, with the maximum pH of 9.5 observed at 0.3 m in May. Bottom pH values dropped at 2 m, and near the bottom values were in the range of 7.5 to 8.3 throughout the year. In the deepest region of the lake, highest pH values were observed in June, which were above 10 to a depth of 2.5 m, while the lowest pH value was 6.5 near bottom at 5.5 m depth in July. In most months, pH started to drop at 3 m, and the bottom pH results were around 7. The greatest vertical differences in pH for the entire water column were 1.2 and 2.9 in shallow and deep regions respectively.

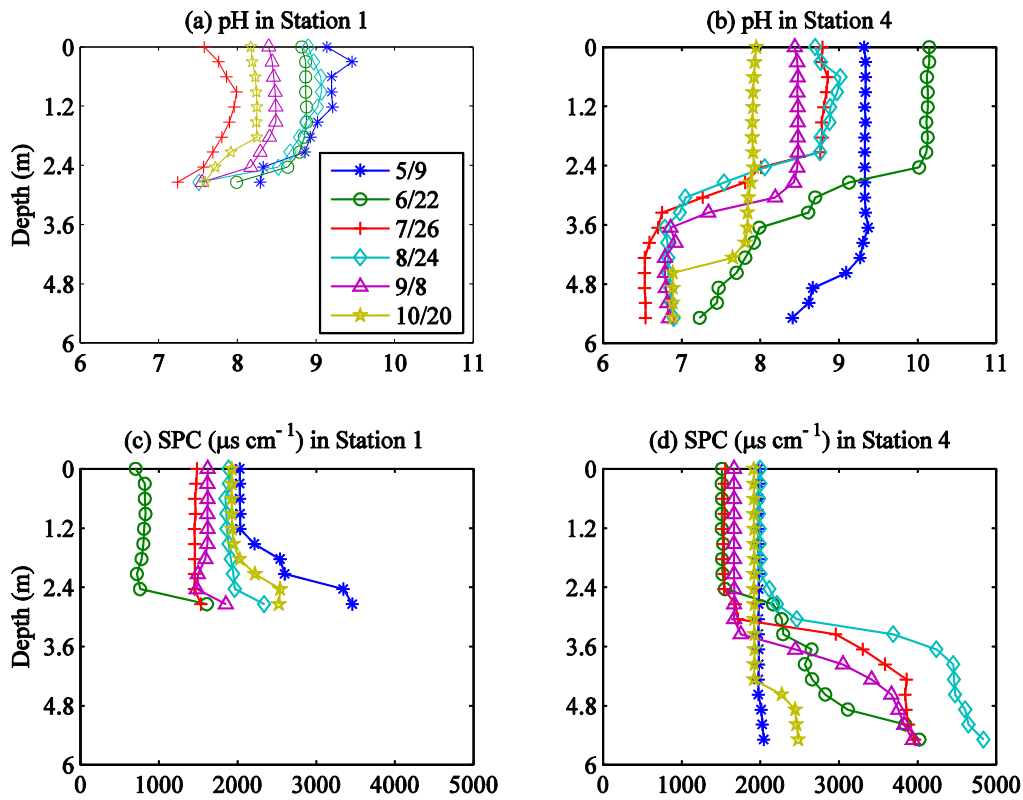


Figure 2-4 Vertical distributions of pH and specific conductance in Stations 1 and 4

In contrast to the previous three parameters, an inverse distribution of SPC was observed in 2006 (Figure 2-4 (c) and Figure 2-4 (d)). It increased with depth at 2 m and 3 m in Station 1 and 4

respectively. The strongest vertical stratification in SPC was encountered in August in Station 4, at $5000 \mu\text{S cm}^{-1}$ near the lake bottom, when it was about 2.5 times that of the surface. In contrast, the maximum vertical difference in Station 1 was lower, which were within a factor of 2. Anoxic conditions in the bottom from May to September in Station 4 were coincident with differences in the vertical distribution of specific conductance by factors of two to three times.

2.4.2 Effects of aeration on vertical distributions of physiochemical parameters

To characterize the effects of aeration on stratification, the vertical distributions in the deep region (Station 4) are first presented for the surface, middle (2.8 m) to the bottom (5.5 m). Aeration had a marked influence on breaking up the lake thermal stratification (Figure 2-5). During pre-aeration, temperature differences, from surface to the bottom, ranged from 0.9°C during lake overturns to 7.4°C during stratification periods. With the introduction of aeration, the differences at both depths decreased markedly. Statistical analysis showed that the surface and bottom differences in 2010 and 2011 were reduced significantly ($p < 0.05$) compared to the pre-aeration condition in 2006 and May to August 2009. Other than destroying the thermal stratification, aeration had little effect on surface seasonal temperature patterns. Maximum temperatures for each year were around 23°C to 26°C usually from June to August, and minimum values of 3.4 to 5.0°C usually started in November and lasted until the following February.

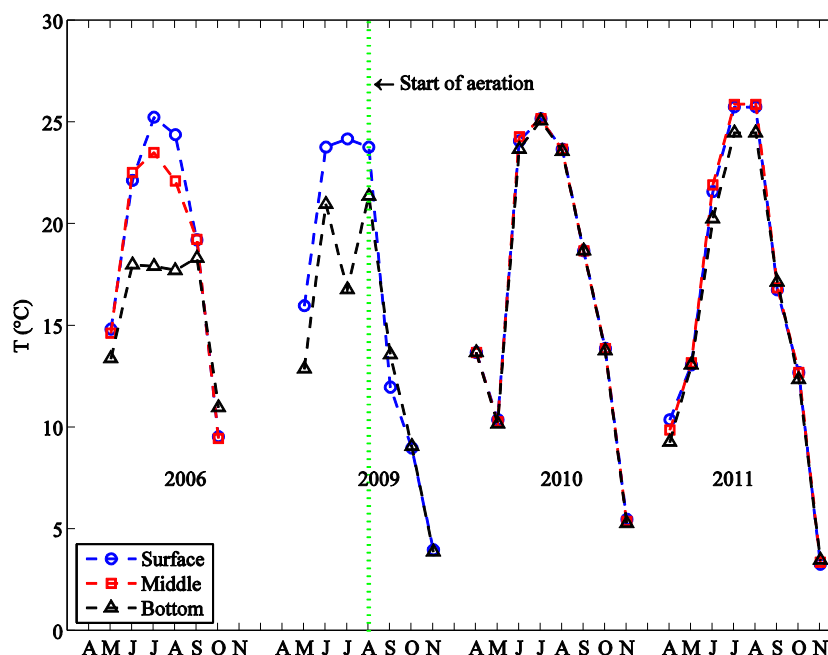


Figure 2-5 Temperature profile at surface, middle and bottom in sampling location 4 Start of aeration is indicated by a vertical line

With the start of aeration, differences in dissolved oxygen concentrations between the surface and 5.5 m bottom also decreased markedly (Figure 2-6). Nevertheless, because the hypolimnetic water was hypoxic with accumulated sulfide and organic matter, the mix of surface and bottom water consumed dissolved oxygen in the entire lake rapidly. As a result, the dissolved oxygen levels in the entire lake declined remarkably to lower than 1 mg L^{-1} in the first several weeks during the start of aeration system, causing a severe fish kill. This condition lasted for about a six weeks. The dissolved oxygen level started to recover to above 2 mg L^{-1} in October and kept increasing to 8 mg L^{-1} in November when the lake iced over. The variations of dissolved oxygen throughout the water column were significantly reduced in 2010 and 2011 ($p < 0.05$) compared with the pre-aeration conditions. In the year 2010, dissolved oxygen concentrations in surface water ranged from 5.1 mg L^{-1} to 9.5 mg L^{-1} , and the differences through water column were within 0.5 mg L^{-1} for most sampling dates except July, when the highest lake water temperature was observed in the year, and the variations in the water column increased to 2 mg L^{-1} . A similar

trend was documented in July 2011, and the dissolved oxygen level dropped from 4.56 to 1.46 mg L⁻¹ from surface water to the bottom of the deepest pockets (Stations 2 and 4) when the highest lake water temperature was monitored. The differences are within 1 mg L⁻¹ in the other sampling dates in 2011. The aeration system was adjusted to reduce the low dissolved oxygen observations in July of 2010 and 2011 by increasing the air pressure to the diffusers in the deepest locations, which are Stations 2 and 4. This adjustment enhanced the circulation in these locations and the anoxic bottom conditions were eased within two weeks of making the adjustment. Crayfish, sunfish and other fish species were observed in the spring of 2010 and kept increasing in 2011, which indicated the improvement of the lake water dissolved oxygen concentrations.

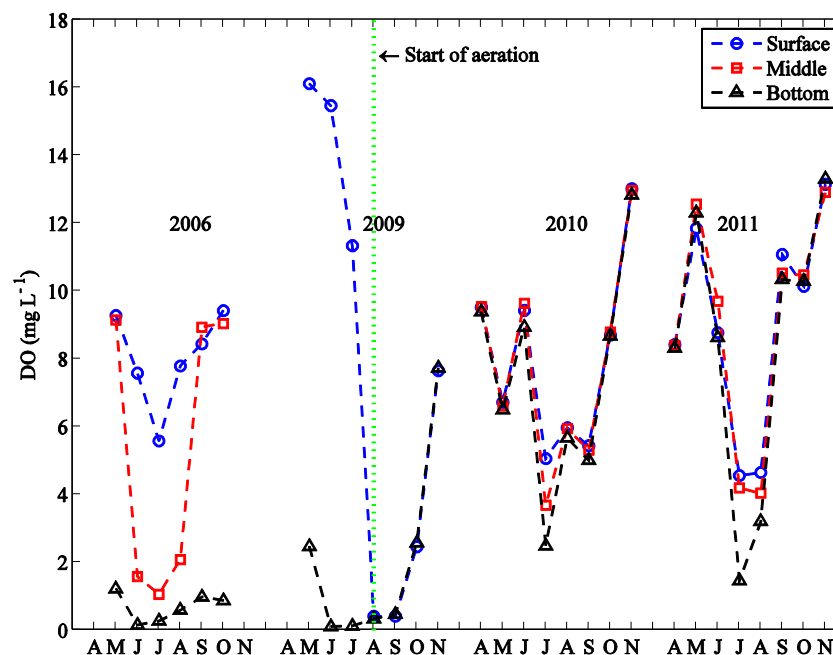


Figure 2-6 Dissolved oxygen profile at surface, middle and bottom in sampling location 4. Start of aeration is indicated by a vertical line

The maximum pH of Fossil Creek Lake before aeration was above 10 during the summer season in the surface layer, and the bottom layer was usually 2-3 units lower. The pH variations of the water column declined significantly ($p < 0.05$) following installation of the aeration system,

especially in 2010 and 2011(Appendix B-1). However, the *average* pH values increased from 2010 to 2011. Peak values often were associated with photosynthesis of aquatic macrophyte during summer time.

Before aeration, specific conductance increased with depth, and the bottom values were about two to three times that of the surface. The vertical distribution differences were significantly reduced ($p<0.05$) in the two years of aeration (Appendix B-2). However, there was little effect on seasonal specific conductance patterns. Maximum values were observed in July of 2010 and 2011.

The shallow regions of Fossil Creek Lake are usually mixed by wind except summer time (Figure 2-3 and 2-4). As the deep regions, the pre-aeration stratification in temperature, dissolved oxygen, pH and specific conductance during summer time was also successfully eliminated in the shallow region, with the introduction of vertical mixing of the water column by the aeration system.

2.4.3 Effects of aeration on spatial variation of physiochemical parameters

To investigate the effects of aeration on the water quality of the entire lake, physiochemical parameters at the water surface and at the bottom from the 11 sampling locations of Fossil Creek Lake are presented by box plots, which illustrate the maximum, minimum, median, the 25th and 75th percentile of all data monitored in each month in years 2006, 2009, 2010 and 2011 from April to November. Paired surface and bottom values in temperature, dissolved oxygen, pH and specific conductance are demonstrated two years before aeration and two years after aeration.

Temperature

The paired surface and bottom values from the 11 sampling locations are displayed in Figures 2-7 and 2-8 from 2006 to 2011 by box plots. The plot on the left in blue represents the surface results, while the bottom values are shown in red by the notched box plots on the right.

The plots show temperature differences at all the sampling locations (deep and shallow regions) between the surface and bottom were significant ($p < 0.05$) in the months before aeration. With the start of aeration, the vertical temperature differences were reduced significantly from August 2009 to the year 2011, which indicates that the entire lake was well mixed by the aeration system, and thermal stratification was successfully eliminated.

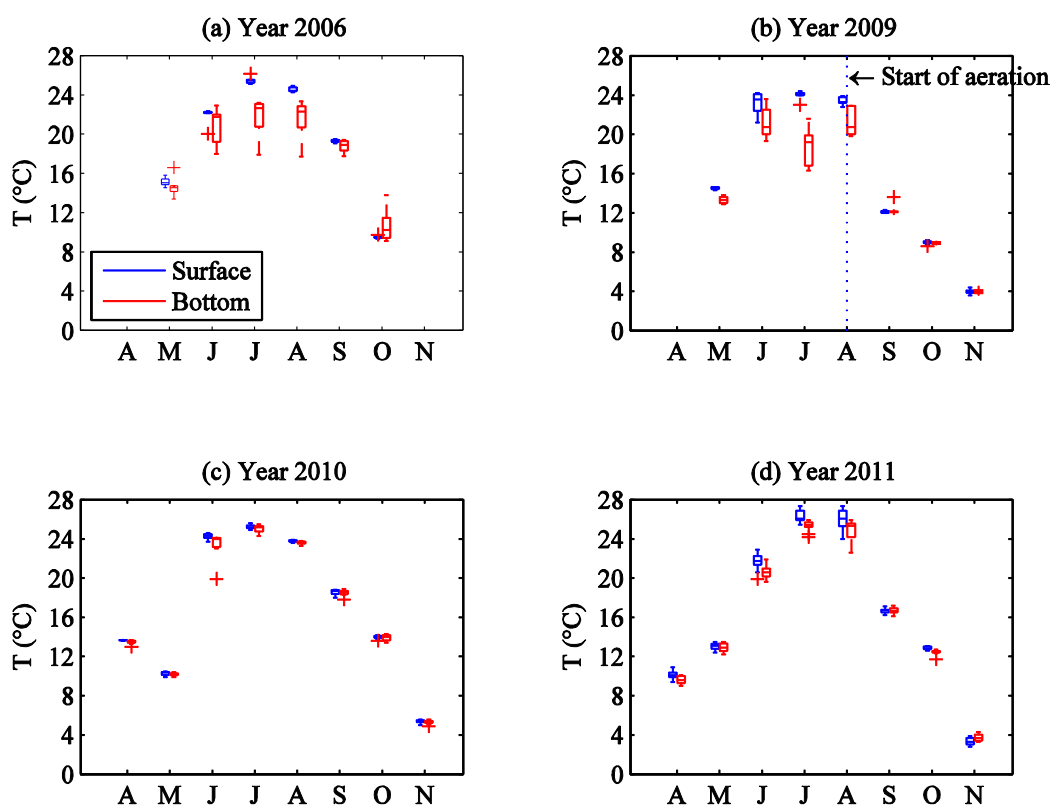


Figure 2-7 Surface and bottom temperature from 11 sampling locations. Start of aeration is indicated by a vertical line

Dissolved oxygen

In the months prior to the initiation of aeration, dissolved oxygen in surface layers from all the sampling locations of the lake was in the range of 6 to 14 mg L⁻¹, except at the deepest sampling location in July when the minimum surface value around 2 mg L⁻¹ was observed. Minimum values of the bottom dissolved oxygen contents indicated that the near bottom layer oxygen was depleted from June to October. With the start of aeration, the vertical dissolved oxygen differences declined significantly ($p < 0.05$) from August 2009 to 2011 (Figure 2-8).

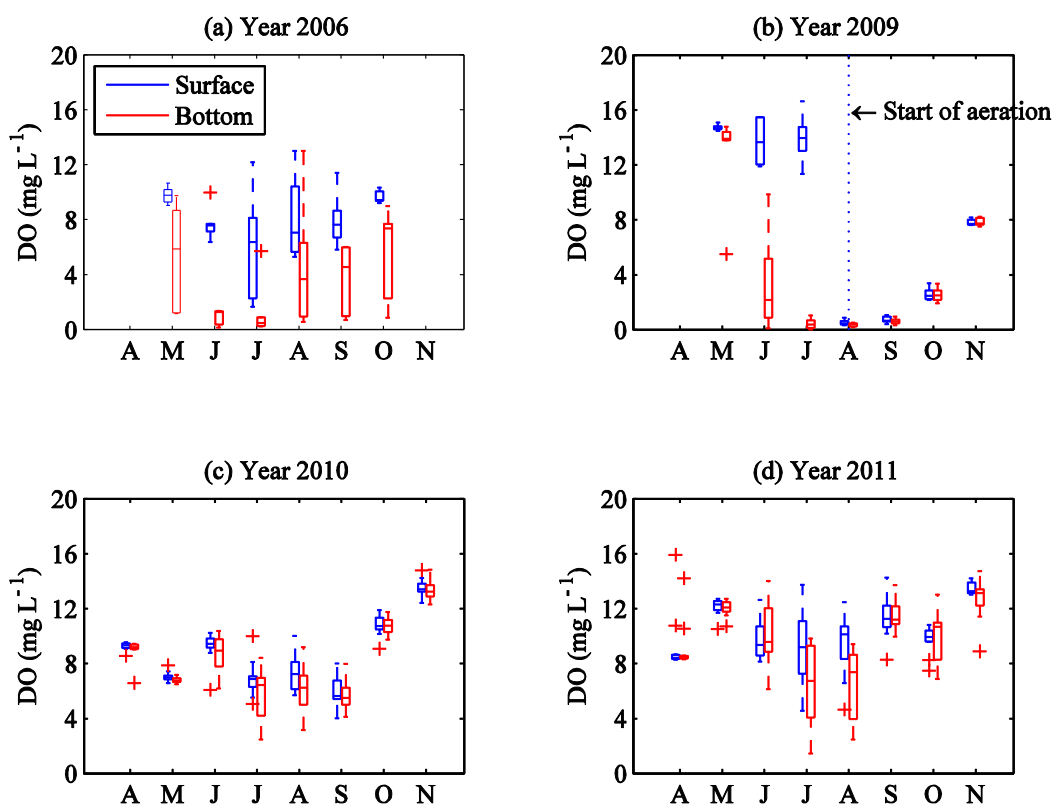


Figure 2-8 Surface and bottom dissolved oxygen from 11 sampling locations. Start of aeration is indicated by a vertical line

pH

The vertical pH variation decreased steadily following the start of aeration (Appendix B-3). During the pre-aeration year 2006, pH ranged from 6.5 to 10.3 and peak values often were associated with photosynthesis by phytoplankton and macrophytes. This variability was reduced significantly ($p < 0.05$) with the introduction of the aeration system during the second and third

year of aeration. Maximum pH values were usually observed in late June or July and subsequently declined in the following months of the year. The pH values in the entire lake in 2011 were higher than those in 2010, which was mainly because the growth of macrophytes was promoted by the aeration system. As a result, the uptake of CO₂ by photosynthesis, combined with the loss of CO₂ via the aerator in the water increased the pH of the water.

Specific conductance

Specific conductance values near the bottom layers were significantly ($p<0.05$) reduced after the aeration system started (Appendix B-4). Similar to the previous parameters, the surface and bottom results were stable in 2010 and 2011, and the vertical variations were significantly ($p<0.05$) reduced compared to the pre-aeration period.

In addition to the in-field water quality parameters, other parameters including phosphorus, nitrogen, carbon, alkalinity, suspended solids, chlorophyll a, Secchi disk transparency, sulfate, hardness and hydrogen sulfide were monitored from 2009 to 2011 from the 11 sampling locations. Surface and bottom values during the pre-aeration period of 2009, and the after aeration period of 2010 and 2011 are shown in the following sections.

Total phosphorus and orthophosphate

The surface TP concentrations during the pre-aeration period of 2009 were in the range of 0.09 and 1.15 mg L⁻¹ (Figure 2-9 (a)), while the TP levels near bottom layer were 5 to 10 times higher (Figure 2-9 (b)), which showed significant variability ($p<0.05$) through the water column. Surface TP concentrations varied from 0.04 to 0.76 mg L⁻¹ in 2010, and 0.16 to 0.68 mg L⁻¹ in 2011, and there were no statistical differences ($p>0.05$) of surface TP levels from all the sampling locations from 2009 to 2011 (Figure 2-9 (a)). With the mixing of the vertical water

column by the aeration system, TP concentrations of the near bottom water were significantly reduced ($p<0.05$) in 2010 and 2011 (Figure 2-9 (b)). Meanwhile, the surface and bottom TP levels were not significantly different ($p>0.05$) in these two years.

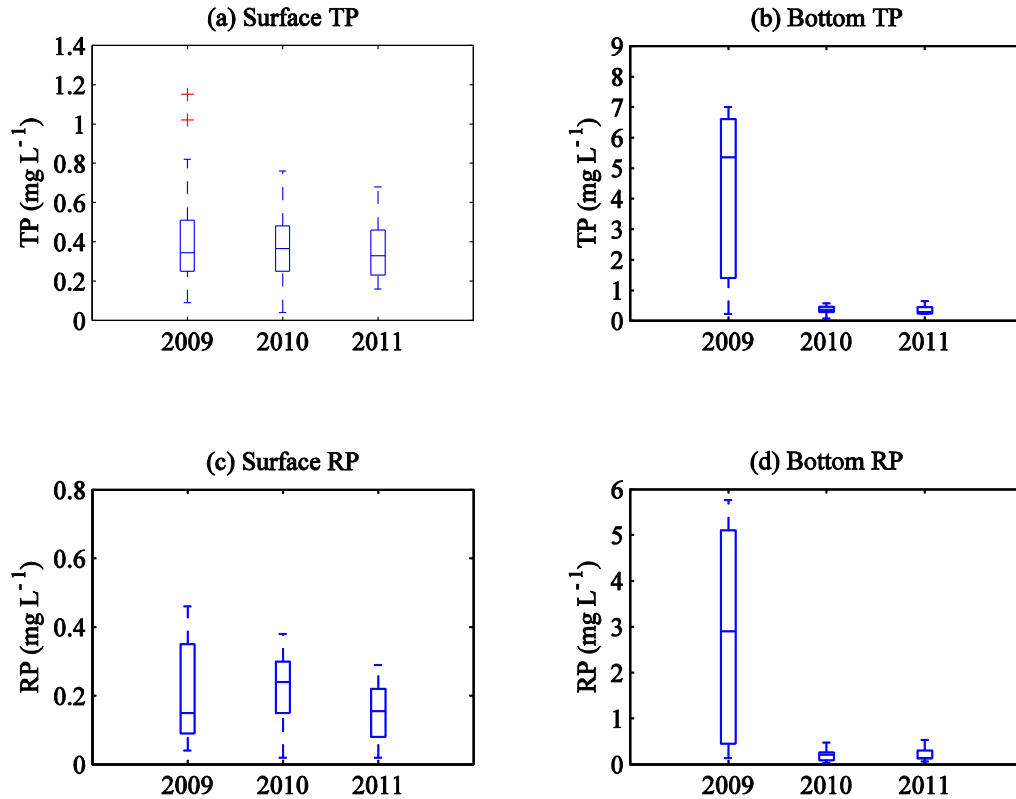


Figure 2-9 Surface and bottom TP and RP in 11 sampling locations from 2009 to 2011

The RP concentrations were detected in the range of 0.04 to 0.46 mg L^{-1} at the lake surface before the aeration system was started ((Figure 2-9 (c)), while the near bottom layer concentrations varied from 0.14 to 5.76 mg L^{-1} (Figure 2-9 (d)), which showed significant differences of epilimnion and hypolimnion ($p<0.05$). Surface RP concentrations varied from 0.02 to 0.38 mg L^{-1} in 2010, and 0.02 to 0.29 mg L^{-1} in 2011, and there were no statistical differences ($p>0.05$) of surface RP levels in the three years (Figure 2-9 (c)). There was a significant reduction ($p<0.05$) of RP in the near bottom layers following the installation of the aeration system. Near bottom water RP levels were in the range of 0.03 and 0.47 mg L^{-1} in 2010, and 0.06

to 0.53 mg L^{-1} in 2011, and the surface and bottom differences were insignificant ($p>0.05$) in these two years.

Total nitrogen and total organic carbon

During the pre-aeration year 2009, surface TN throughout the whole lake ranged from $0.23\text{-}1.80 \text{ mg L}^{-1}$ (Figure 2-10 (a)), while the bottom layers were in the range of $1.56\text{-}30.13 \text{ mg L}^{-1}$ (Figure 2-10 (b)), and the high values of bottom TN were usually observed during summer time. The bottom TN concentrations were significantly ($p<0.05$) reduced after aeration (Figure 2-10 (b)), and there were no significant differences ($p>0.05$) between surface and bottom in 2010 and 2011. TN concentrations of the surface water varied from 0.38 to 1.97 mg L^{-1} in 2010 and 0.44 to 1.58 mg L^{-1} in 2011, and the differences between the three years were insignificant ($p>0.05$).

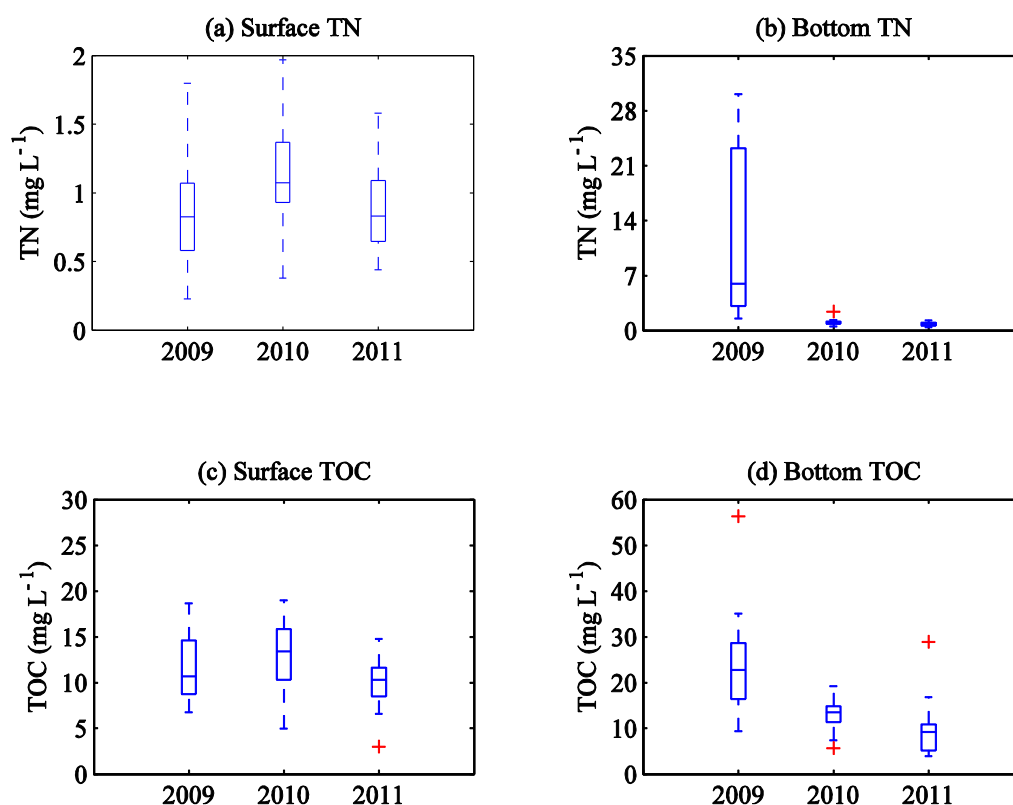


Figure 2-10 Surface and bottom TN in 11 sampling locations from 2009 to 2011

Similar trends in TOC were also observed before and after aeration in Fossil Creek Lake. During the pre-aeration period, significant differences ($p < 0.05$) were observed in TOC between surface (Figure 2-10 (c)) and bottom water (Figure 2-10 (d)). Bottom TOC concentrations were significantly reduced with the introduction of aeration in 2010 and 2011 (Figure 2-10 (d)). Meanwhile, there were no significant differences through the water column in 2010 and 2011. Moreover, the variations of surface TOC concentrations before and after aeration were insignificant ($p > 0.05$).

Total suspended solids and alkalinity

Surface TSS concentrations before and after aeration were lower than 10 mg L^{-1} , and there were no significant differences ($p > 0.05$) during the three years (Figure 2-12 (a)). Aeration produced a significant decline ($p < 0.05$) of near bottom water TSS levels (Figure 2-12 (b)). Maximum TSS was monitored at 163 mg L^{-1} in the lake bottom, which were significantly reduced to lower than 10 mg L^{-1} after aeration (Figure 2-12 (b)). The vertical differences through water column in 2010 and 2011 were insignificant ($p > 0.05$).

Total alkalinity concentrations of surface water averaged at 163 mg L^{-1} in 2009 and 165 mg L^{-1} in 2010, and no significant differences ($p > 0.05$) were observed in 2009 and 2010. However, the levels dropped significantly in 2011 ($p < 0.05$), with average values of 95 mg L^{-1} (Figure 2-11 (c)). The highest total alkalinity results observed near the bottom water was 1224 mg L^{-1} (Figure 2-11 (d)) were also encountered during the summer time before aeration, when the stratification was the strongest, coincident with completely anoxic bottom water. The bottom concentrations were significantly reduced ($p < 0.05$) in 2010 and 2011 (Figure 2-11(d)). There were no significant differences ($p > 0.05$) between lake surface and bottom alkalinity levels in 2010 and 2011.

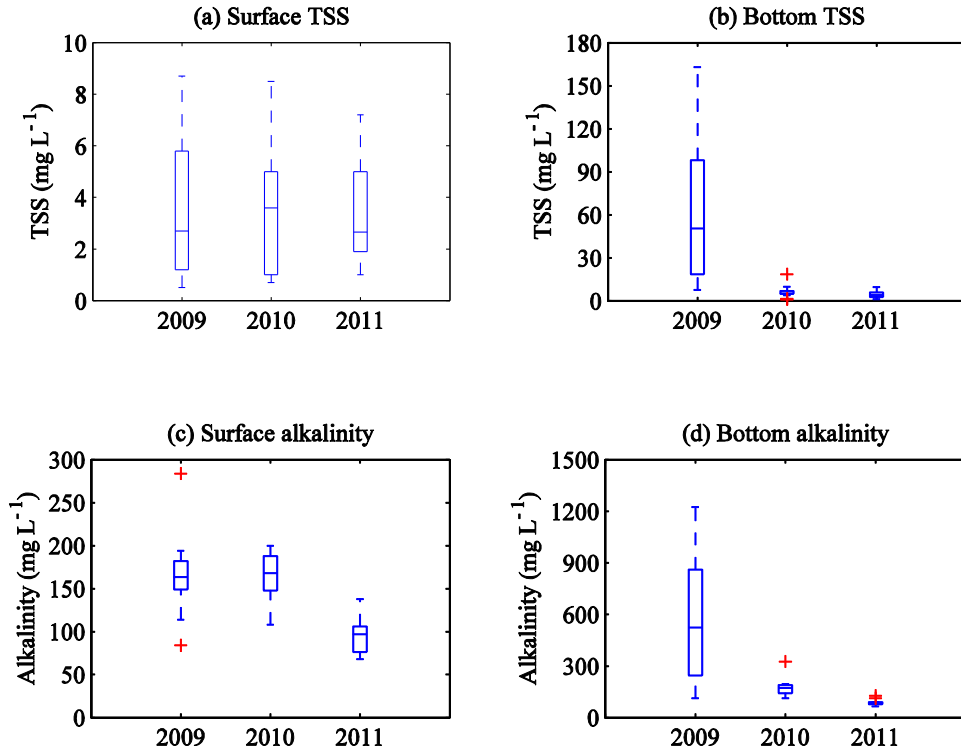


Figure 2-11 Surface and bottom TSS and alkalinity in 11 sampling locations from 2009 to 2011

Chlorophyll a and Secchi disk depth

Chlorophyll a concentrations during the pre-aeration year (2009) ranged from 8.44-55.14 $\mu\text{g L}^{-1}$ and averaged 30.4 $\mu\text{g L}^{-1}$ (Figure 2-12). Chlorophyll a concentrations declined significantly ($p < 0.05$) during aeration, ranging from 9.12-42.93 $\mu\text{g L}^{-1}$, and mean concentrations were 24.89 $\mu\text{g L}^{-1}$ in 2010, and ranging from 11.13-48.43 $\mu\text{g L}^{-1}$, and averaged 24.48 $\mu\text{g L}^{-1}$ in 2011. There were no statistical differences between 2010 and 2011 in chlorophyll a concentrations. Fossil Creek Lake is a submerged macrophytes dominated clear water lake before and after aeration. Peak values of Secchi disk transparency were 2.13 m, 3.05 m and 2.9 m for 2009, 2010 and 2011 respectively (Figure 2-13). The average values of the three years were 1.33m, 1.42m and 1.28 m. The changes in transparency appear to be associated with phytoplankton and total suspended solids. Even TSS and phytoplankton biomass were significantly reduced ($p < 0.05$) by aeration, the Secchi disk transparency in the three years were not significant ($p > 0.05$).

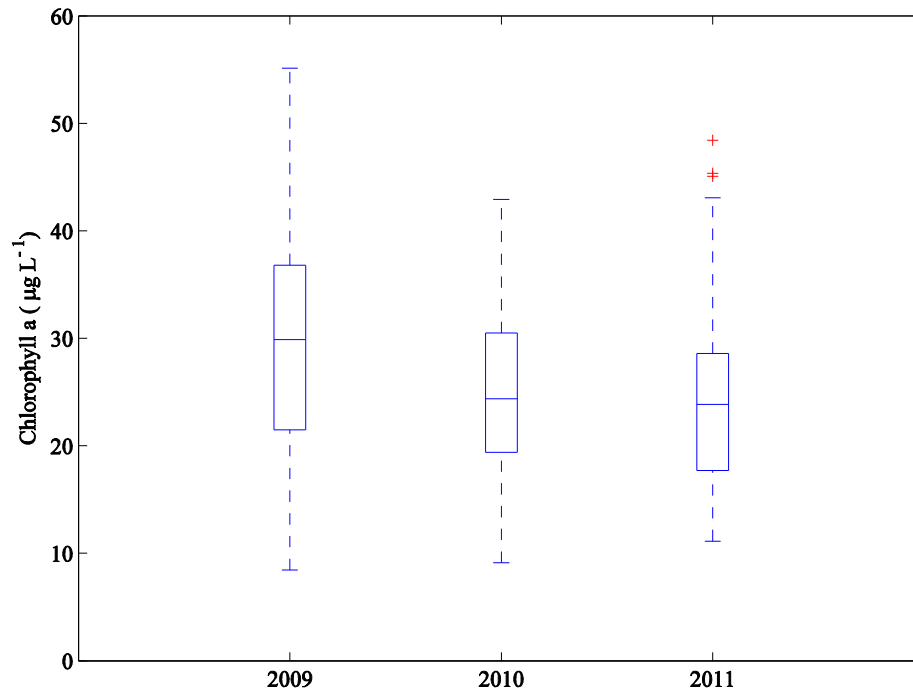


Figure 2-12 Chlorophyll a in 11 sampling locations from 2009 to 2011

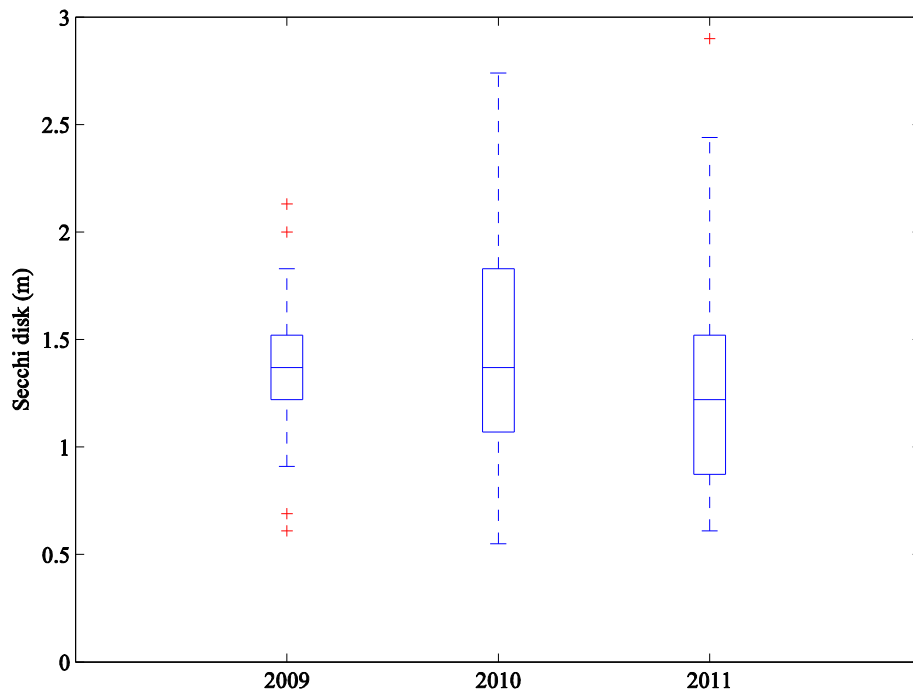


Figure 2-13 Secchi disk transparency in 11 sampling locations from 2009 to 2011

Sulfate, hardness and hydrogen sulfide

Sulfate concentration during the pre-aeration year ranged from 575 to 1980 mg L⁻¹, and averaged 1097 mg L⁻¹ in the surface layer of Fossil Creek Lake. A significant increase ($p < 0.05$) of sulfate

levels were monitored in the lake surface between 2010 and 2011, with means of 1414 and 1393 mg L⁻¹ respectively (Figure 2-14 (a)). The bottom values were between 785 and 2818 mg L⁻¹, with a mean concentration of 1514 mg L⁻¹ before aeration, and no significant differences were monitored through the water column during the pre-aeration period. However, aeration barely affected the bottom sulfate concentrations and the differences in the three years were insignificant ($p>0.05$). The major reason is that the sulfate contents were highly affected by the incoming water, Fossil Creek.

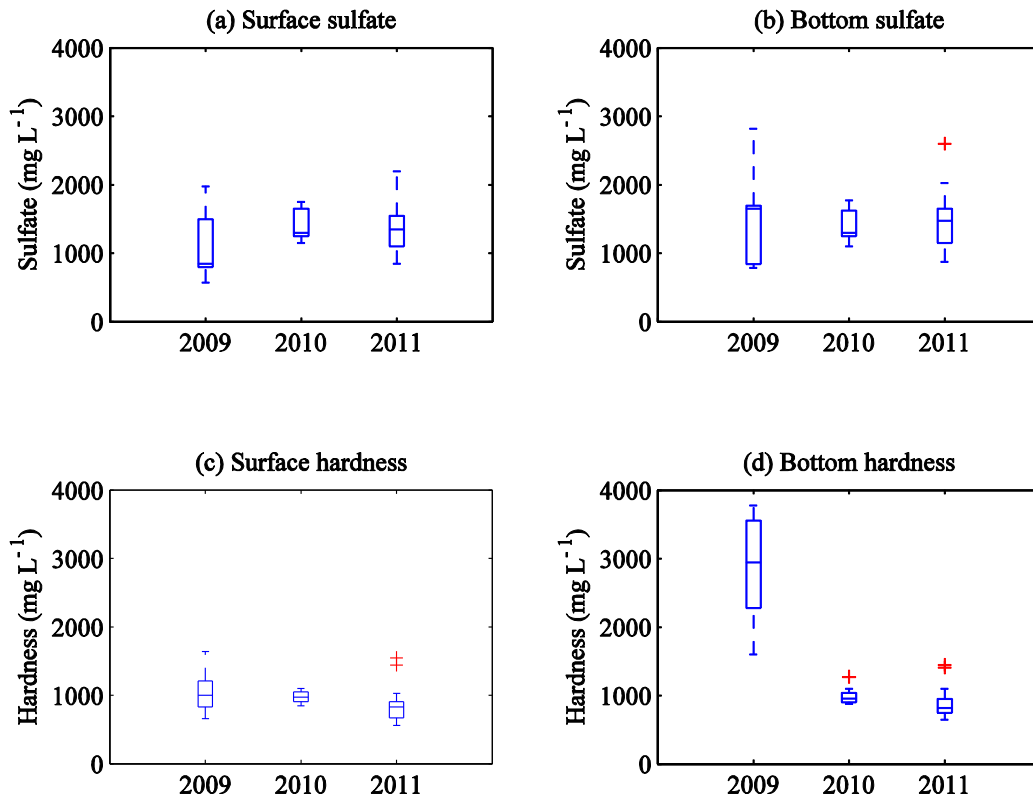


Figure 2-14 Surface and bottom sulfate and hardness in 11 sampling locations from 2009 to 2011

There was little variation ($p>0.05$) of surface hardness values from 2009 to 2011. Hardness concentrations near the bottom water were significantly reduced ($p<0.05$) after aeration in 2010 and 2011. The differences between lake surface and bottom were insignificant ($p>0.05$) during aeration.

Hydrogen sulfide concentrations kept increasing at the bottom near the mud-water interface as the lake stratified before aeration. The maximum concentrations observed were significantly higher at depths greater than 3m in all seasons except spring and fall lake turnover during the pre-aeration years. Vertical variations of hydrogen sulfide contents through the water column were greatest, ranging from 3 to 4 orders of magnitude when the surface results were usually non-detectable ($<20 \mu\text{g L}^{-1}$) and the bottom concentrations were supersaturated, ranging from 80 to 382 mg L^{-1} . In a super saturation condition, hydrogen sulfide will transfer from aqueous phase to gas, and then is released from the lake bottom to the atmosphere until the equilibrium is reached.

When aeration began, it was very evident from the repugnant odor (odor threshold of H_2S is 0.002 mg L^{-1}) that hydrogen sulfide was being vented to the atmosphere as the bubbles of the air plume were rupturing at the water surface. Surface H_2S concentrations increased from non-detectable to 2.5 mg L^{-1} in the first week, attributable to the circulation of the entire vertical water column. During the initial aeration period, while hydrogen sulfide was being vented, there were significant visual adverse effects of hydrogen sulfide toxicity to fish, and essentially all of fish in Fossil Creek Lake were killed as well as the lake invertebrates. The surface hydrogen sulfide concentrations dropped to 1 mg L^{-1} a month after aeration began, and continued decreasing to non-detectable levels two months later, which was in the middle of October of 2009. The bottom concentrations had been reduced to around $0.5\text{-}2 \text{ mg L}^{-1}$ by the end of 2009. But since the surface concentration was lower than the bottom, it is clear that the hydrogen sulfide was being oxidized before it reached the surface. In the aeration years 2010 and 2011, surface and bottom hydrogen sulfide levels were at or below the lower limits of sensitivity (0.02 mg L^{-1}) based on the colorimetric method employed during most times of the year, and a

significant decrease ($p < 0.05$) of hydrogen sulfide was achieved by the aeration system. During July, when the water temperature of the lake was usually between around 26 °C, while DO declined, hydrogen sulfide easily increased near the mud-water interface, and the concentrations in the deepest region of the lake were observed at 5 to 100 mg L⁻¹. Hydrogen sulfide declined to non-detectable levels when the air flow rate to the diffuser in the deepest region was increased.

2.5 Discussion

Aeration of Fossil Creek Lake generally improved water quality in the entire water body. Similar to the observations of many other investigators, the following trends were noted in Fossil Creek Lake:

- 1) The thermal stratification is eliminated;
- 2) Dissolved oxygen concentrations of the hypolimnion increased significantly;
- 3) The pH of the water column increased;
- 4) Nutrient levels (total phosphorus, orthophosphate, total nitrogen and total organic carbon) declined in the bottom water;
- 5) The phytoplankton biomass was lowered;
- 6) Most chemical parameters rarely varied with depth.

2.5.1 Physiochemical parameters

Aeration on CO₂, alkalinity and pH

Hutchinson (1957) pointed out that the relationship between oxygen and carbon dioxide, including bicarbonate, is roughly reciprocal. This trend was observed in Fossil Creek Lake. During the pre-aeration period, the total alkalinity and total inorganic carbon levels below 5m depth increased significantly when the dissolved oxygen was completely depleted during summer time. The late summer in which anoxic bottom conditions were improved by aeration

had less pronounced clinograde distributions than the pre-aeration period. Smith et al. (1975) reported there was no change in either alkalinity or pH during summer aeration in the hypolimnion of Larson Lake, Wisconsin. Some other researchers observed lowered values of total alkalinity in the hypolimnion (Fast et al. 1973 and Toetz, 1977), and an increase in pH by artificial aeration (Fast et al. 1973). Cowell et al. (1987) concluded that the artificial aeration steadily reduced the pH of Lake Brooker. In Fossil Creek Lake aeration produced a significant decline ($p < 0.05$) of hypolimnion total alkalinity and total inorganic carbon in both 2010 and 2011. The aeration of water increases the rate of CO_2 removal from the water column. The reduction of CO_2 in water increases the pH of the water. Similarly, the uptake of CO_2 by photosynthesis, combining with the loss of CO_2 via the aerator in water, increased the pH of the water in 2010 and 2011.

Aeration on organic matters, total suspended solids and clarity

The organic matter in Fossil Creek Lake mainly originates from two sources: (1) aquatic animals and plant decay and (2) carbon fixation by algae and aquatic plants. The organic carbon that reaches the lake sediment surface will partly be mineralized to CO_2 or CH_4 by heterotrophic microorganisms and passed forward through the food chain and incorporated also into other organisms, or be partly buried in the sediments (Brönmark et al., 2005 and Gudas et al., 2010). Hence, the total organic carbon can enhance organism abundance beyond levels supported by photosynthesis in the lake (Brönmark et al., 2005). In the lake bottom, the degradation of organic matter, the generation of CO_2 and reduction of pH are augmented by microbial methane fermentation, nitrification of ammonia, and sulfide oxidation, which result in a decline of dissolved oxygen in the bottom, and in turn enhance the process of degradation. Fee et al. (1996) and Schindler et al., (1997) reported that the reduction in dissolved organic carbon

concentrations can improve clarity of lake water, producing deeper euphotic zones and thermoclines. The response of total organic carbon in Fossil Creek Lake to artificial aeration has been observed to be positive. With the elimination of the anoxic condition in the lake bottom, the total organic carbon concentrations in the bottom layers of the lake decreased significantly ($p < 0.05$) in 2010 and 2011. In addition, the total suspended solids in the water at the lake bottom declined remarkably ($p < 0.05$) during the two year aeration period along with the reduction of total organic carbon. As expected, the clarity of the lake showed significant improvement after aeration with the reduction of total organic carbon and total suspended solids.

2.5.2 Reasons for high sulfate

Sulfate levels above 1000 mg L^{-1} have not been commonly observed in most researched lakes. Lake Mendota in Wisconsin, sulfate concentrations were lower than 7 mg L^{-1} , decreasing with depth, and bottom concentrations were around 3 mg L^{-1} . Hydrogen sulfide near the bottom layer was 5 mg L^{-1} (Ingvorsen et al, 1981). Sulfate concentrations between 3 and 4 mg L^{-1} were observed in the water column during 1986 at 5-meter depth in Little Rock Lake, Wisconsin (Gantzer and Stefan, 2003). Dunnette et al. (1985) demonstrated the mean concentration of epilimnic and hypolimnic in sulfate were essentially the same (3.9 and 4.3 mg L^{-1}) in Third Sister Lake, a moderately eutrophic lake in Michigan. Due to mineral weathering, sulfate concentrations in the range of 8.93 to 34.08 mg L^{-1} were observed in streams of the Marmot Basin watershed of the Rocky Mountains (Alberta) (Mitchell et al., 1986). Pore water sulfate contents from Mine Pit Lake were between 0.86 and 15.55 mg L^{-1} (9.0 and $16.2 \text{ } \mu\text{mol L}^{-1}$) (Meier et al., 2012). Based on this research, Fossil Creek Lake is a very hard water lake with very high sulfate concentrations compared with most lakes in North America.

Sampling results from the tributaries of Fossil Creek showed that Tributary 1 (Figure 2-15) of Fossil Creek flows through a sedimentary layer with high sulfate content in the groundwater, resulting in a sulfate level of 3100 mg L^{-1} . The sulfate concentrations downstream were diluted by flows from other tributaries. As a result, the sulfate concentration that flows into Fossil Creek Lake is around 1000 mg L^{-1} . Similar conditions were reported by Wu et al. (2012) in the local groundwater in Bitterfeld, Germany with sulfate concentration between 740 and 1010 mg L^{-1} , which indicated that the high sulfate concentrations in groundwater are not without precedent.

In freshwater sediment, organic matter decomposition by denitrification and sulfate reduction is usually limited due to the restricted supply of NO_3^- and SO_4^{2-} ; allowing for methanogenesis predominating. In marine sediments or sediments with high sulfate contents, sulfate reduction exceeds methanogenesis because of the high availability of sulfate, and sulfate reduction is the main process in organic material degradation (Varjo et al., 2003). The extreme high sulfate availability in Fossil Creek Lake enhances the activity of sulfate-reducing bacteria, which makes sulfate reduction the dominant process and increases the overall mineralization rate of organic matter, in turn increasing the hydrogen sulfide concentrations in the near bottom layer. Sediment production of hydrogen sulfide can increase the oxygen demand rate of sediment leading to a reduction in dissolved oxygen in overlying waters. Hydrogen sulfide is toxic to fish and macroinvertebrates, particularly at $\text{pH} < 7$ when 50-100% of the hydrogen sulfide present is in the undissociated form, H_2S (Dunnette et al., 1985).



Figure 2-15 Tributaries of Fossil Creek

The number 1 marked in Figure 2-15 indicates that the location of the sedimentary outcropping that contains a high concentration of sulfate.

2.5.3 Trophic state analysis

The ratio of N: P is often used to assess nutrient limitation, which is based on the assumption that algal growth is proportional to the quantity of either one or the other nutrient in the water body (Ryding and Rast 1989). Although there is widespread agreement that high ratios are indicative of P limitation and low ratios indicative of N limitation, there is no widespread agreement on the levels at which changes occur. The following indicators were used to evaluate the nutrient limiting phytoplankton growth: lake water TN: TP < 20 (molar based), a lake is considered N limited; above 38, it is considered P limited (Sakamoto 1966 and Kosten et al., 2009). The annual average TN: TP ratios were used in this section to examine the limiting nutrient in Fossil Creek Lake. The annual TN: TP ratios were 18.39, 19.92 and 19.70 in 2009, 2010 and 2011 respectively, which indicated that for Fossil Creek Lake, nitrogen is the limiting nutrient both before and after

aeration. Aeration produced a remarkable decline of bottom TN and TP in the lake, but had little effect on switching the limiting nutrient.

Soluble orthophosphate was reported to increase under anoxic conditions due to mineralization of organic matter in the water-sediment interface or the release from bottom sediment, which in turn increased the total phosphorus in the hypolimnion. This condition was observed during the pre-aeration period, with extremely large values of soluble phosphorus and total phosphorous at lake depths lower than 5 m especially during summer time. Consequently, there was a decrease of both soluble and total phosphorus during the summers of 2010 and 2011 with improved dissolved oxygen levels in the bottom waters.

The decrease of nitrogen and phosphorus through aeration in the lake is expected to control the eutrophication condition. To examine the nutritional status of Fossil Creek Lake before and after aeration, two trophic status indices were applied: the Carlson Trophic Status Index (TSI) and Kratzer and Brezonik trophic status index. The Carlson Index was developed for phosphorous limited lakes and reservoirs (Carlson 1977), but Fossil Creek Lake is nitrogen limited. Work by Kratzer and Brezonik (1981) allows the index to be predicted from nitrogen concentrations as well. The indices of total phosphorus, total nitrogen, chlorophyll a, and Secchi disk are shown in Figure 2-16, and the yearly average values are summarized in Table 2-2.

Table 2-2 Yearly average values of Fossil Creek Lake trophic status index

TSI	2009	2010	2011
TN (Kratzer and Brezonik, 1981)	67.24	54.91	51.4
TP (Carlson, 1977)	104.09	86.48	87.53
Chlorophyll a (Carlson, 1977)	63.09	61.56	61.97
Secchi disk (Carlson, 1977)	56.31	56.21	57.53

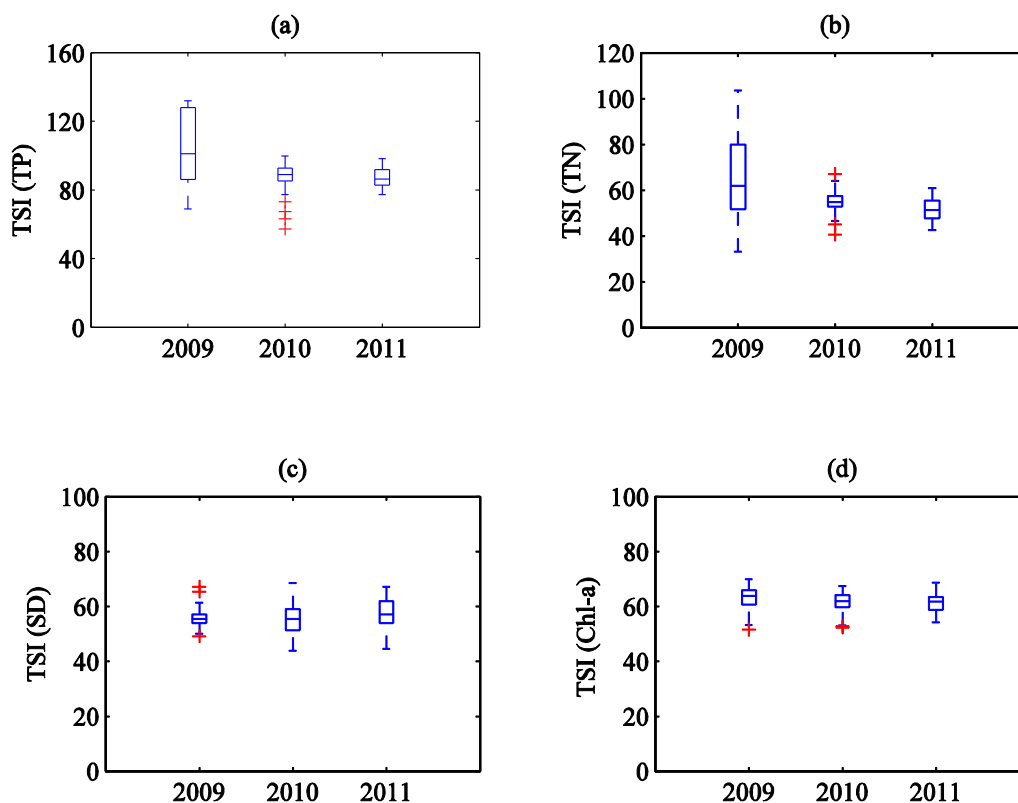


Figure 2-16 Distributions of Fossil Creek Lake trophic status index

TSI of TP was in the range of 69.04 and 131.82 in the epilimnion, and averaged at 104.09 during the pre-aeration period of 2009. In the following two years of aeration, the TSI values of total phosphorus were 57.34-99.80 with a mean of 86.50 and 77.33-98.20 with average values of 87.53 separately (Figure 2-16). The mean TSI of TN values varied from 33.18 to 103.59 and averaged at 67.24 during the pre-aeration period. The TSI values of TN were in the range of 40.6-67.08 with a mean of 54.91 in 2010, decreasing to 42.60-61.05, and averaging 51.40 in 2011, which indicates the lake is between mesotrophic and eutrophic condition. The average TSI values of Secchi disk depth changed little before and after aeration, which are 56.31, 56.21, and 57.53 in 2009, 2010 and 2011 (Table 2-2). Mean TSI values of chlorophyll a decreased from 63.09 in 2009 to 61.56 in 2010, and was 61.97 in 2011. The Carlson index of total phosphorus, chlorophyll a and Secchi disk indicated that the lake is between a eutrophic and hypereutrophic condition based on the criteria of Figure 1-3. The TSI values of total nitrogen indicated that

Fossil Creek Lake was eutrophic in 2009, 2010 and 2011. Figure 2-17 shows the lake trophic status in 2011 based on the Kratzer, Brezonik and Carlson status index.

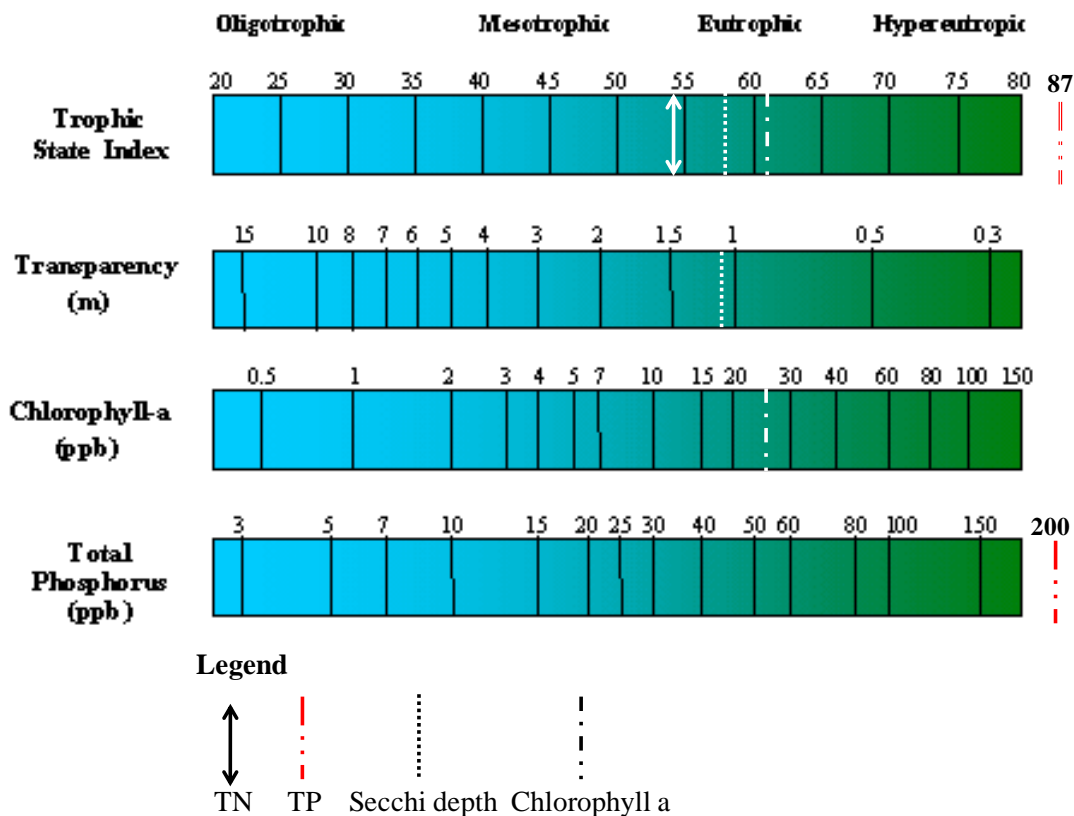


Figure 2-17 Fossil Creek Lake trophic status in 2011 based on Carlson's Trophic Status Index

Besides the Carlson trophic state indices, the other trophic state classification systems from U.S. EPA National Eutrophication Survey (U.S. EPA, 1974), Vollenweider Phosphorous Loading Index (1975), and OECD index (Vollenweider and Kerekes, 1980) also indicated that Fossil Creek Lake is a eutrophic lake. As a result, the aeration system has had little effect on the lake trophic status, which is similar to Lake Brooker, Florida (Cowell, 1987) and Lake Särkinen, Finland (Sandman et al. 1990) which also used artificial aeration, but had less effect on reducing the nutrient levels.

2.5.4 Low DO observed during summer time

DO between 2-4 mg L⁻¹ was observed below a depth of 5m in the deepest region of the lake in July during the first (2010) and second (2011) years of aeration. This phenomenon is commonly observed in lakes during summer time. Spruce Knob Lake, West Virginia (Labaugh, 1980), with a maximum depth of 5 meters, which was artificially aerated with a modified full lift aerator, experienced oxygen levels of 4 mg L⁻¹ and above 2 mg L⁻¹ in the summer of the first and second years of aeration respectively in the bottom waters. This condition is attributable to the limitation of aeration in shallow lakes. Aeration provides little new oxygen and merely moves dissolved oxygen that is produced by photosynthesis and infusion from the atmosphere in the surface waters down to the bottom. But the less dense, warmer surface water tends to rise before it finally reaches the sediments, which results in a layer of dense lower-DO water over the sediments that is affected if the bottom sediments are still anoxic, especially during summer time with the highest water temperatures. The increase of pH due to CO₂ removal by aeration in water causes the equilibrium of H₂S ionization to move towards the formation of HS⁻ and S²⁻, which cannot be removed directly and efficiently by aeration. Fossil Creek Lake has an extremely high level of sulfate concentrations. The decline of dissolved oxygen in combination with sufficient sulfate supply and organic matters from near bottom waters during summer time provide a favorable condition for sulfide reducing bacteria, which reduces sulfate to sulfide. High hydrogen sulfide concentrations in the deepest region were encountered at 5-100 mg L⁻¹ in the summers of the two years of aeration. In turn, the production of hydrogen sulfide from mud-water interface can increase the oxygen demand rate of sediment, which leads to a further reduction of dissolved oxygen in overlying waters. However, the low dissolved oxygen conditions and high sulfide levels were only observed in the deepest sampling location in July

when the lake water temperature was highest, and they were successfully within eliminated two weeks after the increasing the air flow rate at the deepest spot.

2.6 Conclusions

Submerged aeration is a technique intended to eliminate thermal stratification and improve bottom oxygen levels. The primary strengths are simplicity and relatively low cost. The submerged aeration system in Fossil Creek Lake successfully eliminated most of its undesirable features, such as oxygen depletion, thermal stratification, hydrogen sulfide odor and fish kills during lake overturns. Moreover, the vertical variations of pH, SPC, TP, RP, TN, TOC, alkalinity, TSS, sulfate, and hardness have been significantly reduced with the complete mixing of the entire lake. After two years of aeration, pH values have increased gradually, which is mainly due to the photosynthesis and the loss of CO₂ by the aeration system. Aeration has had little effect on reducing sulfate levels of the entire lake; the sulfate levels have stayed the same as in the pre-aeration condition, and are strongly affected by the sulfate contents of the water supply. The trophic state index and limiting nutrient were not affected by aeration. Even though the trophic state has remained the same as the pre-aeration period, Fossil Creek Lake has remained in a clear water state. Therefore, aeration of Fossil Creek Lake has successfully achieved its primary goals of eliminating the hydrogen sulfide odor, improving water quality for aquatic life, and making it aesthetically pleasing to park visitors. Nevertheless, continuous aeration may be necessary for multiple years to achieve desired trophic conditions.

CHAPTER 3

COMPARISON OF SUBMERGED AERATION IN TWO SMALL, SHALLOW, EUTROPHIC LAKES IN COLORADO

3.1 Introduction

Urban lakes are usually shallow, and man-made, accommodating many different uses, such as fisheries, irrigation, recreation and aesthetic landscaping. This makes them more vulnerable to changes in water quality through nutrient accumulation than are rural, natural lakes. A "shallow lake" is usually defined as a permanent standing body of water that is sufficiently shallow to allow light penetration to the bottom sediments to potentially support photosynthesis of higher aquatic plants over the entire bottom (Wetzel, 2001). Shallow lakes tend to accumulate terrestrial organic matter and nutrients. Being of relative volume, the loading of nutrients per unit volume can be high (Wetzel, 2001). The shallowness makes the lake more vulnerable to variations in weather and nutrient inputs from runoff. Moreover, nutrients in shallow lakes may be significantly increased by sediment resuspension, which makes the eutrophication process more rapid and drastic (Schallenberg and Burns 2004). The water quality deterioration caused by the speeding up of the eutrophication process manifests itself in changes in the chemical, physical and biological characteristics of the receiving waters. These alterations include (i) decreased water transparency; (ii) increased pH; (iii) hypolimnetic deoxygenation and the accompanying production of hydrogen sulfide (H_2S) and methane (CH_4); and (iv) release of nitrogen (N), phosphorus (P) and iron (Fe) from lake bottom sediments during periods of anoxia (Kalff 2002, and Oberholster, et al., 2007).

External nutrient loading control could feature a watershed scale approach or focus on the inlet, which would lower the inflow of nutrients loading to the lakes, and over time might lessen the degree of eutrophication (Hu et al., 2010). Nevertheless, shallow lakes are usually less responsive to external nutrient loading reductions due to the internal loading of sediment resuspension. As a result, the effectiveness of the external method could take a substantial length of time to realize. In contrast, in-lake restoration techniques are believed to accelerate lakes recovery period.

Sediment dredging is one effective lake restoration technique to remove surface bottom layers rich in organic matters and nutrients and control their release or nutrient bioavailability (Zhong et al., 2008 and Hu et al., 2010). However, dredging changes the topography of the bottom of the lake. Furthermore, this expensive and aesthetically displeasing process is unavoidably exacerbated by the disposal of the dredged sludge, and is not always successful in controlling the release of pollutants (Ryding, 1982; Annadotter et al., 1999; and Hu et al., 2010). The process may negatively affect the products and services delivered by aquatic ecosystems, including submerged aquatic vegetation, fish, good-quality water, recreation, the potential for nature conservation, and increased water turbidity (Wu, 2007). Awareness about the side effects of dredging has resulted in a demand to monitor its long-term impacts.

Dissolved oxygen is often the most important end-point measurement in the management of lake and reservoir water quality (Welch and Gibbison, 2010). Given the importance of dissolved oxygen and its response to eutrophication, the emphasis on maintenance of aerobic conditions is an important consideration for successful lake nutrient management. Complete artificial aeration

is an effective way to destroy or prevent thermal stratification and increase dissolved oxygen levels in affected water. Complete aeration is accomplished by introducing compressed air through diffuser hoses to the bottom of lakes. When air bubbles begin to migrate towards the surface, they draw oxygen-depleted water from the bottom, oxygenating it and transporting it to the surface. The compressed air flow also carries water and creates a permanent circular circulation pattern. Research has yielded different results from complete aeration in small, shallow lakes, but the phosphorus budget and the primary production was unaffected by aeration (Labaugh, 1980; Cowell et al., 1987; Sandman et al., 1990). Some researchers have found that the algal biomass was increased and water clarity was decreased by aeration, especially with phosphorus levels above 20 $\mu\text{g L}^{-1}$ (Toetz, 1981; Nürnberg et al., 2003; and Moore and Christensen, 2009).

In this study, we compared the water quality parameters, trophic status and nutrient mass balance of Sheldon Lake and Fossil Creek Lake. Sheldon Lake was restored by complete aeration after bottom sediment removal, and Fossil Creek Lake was restored by complete aeration only. In addition, this study identified additional future lake management and restoration measures when the above methods could not reach the ideal nutrient control targets.

3.2 Lake description and restoration approaches

Fossil Creek Lake, located at an elevation of 1,516 meters of Colorado, is a small shallow aesthetic and recreational amenity to park visitors since 1990s. It is the remnant of the old Portner Reservoir, which was excavated more than 100 years ago. During construction, the reservoir remnant was divided into three connected lakes as shown in Figure 1 by constructing causeways between the East Lake and the Big Lake and between the Big Lake and the West Lake.

Fossil Creek Lake in the remainder of this paper refers to the Big Lake shown in Figure 1. The heavy buildup of organic matters and nutrients, high availability of sulfate coupled with summer and winter thermal stratification, results in sulfate reduction the main process in organic material degradation. The presence of “rotten egg” odor- $\text{H}_2\text{S}(\text{g})$ particularly during lake turnover, occasional fish kills, and gradually deteriorated water quality are the major problems of Fossil Creek Lake. The major water supply to the lake is a diversion pipe from Fossil Creek, which delivers an average flow of $0.02 \text{ m}^3 \text{ s}^{-1}$ from May to October, flowing through a free water surface wetland, and then entering the lake. There is some direct storm runoff (around 5%) from suburban areas near the lakes. The aeration system has been installed and operated in Fossil Creek Lake since August 22th, 2009, with six diffusers in the deepest pockets in the lake (Figure 3-1). The system is powered by two 250W shore-installed air compressors with a total air capacity of $0.5 \text{ m}^3 \text{ min}^{-1}$, and each compressor is connected to three diffusers by weighted rubber pressure pipes. The aeration system has been operated continuously under strictly controlled conditions which enable adjustment of the air flow rate according to the environmental conditions in the lake.

Sheldon Lake, a small, shallow, man-made lake, was excavated for irrigation water storage in 1874 in Fort Collins, Colorado. Like Fossil Creek Lake, Sheldon Lake is a popular recreational place for park visitors, but it has been experiencing different water deterioration problems (Figure 3-2). The direct inflow of Sheldon Lake is piped from Pleasant Valley, Cache la Poudre River and the Colorado-Big Thompson watersheds (Oberholster, 2007). The accumulation of 100 years of bottom sediment had resulted in water quality degradation issues and a decrease of lake volume. Sheldon Lake was dredged in December of 2002 to provide flood protection by the

increasing its stormwater detention capacity, and artificial aeration was introduced in June 2003 to improve the water quality and prevent the growth of cyanobacteria. The lake was mixed artificially by placing aerators on the bottom. The compressed air was passed from the aerators into the water column through a weighted pipe. Three compressors were connected to 21 diffusers (each compressor was connected to 7 diffusers); with a total air capacity of $1.0 \text{ m}^3 \text{ min}^{-1}$ delivery rate. In the summer of 2004, one year after the restoration, an algae bloom was observed in the lake, and Sheldon Lake exhibited a turbid state. Table 3-1 summarizes the morphometric characteristics of both lakes.



Figure 3-1 Map of Fossil Creek Lake with locations of aerators and contour lines



Figure 3-2 Map of Sheldon Lake with locations of aerators and contour lines

Table 3-1 Morphometric characteristics of Fossil Creek Lake and Sheldon Lake

Parameter [unit]	Fossil Creek Lake	Sheldon Lake
Surface area, ha	4.2	5.9
Lake volume, m ³	68,932	133,238
Maximum depth, m	5.5	3.1
Mean depth, m	2.0	2.0
Water retention time, d	40	77

3.3 Methods

Fossil Creek Lake was sampled on a biweekly basis from April to November of 2010 and 2011, except months from December to the following March when the lake is iced over. For Sheldon Lake, water quality parameters after restoration were taken on the same sampling dates as Fossil Creek Lake from July to November in 2010, and from April to November in 2011. There are 11 sampling locations in Fossil Creek Lake, which are the locations of the 6 aerators and midway between each aerator (Figure 3-1). For Sheldon Lake, the sampling locations are located at aerators and midways between each aerator (Figure 3-2). Temperature, dissolved oxygen, pH,

specific conductance and Secchi disk depth were sampled at each sampling site at surface (0.3m depth), middle and bottom in both lakes. Water quality parameters for lab analyses such as TN, TP, RP, TOC, alkalinity, hardness, etc, were sampled and composited across all the sampling sites at each depth (surface and bottom) separately.

For all statistical analyses, values below the detection limit were treated as zero. Two factor analysis of variance (ANOVA) was applied to temperature and dissolved oxygen means from lake surface, middle and bottom of Fossil Creek Lake and Sheldon Lake at each sampling site to check whether the aeration system mixed the entire water column. The single factor ANOVA test was applied to compare the monthly means between 2010 and 2011 in each lake. In Fossil Creek Lake, the monthly means were compared from April to November in 2010 and 2011 respectively. For Sheldon Lake, since sampling of the lake began in mid July of 2010, data for comparison were chosen from July to November of the two study years. We also examined relationships of water quality parameters between Fossil Creek Lake and Sheldon Lake using independent T test analysis, and data were compared from July 2010 to November 2011. All statistical tests were performed using Matlab 2009, and statistical significance was considered to be $p < 0.05$.

3.4 Results

3.4.1 Water quality of source waters

The major water supply to Fossil Creek Lake is a diversion pipe from Fossil Creek (FC), and the direct inflow of Sheldon Lake is piped from Pleasant Valley and Cache la Poudre River (Sheldon). Table 3-2 summarizes the water quality of the source water in both Lakes. There were no significant differences ($p > 0.05$) in TOC, RP, TP and TSS concentrations in source water of both lakes. Hardness, sulfate, IC, alkalinity and TN concentrations were significantly higher

($p < 0.05$) in source water of Fossil Creek Lake than Sheldon Lake. The sulfate levels in Fossil Creek were 30 times higher than the source water of Sheldon Lake, which will highly affect the sulfate levels in Fossil Creek Lake.

Table 3-2 Water quality of source waters in Fossil Creek Lake and Sheldon Lake from 2010 to 2011

Parameter	Water Source	Average	SE	Max	Min
IC (mg L^{-1})	FC	43.42	2.64	63.07	20.43
	Sheldon	11.86	1.61	29.66	6.13
TOC (mg L^{-1})	FC	9.62	1.15	16.59	1.01
	Sheldon	8.12	0.88	14.28	3.55
TN (mg L^{-1})	FC	1.34	0.20	3.36	0.49
	Sheldon	0.52	0.05	1.08	0.26
RP (mg L^{-1})	FC	0.34	0.05	0.73	0.12
	Sheldon	0.25	0.04	0.56	0.06
TP (mg L^{-1})	FC	0.36	0.04	0.77	0.17
	Sheldon	0.38	0.03	0.58	0.19
Hardness (mg L^{-1})	FC	699.09	59.58	1120.00	420.00
	Sheldon	122.36	14.54	237.00	50.00
Alkalinity (mg L^{-1})	FC	194.91	12.93	292.00	86.00
	Sheldon	56.73	7.25	130.00	32.00
Sulfate (mg L^{-1})	FC	867.73	89.73	1675.00	360.00
	Sheldon	26.91	4.12	76.00	12.00
TSS (mg L^{-1})	FC	9.00	1.85	31.20	3.10
	Sheldon	9.77	1.40	22.00	0.40

3.4.2 Physical characterization

Temperature

The ANOVA two factors analysis showed no significant differences ($p > 0.05$) in temperature and dissolved oxygen through the water column (surface, middle and bottom) in each sampling site during the sampling dates of 2010 and 2011 in each lake. It indicates that both of the lakes are destratified and well mixed through the entire water column.

Water temperature at Fossil Creek Lake ranged between 5.4 °C in November to a maximum of 25.5 °C in July in 2010. Mean values and standard deviation of all sample sites at lake surface,

middle and bottom showed that the lake was thermally well mixed on each sampling day (Figure 3-3 a). At Sheldon Lake, differences in mean temperature through the water column were within 1 °C (Figure 3-3 b). Lake water temperature was similar at Sheldon Lake with a range of 8.8 °C in November to 25.5 °C in July 2010.

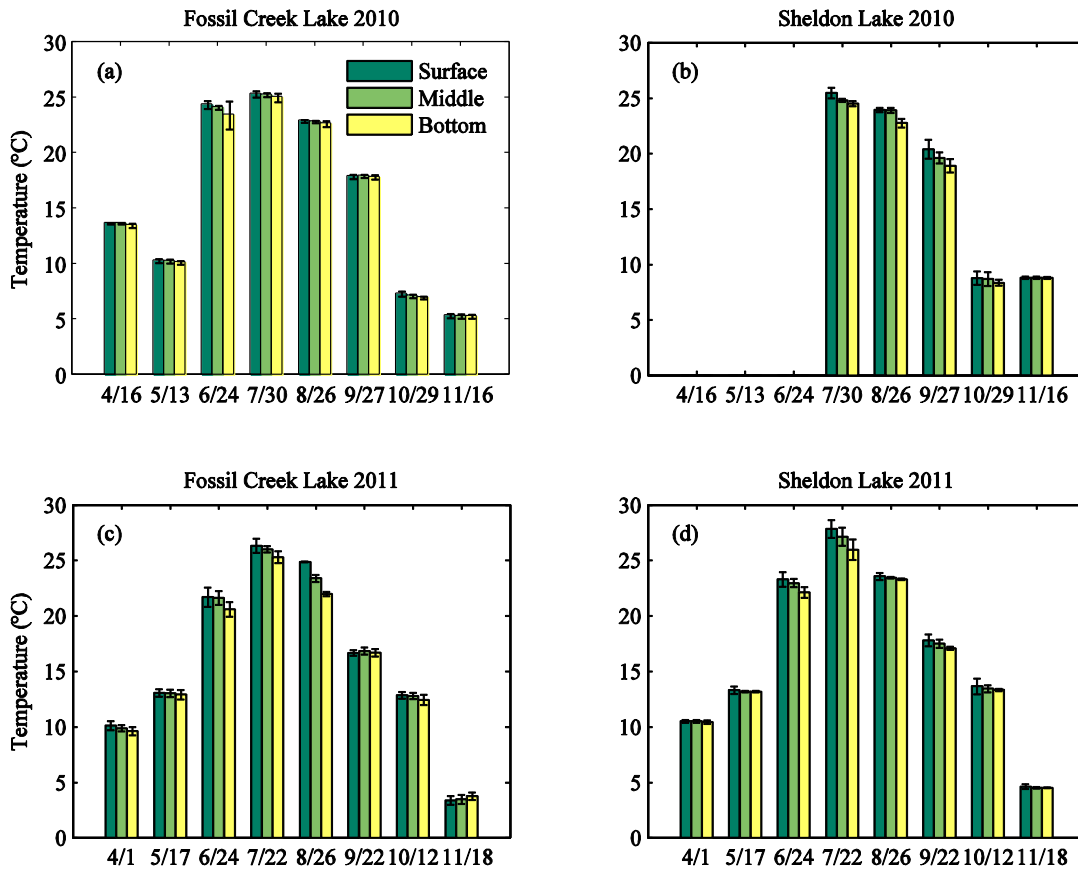


Figure 3-3 Mean water temperature from all sampling sites from surface, middle and bottom at Fossil Creek Lake and Sheldon Lake between 2010 and 2011. Vertical lines at the top of each bar demonstrates the standard deviation values

Similar water temperature records were observed in 2011 in both lakes. Mean water temperature ranged from 4.5 °C in November to 26.3 °C (Figure 3-3 c) in July in Fossil Creek Lake, and varied from 3.4 °C to 27.9 °C in Sheldon Lake (Figure 3-3 d). Differences in mean temperature through the water column were within 1 °C in both lakes. Independent T tests on both lakes in 2010 and 2011 showed no significant temperature differences between these two lakes ($p > 0.05$).

Dissolved oxygen

In general, dissolved oxygen concentrations decreased as water depth increased at both lakes before aeration. With the destratification caused by the aeration systems, the differences between the highest dissolved oxygen concentration at the surface and that of the bottom of the deepest sample point at Fossil Creek Lake and Sheldon Lake were within 3.5 mg L^{-1} of one another during summer time and lower than 1 mg L^{-1} the rest of the year. Dissolved oxygen concentrations were typically lower during the summer and higher during spring and fall (Figure 3-4).

At Fossil Creek Lake, the mean and standard deviations of all measurements from all sample sites between surface, middle and bottom of the lakes for each sampling show November to have the highest concentrations at 13.6 mg L^{-1} (Figure 3-4 a) and 13.5 mg L^{-1} (Figure 3-4 c) in 2010 and 2011 respectively. Minimum dissolved oxygen values monitored at the lake bottom at the end of July of 2010 and 2011, were around 5 and 6 mg L^{-1} . The dissolved oxygen profile in Sheldon Lake demonstrates that the maximum results were in late October of 2010 at 12.9 mg L^{-1} (Figure 3-4 b) and November of 2011 at 13 mg L^{-1} respectively (Figure 3-4 d). The lowest values were monitored in the lake bottom in late July of 2010 and late August in 2011, at around 5 mg L^{-1} .

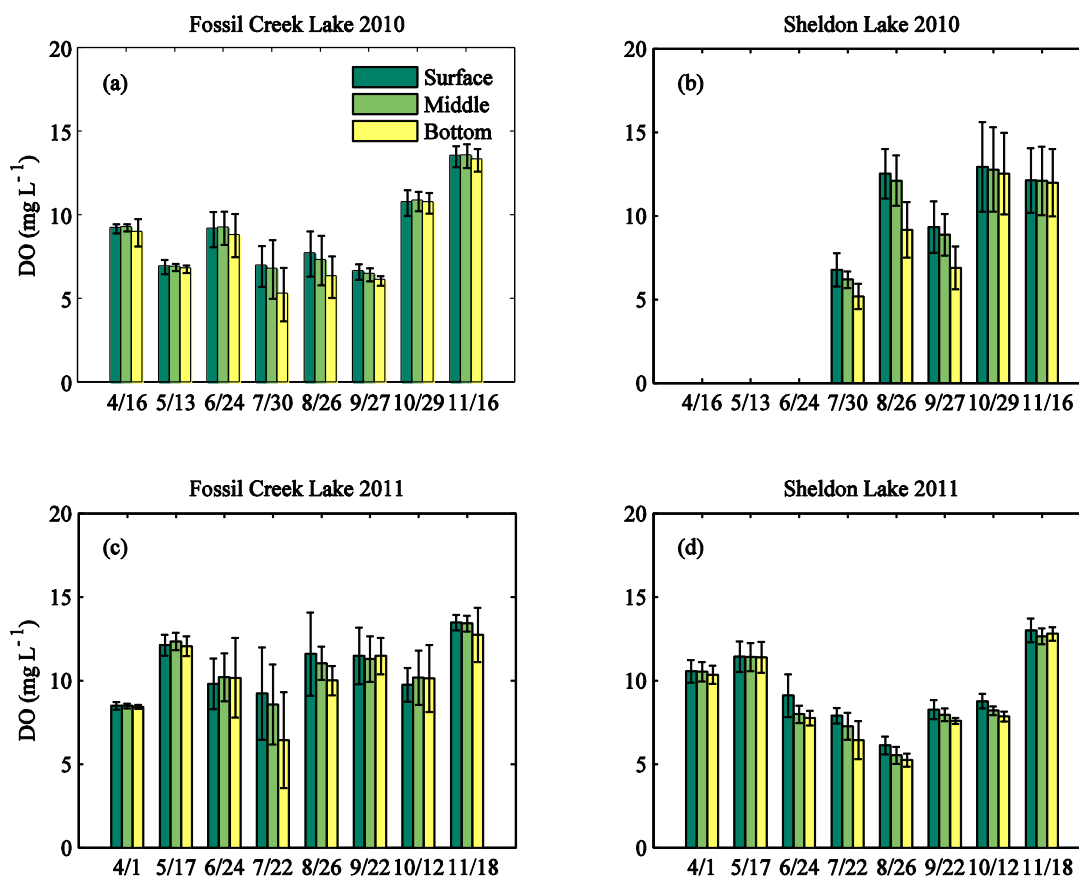


Figure 3-4 Mean dissolved oxygen concentrations from all sampling sites from surface, middle and bottom at Fossil Creek Lake and Sheldon Lake between 2010 and 2011

3.4.3 Chemical characterization

Composite samples from surface and bottom at different sampling locations were measured in 2010 and 2011 for the following water quality parameters: total inorganic carbon, alkalinity, hardness, sulfate, specific conductance, total suspended solids, chlorophyll a, and Secchi disk.

pH, total inorganic carbon and alkalinity

The monthly mean pH values of Fossil Creek Lake in 2010 were steady following the installation of the aeration system, ranging from 8.2 to 8.8 with peak values in July. In the following year, pH increased significantly ($p < 0.05$), and was in the range of 8.3 to 9.4 with peak values in July with submerged macrophyte photosynthesis and loss of inorganic carbon due to the aeration system (Figure 3-5 a). Concentrations of inorganic carbon were 26-62.44 mg L⁻¹ and

15.3-55.5 mg L⁻¹ in 2010 and 2011, respectively. Aeration produced a significant decrease in inorganic carbon in 2011 ($p < 0.05$). Peak values were observed in April in both lakes, after which the concentrations kept decreasing from April to July and then increased slightly from July to November (Figure 3-5b). Alkalinity followed the same general trend as total inorganic carbon during the two years of aeration, and alkalinity levels were significantly reduced ($p < 0.05$) in 2011 (Figure 3-5c).

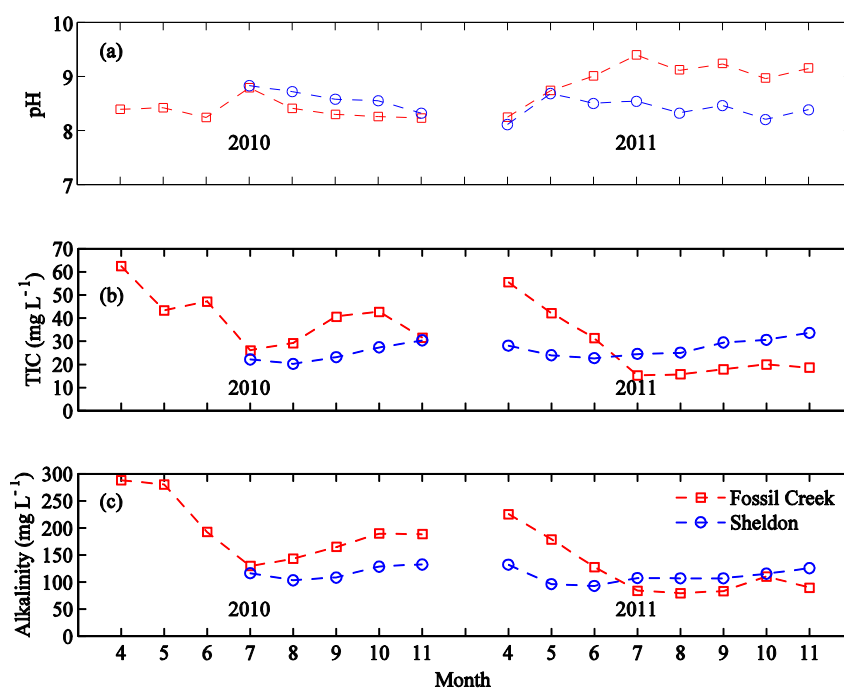


Figure 3-5 Monthly mean values of pH, total inorganic carbon and alkalinity in Fossil Creek Lake and Sheldon Lake

At Sheldon Lake in 2010, the seventh year of aeration, monthly mean pH values ranged from 8.1 to 9.0 and peak values, which were monitored in July, were attributed to phytoplankton bloom. The monthly average pH values were in the range of 8.1 to 8.7 in 2011. The difference between the pH values from July to November in 2011, the eighth year of aeration, and those in 2010 were not significant ($p > 0.05$). Unlike Fossil Creek Lake, the variations of the inorganic carbon and alkalinity between 2010 and 2011 were insignificant ($p > 0.05$). Independent T test results indicated that total inorganic carbon and alkalinity values from the two lakes varied

insignificantly ($p>0.05$) from each other in these two years but for pH, where there was a significant difference ($p<0.05$).

Hardness, sulfate and specific conductance

In Fossil Creek Lake, mean hardness concentrations were 861-1497mg L⁻¹ and 576-1473 mg L⁻¹ in 2010 and 2011, respectively. Hardness decreased from April to July, and then kept increasing in the following months, reaching peak value in November in 2010. Hardness levels in 2011 declined from April to September, and increased in the following three months. Sulfate varied from 1023 to 1533mg L⁻¹ in 2010, and was in the range of 867 and 1890 mg L⁻¹ in 2011, with peak values monitored in June of each year (Figure 3-6a). Peak values of SPC were observed during summer time, July in 2010 and late June of 2011 at 2937 and 2949 $\mu\text{S cm}^{-1}$. Specific conductance values kept decreasing from late June or July to November. The changes in the three water quality parameters for the two years in Fossil Creek Lake were insignificant ($p>0.05$). The changes of hardness and sulfate were highly related to the lake water supply from Fossil Creek, which passes through groundwater rich in sulfate and hardness.

Average values of hardness, sulfate and SPC in Sheldon Lake showed remarkable differences from those of Fossil Creek Lake ($p<0.05$). The variations of the three parameters from July to November in the two years of aeration were insignificant ($p>0.05$). Hardness levels were in the range of 132 to 281mg L⁻¹; sulfate varied from 32 to 75 mg L⁻¹, while SPC ranged from 315 to 418mg L⁻¹ in 2010 and 2011. The three parameters all followed the same patterns, when peak values were observed in April and the concentrations kept decreasing until July or August, after which the levels of the three parameters increased gradually until November (Figure 3-6b).

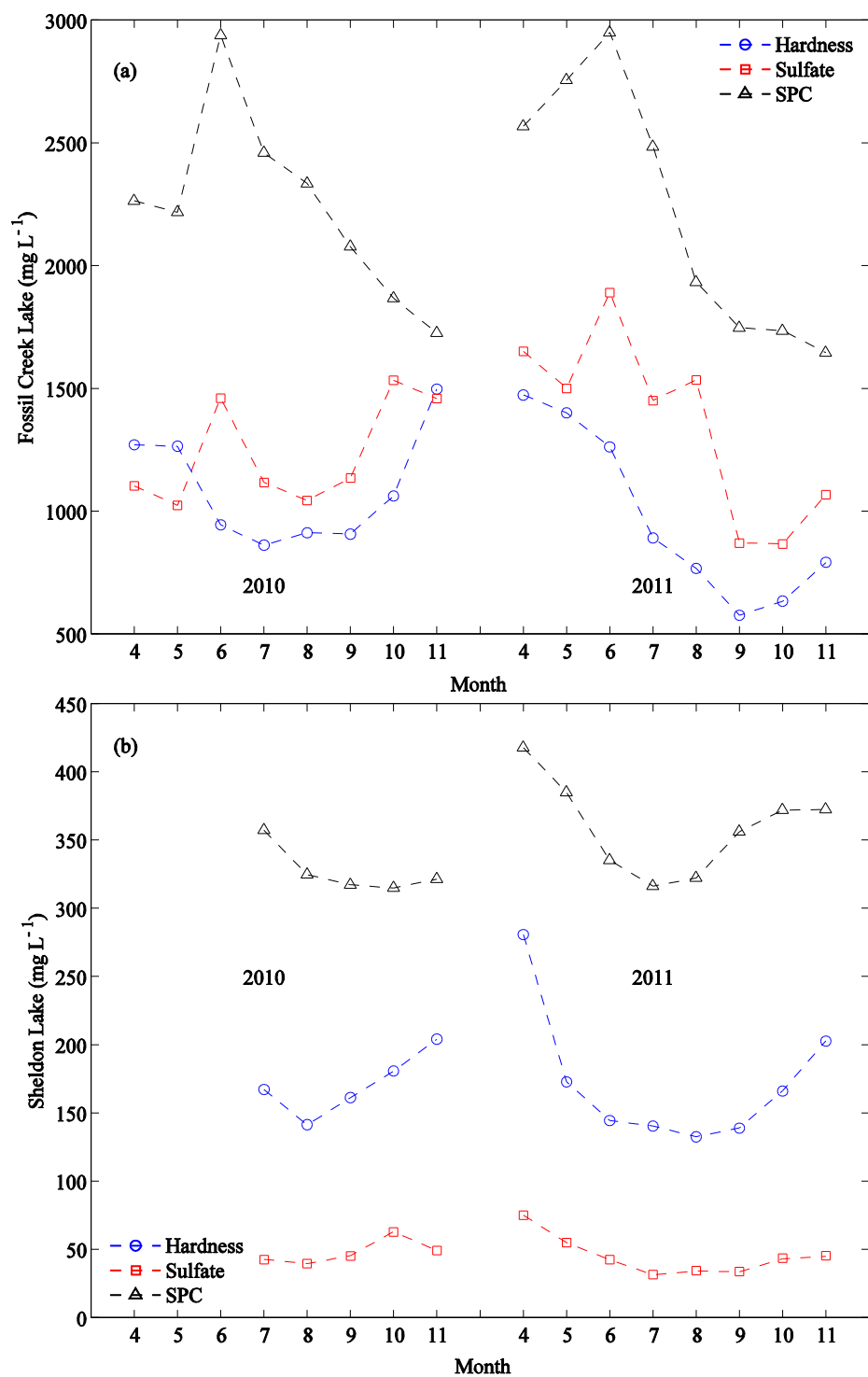


Figure 3-6 Monthly mean hardness, sulfate and specific conductance values of Fossil Creek Lake and Sheldon Lake between July 2010 and November 2011

Total suspended solids, chlorophyll a and Secchi depth

Total suspended solids levels in Fossil Creek Lake were in the range of 2.5 to 8.2 mg L⁻¹ (Figure 3-7a), which was significantly different from the condition in Sheldon Lake ($p < 0.05$) at 7.7 to 23.8 mg L⁻¹ (Figure 3-7 b). Similar large variations in chlorophyll a concentrations between the two studied lakes were observed ($p < 0.05$). The chlorophyll a levels at Fossil Creek Lake were between 22 and 34 µg L⁻¹, while most results monitored in Sheldon Lake were in the range of 30 and 50 µg L⁻¹ with peak values at 92 µg L⁻¹, which indicated that the phytoplankton bloom was more severe in Sheldon Lake. The Secchi disk depth range at Fossil Creek Lake was 0.68 to 1.76 meters while the range at Sheldon Lake varied from 0.25 to 0.55 meters in the two years of research. Statistical analysis results showed that the Secchi disk transparency in Sheldon Lake was significantly lower than that of Fossil Creek Lake ($p < 0.05$). The difference of lake transparency was attributed to the aquatic plants in the two lakes.

Lake nutrient, limiting nutrient and trophic status

At Fossil Creek Lake, total nitrogen concentrations were 0.64-1.82 mg L⁻¹ (mean 1.25 mg L⁻¹) in the two years of research, and the variation between the two years was insignificant ($p > 0.05$). There was little change ($p > 0.05$) in total phosphorus levels from 2010 to 2011, which were in the range of 0.19-0.65 mg L⁻¹ (mean 0.37 mg L⁻¹). Similarly, both orthophosphate and total organic carbon concentrations in Fossil Creek Lake showed insignificant differences from 2010 to 2011 ($p > 0.05$), which varied from 0.07 to 0.36 mg L⁻¹ and from 5.0 to 19.5 mg L⁻¹ respectively (Figure 3-8a).

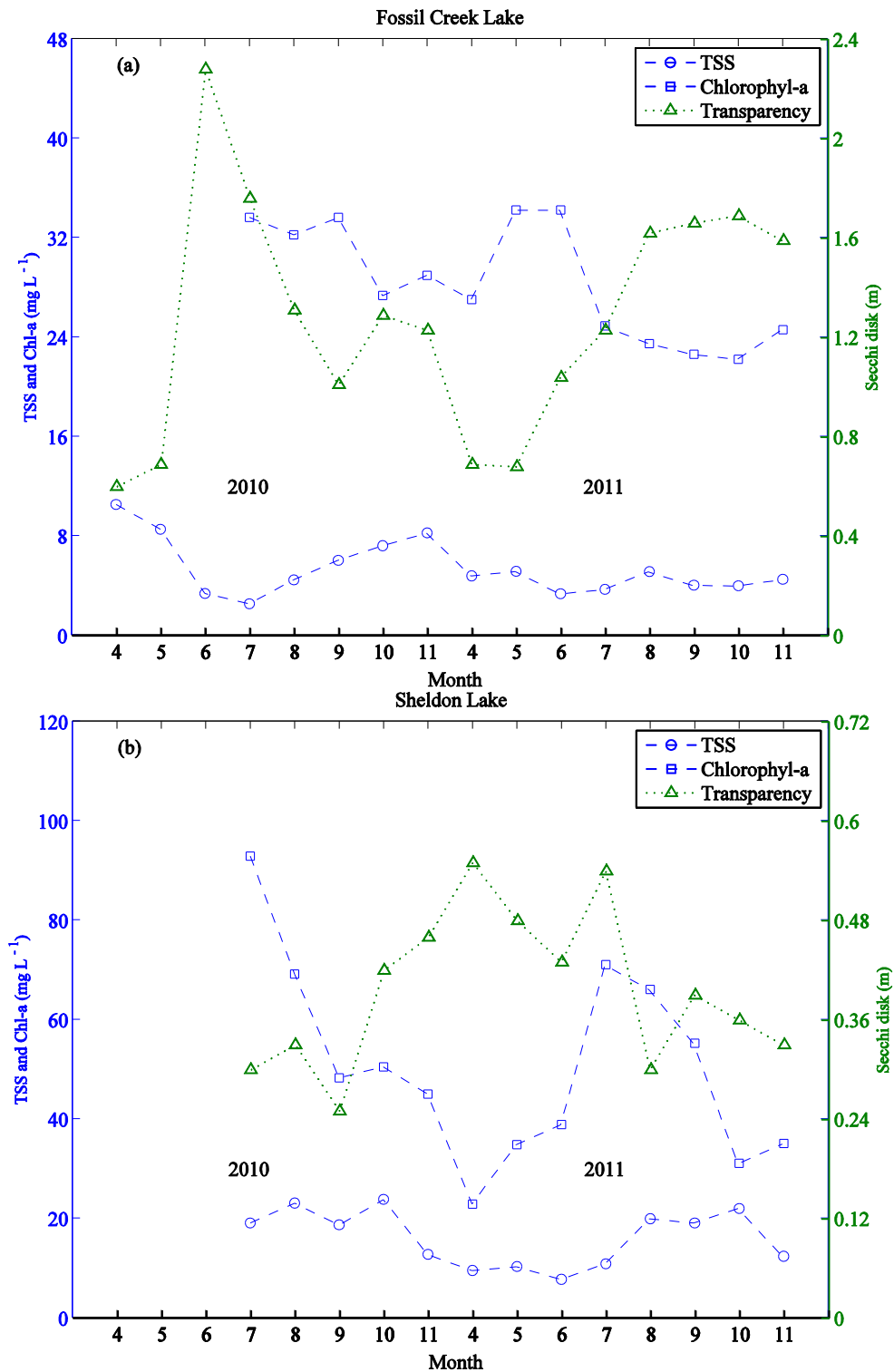


Figure 3-7 Monthly average values of total suspended solids, chlorophyll a and secchi depth of Fossil Creek Lake and Sheldon Lake between July 2010 and November 2011

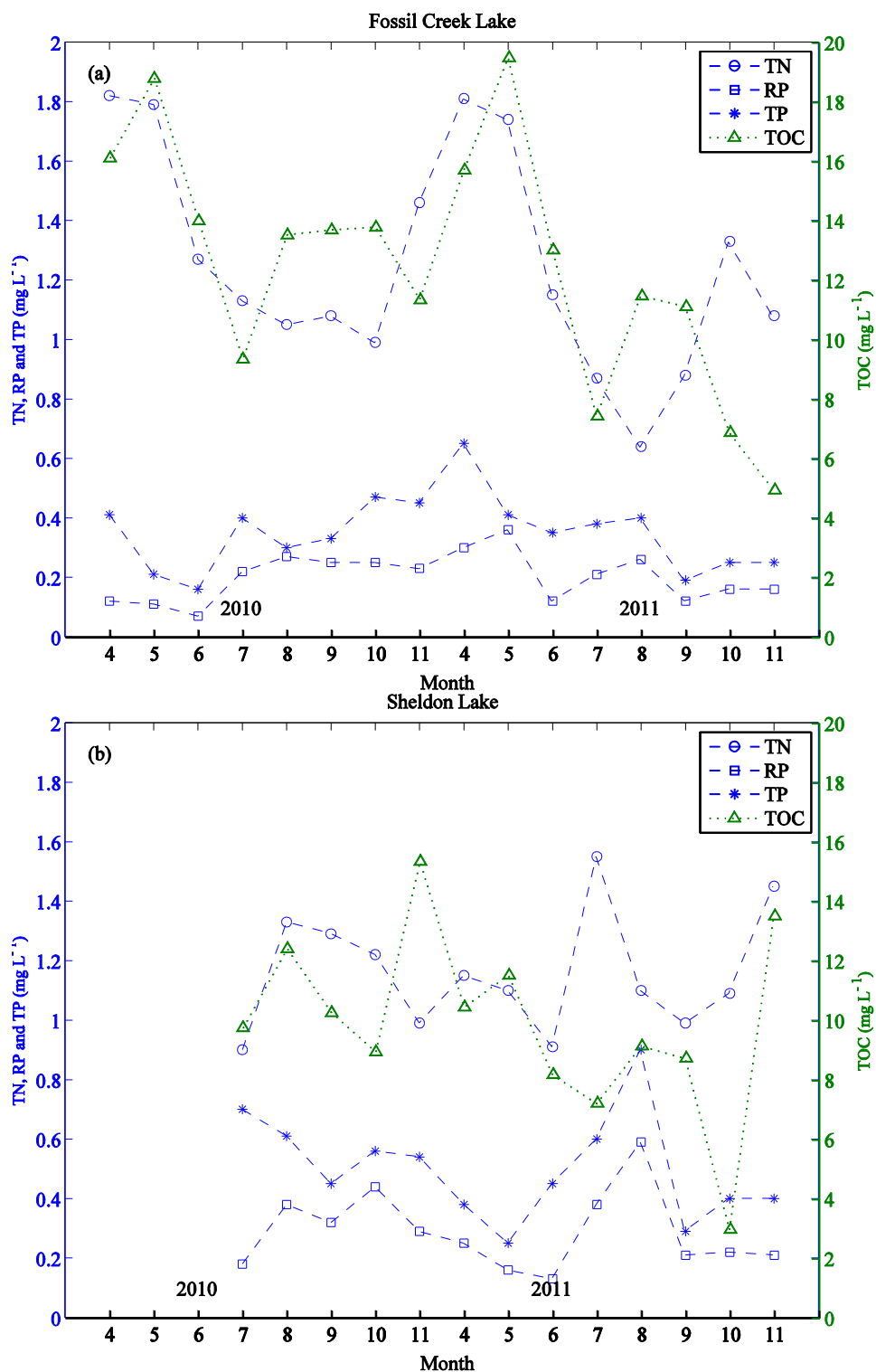


Figure 3-8 Monthly average nutrient levels at Fossil Creek and Sheldon lakes between 2010 and 2011

Like Fossil Creek Lake, the variations of the four water quality parameters were insignificant from July to November in both years in Sheldon Lake ($p > 0.05$). Total nitrogen was 0.9-1.55 mg

L⁻¹; total phosphorus ranged from 0.25 to 0.9 mg L⁻¹ and orthophosphate varied from 0.13 to 0.59 mg L⁻¹ while total organic carbon was in the range of 2.99-15.37 mg L⁻¹ (Figure 3-8 b). There were no significant differences between Fossil Creek Lake and Sheldon Lake in total nitrogen, orthophosphate and total organic carbon (p>0.05). However, the total phosphorus concentrations in Fossil Creek Lake were significantly lower than those in Sheldon Lake (p<0.05).

Ammonia and nitrate concentrations in both lakes from 2010 to 2011 are summarized in Table 3-3 and 3-4 respectively. Average ammonia levels in both lakes in 2010 were around 30-40 µg L⁻¹ while nitrate concentrations were between 46 and 60 µg L⁻¹ in these two lakes. In the following years, ammonia levels dropped to the range of 22-29 µg L⁻¹, while the range of nitrate increased to 35-78 µg L⁻¹.

Table 3-3 Ammonia and nitrate for Fossil Creek and Sheldon Lakes in 2010

Parameters	Lake	n	min	median	mean	max
Ammonia (µg L ⁻¹)	Fossil Creek	27	10	20	39	200
	Sheldon	13	20	45	30	90
Nitrate+Nitrite (µg L ⁻¹)	Fossil Creek	27	20	50	60	370
	Sheldon	13	20	48	46	70

Table 3-4 Ammonia and nitrate for Fossil Creek and Sheldon Lakes in 2011

Parameters	Lake	n	min	median	mean	max
Ammonia (µg L ⁻¹)	Fossil Creek	14	9	30	29	60
	Sheldon	14	10	20	22	40
Nitrate +Nitrite (µg L ⁻¹)	Fossil Creek	14	10	30	78	300
	Sheldon	14	10	20	35	100

3.4.4 Nutrient characterization

Nutrient ratio

Nutrient ratios were used to evaluate the nutrient limiting to primary production. In general, concentrations of biologically available phosphorous (PO₄³⁻) of less than 5 µg L⁻¹ indicate

potential PO_4^{3-} limitation, while concentrations of biologically available nitrogen (inorganic-N) of less than $20\mu\text{g L}^{-1}$ indicate inorganic nitrogen limitation (Ryding and Rast 1989). When $\text{TN:TP} < 20$ (molar based), a lake is considered N limited and when it is above 38, it is considered P limited (Sakamoto 1966 and Kosten et al., 2009). Although the ratio of N: P changes during the course of the growing season, as a general approximation, the annual means of TN and TP were used to estimate the TN: TP ratios in order to describe overall conditions. Total nutrient ratios were lower than 20, while the bioavailable nutrient ratios were lower than 4 in both lakes during the two-year research (Table 3-5), indicating that both of the lakes are nitrogen limited.

Table 3-5 Total and bioavailable nitrogen and phosphorus ratios at Fossil Creek and Sheldon Lakes in 2010 and 2011

Year	Fossil Creek Lake		Sheldon Lake	
	TN:TP	DIN*: RP	TN:TP	DIN: RP
2010	19.92	8.21	13.60	1.79
2011	19.70	3.24	17.26	1.16

*DIN: dissolved inorganic nitrogen

Phosphorus balance

According to the modified One-Box model (Gächter and Imboden 1985, Sas 1989, and Schauser and Chorus 2007), the phosphorus balance of a lake is determined by the input (M_{in}), the export through the outlet (M_{out}), and lake net sedimentation (M_{sed}). The phosphorus input M_{in} can be estimated by the inflow weighted average phosphorus concentration of all inlets ($P_{\text{in},i}$) and the inflow rate, $Q_{\text{in},i}$. Similarly, the phosphorus output M_{out} is estimated by the outflow weighted average phosphorus concentration of all outlets ($P_{\text{out},j}$) and the outflow rate, $Q_{\text{out},j}$. The phosphorus content of the lake (M_{lake}) is the product of the lake average phosphorus concentrations (P_{lake}) times the lake volume (V). The net sedimentation M_{sed} can be calculated based on the phosphorus mass balance of the Lake from Equation 3-1.

$$M_{\text{lake}} = M_{\text{in}} - M_{\text{out}} - M_{\text{sed}} \quad \text{Equation 3-1}$$

Where

$$M_{\text{lake}} = P_{\text{lake}} * V$$

$$M_{\text{in}} = P_{\text{in}} * Q_{\text{in}}$$

$$M_{\text{out}} = P_{\text{out}} * Q_{\text{out}}$$

$$P_{\text{lake}} = \text{Lake average phosphorus concentrations}$$

$$V = \text{Lake volume}$$

$$P_{\text{in}} = \text{Phosphorus concentration in inflow}$$

$$P_{\text{out}} = \text{Phosphorus concentration in outflow}$$

$$Q_{\text{in}} = \text{Inflow rate}$$

$$Q_{\text{out}} = \text{Outflow rate}$$

The annual phosphorus mass balance in both lakes before and after restoration is calculated based on Equation 3-1. Appendix C-1 to C-4 show the detailed calculation process and the final results are summarized in Table 3-6.

Table 3-6 Total phosphorus balance in Fossil Creek and Sheldon Lakes

Lake	Fossil Creek Lake			Sheldon Lake		
	Pre-restoration	2010	2011	Pre-restoration	2010	2011
P balance						
Stormwater runoff load, kg	46.0	48.0	61.2	184.4	221.8	171.7
Irrigation water load, kg	151.5	167.9	133.8	273.9	430.4	350.4
Net sedimentation, kg	-61.5*	101.7	17.3	-189.8*	174.5	-43.8*
Outlet export, kg	155.6	90.1	152.9	407.0	399.1	498.0
Lake P contents, kg	103.4	24.1	24.8	241.2	78.6	68.0

*Negative values in net sedimentation indicate that the sediments act as annual P sources.

In Fossil Creek Lakes, the major input of phosphorus is from Fossil Creek and stormwater runoff of the surrounding residential areas, which occupy around 20 ha. Another source of phosphorus is that released from bottom sediment especially during summer time when the lake bottom was anoxic. Before restoration, roughly 76% of the phosphorus load originated from irrigation water

and stormwater runoff, with the remaining phosphorus load coming from the bottom sediment, which released 61.5 kg phosphorus per year, resulting in 103.4 kg phosphorus each year in the lake (Table 3-6). Aeration in 2010 and 2011 not only reduced the phosphorus loading from bottom sediment, but also increased the net sedimentation of phosphorus. As a result, the phosphorus contents in lake were reduced by a factor of 4.3 compared with the pre-restoration condition. The less irrigation load and higher outlet export in both lakes in 2011 result in lower net sedimentation than year 2010.

Similar to Fossil Creek Lake, the incoming irrigation water and stormwater runoff are the two main contributions to phosphorus loading in Sheldon Lake. However, the areas contributing stormwater runoff to Sheldon Lake are around 80 ha, which is 4 times as large as the contributing area of Fossil Creek Lake. The phosphorus releasing from bottom sediment accounted for 30% of the total phosphorus loading before restoration. The application of aeration after dredging significantly reduced the resuspension of phosphorus from bottom sediment, and the phosphorus contents in the lake were 3 times lower than the pre-restoration conditions (Table 3-6).

Trophic status

The trophic states of the two lakes were determined by: Carlson's TSI (1977) model (Figure 1-3), the OECD fixed boundary system (Table 1-1); EPA National Eutrophication Survey 1974 (Table 1-1); and Vollenweider's phosphorus loading plots (Figure 1-4). In Carlson's TSI model, TSI values represent the trophic state as a number from 0 to 100 in order to quantify trophic status and offer more than four descriptive categories for trophic state, which are oligotrophic, mesotrophic, eutrophic and hypereutrophic. Chlorophyll a concentrations had the highest TSI in

both lakes at around 72 to 80 (Table 3-7), which indicated the lakes are in hyper-eutrophic condition. Secchi disk transparency gave the lowest TSI in Fossil Creek Lake at values around 55 representing eutrophic condition (Table 3-7), but the second highest TSI in Sheldon Lake which was between 72 and 75, also indicating eutrophic condition (Table 3-7). The TSI of total nitrogen and total phosphorus was in the range of 56.9 to 60.29 in both lakes, indicating eutrophication of the two lakes in 2010 and 2011 (Table 3-7). Based on the OECD fixed boundary system and the national eutrophic survey results, both of the lakes were hypereutrophic for each parameter. Vollenweider's loading plots use the total input of phosphorus per year per surface area and the ratio between mean lake depth and hydraulic residence time to define the lake trophic status. According to this principal, both of the lakes can be classified as eutrophic.

Table 3-7 Fossil Creek Lake and Sheldon Lake trophic classification summary in 2010 and 2011

Classification Model	Parameters	Lake	TSI			Trophic Status
			Pre	2010	2011	
Carlson's TSI	TP	FCL	59.59	57.97	58.06	Eutrophic
		Sheldon	59.62	58.79	58.49	Eutrophic
	Chlorophyll a	FCL	68.75	73.33	71.80	Hyper-eutrophic
		Sheldon	72.52	79.95	76.79	Hyper-eutrophic
	Secchi disk	FCL	56.21	56.55	56.44	Eutrophic
		Sheldon	69.71	75.15	72.52	Eutrophic
	TN	FCL	65.83	60.29	56.90	Eutrophic
OECD fixed boundary	TP	Sheldon	66.47	56.42	56.69	Eutrophic
		FCL	>100	>100	>100	Hyper-eutrophic
	Chlorophyll a	Sheldon	>100	>100	>100	Hyper-eutrophic
		FCL	>25	>25	>25	Hyper-eutrophic
	Secchi disk	Sheldon	>25	>25	>25	Hyper-eutrophic
National eutrophic Survey	Chlorophyll a	Sheldon	<1.5	<1.5	<1.5	Hyper-eutrophic
		FCL	<1.5	<1.5	<1.5	Hyper-eutrophic
	Secchi disk	Sheldon	<1	<1	<1	Hyper-eutrophic
		FCL	<1	<1	<1	Hyper-eutrophic
Vollenweider's loading plot	Lp, gP m ⁻² yr ⁻¹	Sheldon	>25	>25	>25	Hyper-eutrophic
		FCL	>25	>25	>25	Hyper-eutrophic
	H/τ, m yr ⁻¹	Sheldon	1-2	1-2	1-2	Eutrophic
		FCL	>18.18	18.18	18.18	Eutrophic

*FCL: Fossil Creek Lake

*Sheldon: Sheldon Lake

*H: lake depth; τ : residence time

3.5 Discussion

3.5.1 Comparison of physical and chemical parameters

Complete aeration of the entire Fossil Creek Lake as well as Sheldon Lake greatly altered the limnology of both lakes in general, and their oxygen and temperature regimes in particular. As a result of aeration, thermal stratification was eliminated by mixing the entire water column and the oxygen concentrations were greatly increased (above 5 mg L^{-1}) in both lakes during summer time. The independent T test results showed that there were no significant differences between the two lakes in monthly means of temperature and dissolved oxygen.

The concentrations of TN, TP and chlorophyll a were significantly reduced by complete aeration in both lakes (Table 3-8). The differences of Secchi depth in Fossil Creek Lake before and after aeration were insignificant ($p > 0.05$), while Secchi depth in Sheldon Lake decreased from 2001 without mixing, to 2010-2011 with mixing. Chlorophyll a and TSS concentrations were significantly higher in Sheldon Lake than in Fossil Creek Lake ($p < 0.05$). As a result, the Secchi disk transparency of Sheldon Lake, which rarely exceeded 0.5 m, was significantly lower than that of Fossil Creek Lake ($p < 0.05$) during the two-year study. The main reason for the decreased water transparency and clarity in Sheldon Lake after aeration is probably due to the loss of submerged macrophytes during dredging, which resulted in Sheldon Lake becoming a phytoplankton dominated lake. The resuspension of bottom sediment by aeration system in Sheldon Lake may be the other main reason.

The significant rise in average pH and the decrease in average alkalinity and total inorganic carbon in 2011 compared with year 2010 ($p < 0.05$) were what would be anticipated from studies on naturally stratified lakes during the first several years of aeration. The aeration of water drives CO_2 removal rapidly, and breaks the equilibrium of $\text{H}_2\text{CO}_3/\text{HCO}_3^-/\text{CO}_3^{2-}$ system, which is a very useful buffer system for maintaining the water pH. Furthermore, the uptake of CO_2 by photosynthesis, combined with the loss of CO_2 via the aerator in the water, resulted in an increase in pH of the lake water. Unlike Fossil Creek Lake, after seven years aeration, the pH, alkalinity and TIC concentrations were stable in the two years of study, and there were no significant changes ($p > 0.05$) in Sheldon Lake. Independent T test results showed that although there were remarkable pH variations between the two lakes from July of 2010 to November 2011, the variations of alkalinity and TIC were insignificant ($p > 0.05$).

Table 3-8 Annual mean values for nutrient, phytoplankton biomass and Secchi depth before and after restoration of Fossil Creek Lake and Sheldon Lake

Variable	Fossil Creek Lake			Sheldon Lake		
	2009 Before restoration	2010	2011	2001 Before restoration	2010	2011
TN (mg L^{-1})	6.43	1.5	1.18	2.28	1.15	1.17
TP (mg L^{-1})	2.35	0.34	0.36	1.81	0.57	0.46
Chlorophyll a ($\mu\text{g L}^{-1}$)	30.4	24.89	24.48	72	61.1	44.31
Secchi depth (m)	1.33	1.42	1.28	0.51	0.35	0.42

There were no significant variations in monthly means of hardness, sulfate and specific conductance in the two-year study of each lake ($p > 0.05$). Despite the small change in these parameters in each individual lake, there were statistically significant differences in the monthly means between the two lakes ($p < 0.05$). The three parameters in Fossil Creek Lake were significantly higher than those in Sheldon Lake ($p < 0.05$) in every month during 2010 and 2011. The lake sulfate concentration is greatly influenced by the local geochemical background of the

feeding water from Fossil Creek, which passes through a sulfate rich groundwater, causing extremely high sulfate levels in Fossil Creek Lake at around 1000 mg L^{-1} . Based on the alkalinity content (less than 300 mg L^{-1} as CaCO_3), which was much lower than the hardness level (1000 mg L^{-1} as CaCO_3), the hardness of Fossil Creek Lake is mostly constituted by Ca^{2+} and Mg^{2+} , which also indicates that the sulfate of the lake is mainly calcium and magnesium sulfate. As known, elevated sulfate concentrations such as those in Fossil Creek Lake are favorable for the growth of sulfate reducing bacteria and the production of hydrogen sulfide where anoxic condition exists and organic matter is present.

There were no significant differences between the two lakes in TN, RP and TOC ($p > 0.05$) in this study. However, TP concentrations in Fossil Creek Lake were significantly ($p < 0.05$) lower than those in Sheldon Lake (Figure 3-8). According to Carlson's TSI and Vollenweider's loading plots, both of the lakes can be classified as eutrophic. Based upon the OECD fixed boundary system and the national eutrophic survey results, however, the two lakes were hypereutrophic during the two years of aeration. This conclusion indicates that neither the two-year aeration in Fossil Creek Lake nor dredging after seven years of aeration in Sheldon Lake have effectively reduced eutrophication.

3.5.2 Eutrophication management synthesis

Eutrophication generally promotes excessive plant growth and decay, favoring simple algae and plankton in many freshwater ecosystems. It often has two pronounced negative effects on the oxygen concentration and animal life in a lake when algal bloom becomes excessive. First of all, the excessive phytoplankton consumes a large amount of oxygen during dark phase respiration. The oxygen in the lake may drop to a level that causes stress or death in a population of fish,

especially in hot summer months as the lake heats up and the reduces oxygen saturation levels. The second negative effect of excessive bloom on a lake is a sudden death of phytoplankton population. If the phytoplankton biomass is suddenly killed, it will sink to the lake bottom and decompose rapidly, which could strip oxygen from the lake water to kill a fish population.

The reduction of in-lake P has been generally considered the key to recovery of a eutrophic lake. However, both of the lakes are nitrogen limited lakes. Would the limiting nutrient in the lake be the appropriate choice for reduction in the inflow to reduce the lake's trophic state? However, there is not any published case that demonstrates the effectiveness of N-only reduction, or for the necessity of N reduction in addition to P reduction (Welch, 2009). Lewis and Wurtsbaugh (2008) concluded for management purposes that the answer is P. Evidence from lake responses to manipulations of input nutrients also points to P as the appropriate nutrient to control in order to reduce trophic state. Previous research has shown that algal biomass begins to decline only after soluble reactive phosphorus (SRP) reaches $<10 \mu\text{g L}^{-1}$ (Sas et al. 1989).

A lake can be restored to a stable clear-water state by reducing the total phosphorus level to between 50 and 100 $\mu\text{g L}^{-1}$, and when the SRP reaches lower than 10 $\mu\text{g L}^{-1}$. The commonly used phosphorus management approaches include external phosphorus load reduction as well as internal phosphorus precipitation and inactivation. Target in-lake total phosphorus concentration for both lakes was determined to be about 50 $\mu\text{g L}^{-1}$.

External measures on phosphorus

The current phosphorus loads to both lakes are high. Fossil Creek Lake's load averaged about 4.9 $\text{g m}^{-2} \text{yr}^{-1}$ in 2010-2011 and Sheldon Lake mean value was 9.2 $\text{g m}^{-2} \text{yr}^{-1}$ in 2010-2011. To plan

the external phosphorus control measures, the Vollenweider (1976) model for area loading provided the basis for calculation of the critical external loads (Equation 3-2), a statistical model that does not explicitly consider phosphorus retention in the sediment (See Table 3-1 for parameter values).

$$LP_{crit} = CP_{target} * (1 + \sqrt{\tau}) * z_m / \tau \quad \text{Equation 3-2}$$

Where

$$\begin{aligned} Lp_{crit} &= \text{critical P load [g m}^{-2} \text{ yr}^{-1}] \\ CP_{target} &= \text{target P inlake concentration [g m}^{-3}] \\ \tau &= \text{water retention time [yr]} \\ z_m &= \text{mean lake depth [m]} \end{aligned}$$

The Vollenweider model indicates that 1.21 g m⁻² yr⁻¹ for Fossil Creek Lake and 0.69 g m⁻² yr⁻¹ for Sheldon Lake should not be exceeded to achieve the target total phosphorus in-lake concentration. The larger critical load for Fossil Creek Lake follows from its shorter retention time than that of Sheldon Lake.

Because the modified One-Box model takes internal phosphorus loading via a net sedimentation factor and the export through outflow into account, it presents a different model to estimate critical load (Equation 3-3) from the Vollenweider model. The phosphorus balance of a lake is determined by the input, the export through the outlet, and net sedimentation. The outflow factor can be estimated as the ratio of the annual mean phosphorus concentration in the outflow to the average phosphorus concentration of the entire lake. The sedimentation factor can be estimated from the steady state solution of the One-Box model (Equation 3-4). In lakes with annual phosphorus retention in the sediment, the net sedimentation factor is larger than 0, whereas in

lakes with sediments acting as annual phosphorus sources, the factor is negative. Typically, the factor ranges from -2 to +4 (Schauser and Chorus, 2007).

$$LP_{crit} = CP_{target} * (\beta + \sigma * \tau) * Z_m / \tau \quad \text{Equation 3-3}$$

$$\sigma = (CP_{in} / \tau - CP_{lake} * \beta / \tau) / CP_{lake} \quad \text{Equation 3-4}$$

Where

CP_{lake} = P concentration in the lake [$mg\ L^{-1}$]

CP_{in} = P concentration in the inflow [$mg\ L^{-1}$]

β = outflow factor (mean concentration in outflow/mean lake concentration)

σ = net sedimentation factor

τ = water retention time [year]

For Fossil Creek Lake, the One-Box model indicates a necessary external load reduction from 4.9 to 0.88 $g\ m^{-2}\ yr^{-1}$, which is 27% more reduction than the result of the Vollenweider model because phosphorus retention in the sediment is relatively low in the lake. Similarly, the low sediment retention in Sheldon Lake results in 42% additional external load reduction estimated by the One-Box model than the Vollenweider model, which is reducing from 9.2 to 0.40 $g\ m^{-2}\ yr^{-1}$.

Internal measures on phosphorus

It is believed that in-lake phosphorus control approaches can accelerate lake recovery periods, especially in shallow lakes. Because the effectiveness of the external load control could take substantial time to realize, lake water quality will not improve immediately depending on the amount of accumulated nutrients and organic matter in the bottom sediment. Rather than reducing the external load of phosphorus to achieve the restoration target, commonly used internal techniques include phosphorus losses to the sediment and phosphorus loss to the outflow. The P loss in lake sediment can be achieved by removal of P from the water column (P

precipitation) and by retarding P release from lake sediments (P inactivation) through an aluminum salt, usually aluminum sulfate (alum), sodium aluminate, alkalinity, a precipitate of aluminum phosphate or colloidal aluminum hydroxide, to which certain P fractions are tightly bound. Phosphorus export with the outflow can be increased by hypolimnetic withdrawal. However, both of the study lakes are completely mixed by aeration systems, so this approach will not be useful. The net sedimentation factor was used to estimate the total loss of phosphorus to the sediment, which was calculated with the modified One-Box model.

Fossil Creek Lake results show that when increased P retention is chosen as the only restoration measure, the net sedimentation factor would have to increase from 1.9 to 57.6 yr⁻¹, around 30 times to reach the restoration target. For Sheldon Lake, the net sedimentation factor has to change from -0.5 to 47.8 yr⁻¹.

3.6 Conclusions

The primary objective of this study was to compare the water quality parameters and how they changed over the two-year study in Fossil Creek Lake and Sheldon Lake by different restoration methods. Fossil Creek Lake was studied during the first and second year of complete aeration, while Sheldon Lake was studied during the seventh and eighth years of aeration after dredging.

The aeration systems in both lakes successfully eliminated thermal stratification and increased the lake bottom dissolved oxygen levels ($p < 0.05$). During the two-year study, some water quality parameters showed similar distributions in the two lakes, such as monthly mean values of T and DO in lake surface, middle and bottom, as well as monthly mean total alkalinity, TIC, TOC, RP and TN concentrations.

Significant differences were also observed between Fossil Creek Lake and Sheldon Lake in several water quality parameters. First of all, the pH values in Fossil Creek Lake are higher than those in Sheldon Lake, especially in 2011. The reasons are attributable to submerged macrophytes photosynthesis and loss of inorganic carbon by the aeration system. Influenced by the local geochemical background, the feeding water of Fossil Creek Lake, which passes through a sulfate rich groundwater, causes extremely high sulfate, hardness and specific conductance levels in Fossil Creek Lake, which are around 10 times higher than those in Sheldon Lake. Because of the loss of aquatic vegetation through bottom-sediment removal, Sheldon Lake exhibited a turbid, phytoplankton-dominated state, rather than the clear, macrophyte-dominated state of Fossil Creek Lake. As a result, TSS and chlorophyll a concentrations were significantly higher than those in Fossil Creek Lake. In addition, the Secchi disk transparency of Sheldon Lake rarely exceeded 0.5 m, which was lower than Fossil Creek Lake (>1m of most sampling dates) during the two-year study. Furthermore, total phosphorus concentrations in Fossil Creek Lake were lower than those in Sheldon Lake.

Both of the lakes are nitrogen limited eutrophic lakes. In both lakes aeration produce no positive effects on the limiting nutrient and eutrophic condition. Eutrophication generally promotes excessive simple algae and plankton growth in many freshwater lakes. Without proper management, eutrophication often has pronounced negative effects on the oxygen concentration and animal life in a lake when algal bloom becomes excessive. The reduction of in-lake P has been regarded as the key to recovery of eutrophic lakes (Welch, 2009).

To reduce total phosphorus concentrations in both lakes to $50 \mu\text{g L}^{-1}$, the Vollenweider model and modified One-Box model were used to estimate the required reduction in external loading if it is chosen as the only restoration measure. In Fossil Creek Lake, the external loading from incoming flow and stormwater runoff must be decreased from 4.9 to $1.21 \text{ g m}^{-2} \text{ yr}^{-1}$ indicated by Vollenweider model and to $0.88 \text{ g m}^{-2} \text{ yr}^{-1}$ estimated by One-Box model. In Sheldon Lake, with 4 times more stormwater runoff than Fossil Creek Lake, the external loading must be reduced from 9.9 to 0.69 estimated by Vollenweider model, and to $0.4 \text{ g m}^{-2} \text{ yr}^{-1}$ estimated by the One-Box model. If increased P retentions are chosen as the only restoration measure, the net sedimentation factor would have to increase from 1.9 to 57.6 yr^{-1} , around 30 times to reach the restoration target in Fossil Creek Lake, and change from -0.5 to 47.8 yr^{-1} in Sheldon Lake.

Even though Fossil Creek Lake is still eutrophic as the pre-aeration period, it has remained in a clear water state. But Sheldon Lake, which is also a eutrophic lake after restoration, is in turbid and phytoplankton-dominated state, which may because of three main reasons: 1) lake eutrophic conditions; 2) loss of submerged macrophytes; and 3) bottom sediment suspension by wind, aerators and benthivorous fish. To restore Sheldon Lake to a stable clear-water state, besides phosphorus management, restoration of aquatic macrophytes, shut off some aerators to prevent sediment resuspension from the bottom and biomanipulation which alters the food web to favor grazing on algae by zooplankton, or that eliminate fish species that recycle bottom sediment and nutrients of the lake would be three effective measures.

CHAPTER 4

WATER QUALITY FACTORS IN RELATION TO LAKE CLARITY IN TWO SMALL SHALLOW EUTROPHIC LAKES

4.1 Introduction

Many researchers have found that submerged macrophytes form an important structural habitat in shallow lakes and play a key role in determining lake water clarity. Submerged macrophytes provide cover for fish and substrate for aquatic invertebrates, produce oxygen, enhance water clarity by reducing the resuspension of bottom material, and suppress algal growth by competing for available nutrients. A quantitative study of the effect of submerged plants on water quality and biota under fish free conditions for three weeks by Nakamura et al. (2008) showed that the presence of dense macrophytes was the main factor in creation of a clear water state. Modified local soils mixed with macrophyte seeds were sprayed in the 0.1km² bay of Lake Tai in order to flocculate the algal blooms and sink them down to the bottom of the lake, which turned out to be a quick restoration method to increase water clarity and improve sediment quality (Pan et al., 2011). The water clearing effect of aquatic vegetation may be caused by different physical, chemical, and biological mechanisms. Ginn (2011) identified four significant limnological variables related to plant biomass: (a) lake depth--light; (b) substrate type, nutrient availability, stability and wave pressure; (c) phosphorus loading from closest tributary; and (d) size of the area drained by a tributary. The relative abundance and spatial distribution of charophytes was correlated to water clarity, total phosphorus level and chlorophyll a content (Hutorowicz and Dziedzic, 2008). Sagrario et al. (2005) pointed out that nitrogen may play a far more important role than other factors in the loss of submerged macrophytes at increased nutrient loading and for the delay in the re-establishment of the nutrient loading reduction. However, macrophyte cover

was significantly negatively correlated with turbidity and plankti-benthivorous fish abundance in most researched ponds located around Brussels (Belgium) (Backer et al., 2010). Moreover, the effect of vegetation on water clarity in 83 sampled lakes in South America appears to be smaller than that found in various Northern hemisphere studies (Kosten et al., 2009). For these reasons, the ecological roles of other water quality parameters in relation to lake clarity should be taken into consideration, and not merely that of macrophytes (Cazzanelli, 2008). Greffard et al. (2012) studied 26 shallow lakes of northeastern USA, and showed that chironomids are significant indicators of turbidity, followed by nitrogen in shallow lakes. Hofmann and Dominik (1995) reported that the content of organic matter appeared to greatly influence the factor of light attenuation, as a result influencing the water clarity.

The relation of lake clarity with water quality parameters other than nitrogen, phosphorus and chlorophyll is far less investigated. In an attempt to find out the relationships between water quality parameters and water clarity as measured by Secchi depth, we have sampled from 2010 to 2011 in Fossil Creek Lake, which has been restored by submerged aeration, and Sheldon Lake, with dredging and a submerged aeration system. By using linear and multi-linear regression analysis, we defined relationships between lake clarity and environmental variables in the two urban, shallow, eutrophic lakes in Fort Collins, Colorado based on 10 monitored water quality parameters, which are: inorganic carbon (IC), total organic carbon (TOC), total nitrogen (TN), orthophosphate (RP), total phosphorus (TP), hardness, alkalinity, total suspended solids (TSS), Secchi depth (SD), and chlorophyll a (Chl-a).

The aim of this study is to: (1) identify the relationships between water quality parameters and water clarity in each lake; and (2) develop water clarity-water quality parameter models using

data collected from different sampling locations in both lakes from 2010 to 2011; (3) compare the water quality parameters which have significant relationships with water clarity in both lakes.

4.2 Materials and methods

4.2.1 Study sites

Two similar size, previously eutrophic, nitrogen limiting lakes, Fossil Creek Lake (Figure 3-1) and Sheldon Lake (Figure 3-2) in Fort Collins, Colorado, were chosen as the research objects. Fossil Creek Lake, a clear, macrophyte-dominated lake, was treated by submerged aeration. Dredging and submerged aeration were applied in Sheldon Lake which exhibited a turbid, phytoplankton-dominated state, to improve the water quality and prevent the growth of cyanobacteria. Table 3-1 summarized the morphometric characteristics of both lakes.

4.2.2 Statistical analysis

Simple and multiple linear regression was used to evaluate relationships between water quality parameters and water clarity. All the water quality data were \log_{10} -transformed in order to achieve normal distribution of the variables prior to regression and correlation. The specific hypothesis tested is that there is a correlation between \log water quality parameter concentrations and \log Secchi depth readings. The null hypothesis is that no correlation exists. The test statistic (α) is the probability of observing a correlation when none exists. All statistical tests were performed using Matlab 2009, and statistical significance was considered to be $p < 0.05$.

4.3 Results and discussion

The average inorganic carbon, alkalinity, total organic carbon and total nitrogen values in Fossil Creek Lake are higher than those in Sheldon Lake (Tables 4-1). In addition, the mean hardness concentration in Fossil Creek Lake was generally high, averaging at 969 mg L^{-1} (Table 4-1),

especially when compared with an average hardness of 154 mg L⁻¹ (Table 4-1) of Sheldon Lake. However, the orthophosphate and total phosphorus averaged at 0.18 and 0.34 mg L⁻¹ respectively (Table 4-1), which is less than the averages of 0.32 and 0.60 (Table 4-1) monitored in Sheldon Lake. Furthermore, the average chlorophyll-a and total suspended solids concentrations in Fossil Creek Lake were only half those of Sheldon Lake. Secchi depth readings averaged 1.28m in Fossil Creek Lake and exceeded the average of 0.35m recorded in Sheldon Lake.

Table 4-1 Mean, standard error, and minimum and maximum values for 10 physical and chemical parameters for Fossil Creek Lake and Sheldon Lake

Parameter	Lake	No. samples	Mean	Standard error	Minimum	Maximum
IC (mg L ⁻¹)	FCL	81	31.11	1.49	10.20	67.96
	Sheldon	73	24.22	0.60	12.27	45.04
TOC (mg L ⁻¹)	FCL	81	11.94	0.46	2.80	20.64
	Sheldon	73	9.84	0.36	1.88	17.30
TN (mg L ⁻¹)	FCL	81	1.18	0.07	0.26	3.23
	Sheldon	73	1.12	0.05	0.12	2.71
RP (mg L ⁻¹)	FCL	81	0.18	0.01	0.02	0.53
	Sheldon	73	0.32	0.02	0.02	0.78
TP (mg L ⁻¹)	FCL	81	0.34	0.02	0.04	0.76
	Sheldon	73	0.60	0.03	0.09	1.60
Hardness (mg L ⁻¹)	FCL	81	969.49	24.43	530.00	1550.00
	Sheldon	73	154.07	3.41	64.00	250.00
Alkalinity (mg L ⁻¹)	FCL	81	151.58	7.38	66.00	326.00
	Sheldon	73	110.58	1.78	44.00	152.00
TSS (mg L ⁻¹)	FCL	81	7.17	1.01	0.47	45.00
	Sheldon	73	18.16	1.18	1.00	54.00
Secchi (m)	FCL	81	1.28	0.05	0.55	2.90
	Sheldon	73	0.35	0.01	0.21	0.76
Chl-a (µg L ⁻¹)	FCL	47	26.55	1.15	18.00	65.00
	Sheldon	50	58.74	3.02	24.00	115.00

4.3.1 Water quality parameters related with Secchi depth

Of the 10 monitored water quality parameters, there are 8 water quality parameters which have significant relationships with Secchi disk in Fossil Creek Lake (Table 4-2). The hypothesized

positive relationship between Secchi disk and TP was not significant ($p=0.25$), thus the null hypothesis that no correlation exists is not rejected. There was significant positive relationship ($p<0.05$) between log Secchi disk (SD) and log orthophosphate (RP) in Fossil Creek Lake (Table 4-2). However, orthophosphate alone accounted for approximately 8% of the variability in log Secchi disk. Negative correlations were observed between log SD and the other 7 water quality parameters, which are IC, TOC, TN, hardness, alkalinity, TSS and chlorophyll a. Negative correlations indicate that Secchi depth decreases with higher concentrations of these water quality parameters. Inorganic carbon alone accounted for 30% of the variance in Secchi depth, and approximately 22% of the variability in Secchi depth is explained by the linear regression on total organic carbon concentrations (Table 4-2), whereas total nitrogen alone accounted for 39% of the variance in Secchi depth. Although nitrogen has been shown as the limiting nutrient in Fossil Creek Lake, the data presented here indicate that both the carbon and nitrogen are the limiting nutrients. Multiple regression using RP, IC, TOC, and TN explained the variability in Secchi disk in Fossil Creek Lake, 47% ($p<0.05$). Hardness alone explained 40% of the variance in Secchi depth, and alkalinity explained 35% of the variance in water clarity. When multiple regression was used with both hardness and alkalinity, the correlation was significant ($p<0.05$) with the relationship explaining 46% of the variability in log Secchi depth. Similar to inorganic carbon, total suspended solids accounted for 31% of the variation in Secchi depth. The correlation between chlorophyll a concentrations and Secchi depth was similar to the total organic carbon relationship, describing 22% of the variation in Secchi depth. Multiple regression analysis using all the related water quality parameters explained 60% of the variability in Secchi depth in Fossil Creek Lake.

Table 4-2 Water quality parameters and Secchi depth correlation coefficient square (R^2) and p value at Fossil Creek Lake and Sheldon Lake

Model	R^2	p
Fossil Creek Lake		
$\log_{10} \text{SD} = -0.44 \log_{10} \text{IC} + 0.71$	0.30	1.47E-07
$\log_{10} \text{SD} = -0.41 \log_{10} \text{TOC} + 0.51$	0.22	8.48E-06
$\log_{10} \text{SD} = -0.45 \log_{10} \text{TN} + 0.09$	0.39	5.75E-10
$\log_{10} \text{SD} = 0.15 \log_{10} \text{RP} + 0.2$	0.08	9.00E-03
$\log_{10} \text{SD} = -1.00 \log_{10} \text{Hardness} + 3.03$	0.40	2.35E-10
$\log_{10} \text{SD} = -0.50 \log_{10} \text{Alkalinity} + 1.15$	0.35	4.46E-09
$\log_{10} \text{SD} = -0.22 \log_{10} \text{TSS} + 0.22$	0.31	8.15E-08
$\log_{10} \text{SD} = -0.20 \log_{10} \text{Chl-a} - 0.25$	0.22	8.58E-04
Sheldon Lake		
$\log_{10} \text{SD} = -0.15 \log_{10} \text{TSS} - 0.29$	0.16	4.80E-04
$\log_{10} \text{SD} = -0.18 \log_{10} \text{Chl-a} - 0.71$	0.13	5.00E-04

Unlike Fossil Creek Lake, in Sheldon Lake, the water quality parameters collected between 2010 and 2011 showed that none of the hypothesized positive nutrient and Secchi depth relationships were significant ($p > 0.05$), leading to failure to reject the null hypothesis that no correlation exists. When multiple regression was used with all the nutrient parameters (IC, TOC, TN, RP and TP), the correlation was significant ($p < 0.05$) with the relationship explaining only 19% of the variability in log Secchi depth. Only two water quality parameters have significant relationships ($p < 0.05$) with Secchi depth in Sheldon Lake, which are total suspended solids and chlorophyll a (Table 4-2). Negative correlations were observed between both of them. Approximately 16% of the variability in Secchi depth is explained by total suspended solids concentrations, whereas the chlorophyll a concentrations alone accounted for 13% of the Secchi depth variation. In the model used, total suspended solids and chlorophyll a concentrations together explained 25% of the variability in Secchi depth ($p < 0.05$).

Total suspended solids and chlorophyll a concentrations have significant relationships with Secchi disk ($p < 0.05$) in both lakes (Table 4-2), but the regression equations appeared different for each lake. To examine this difference further, the total suspended solids-Secchi depth, and chlorophyll a-Secchi depth data collected were plotted for both lakes (Figures 4-1, 4-2). In Figure 4-1, all but five sites in the Sheldon Lake data fall well below Fossil Creek Lake data, which indicates that most of the total suspended solids in Fossil Creek Lake are less than those in Sheldon Lake, and the Secchi depth per unit of total suspended solids is much higher in Fossil Creek Lake than in Sheldon Lake.

A similar trend was observed in Figure 4-2, where most of the Sheldon Lake data fall well below the Fossil Creek Lake data. It appears that the Secchi depth per unit of chlorophyll a is much less in Sheldon Lake than in Fossil Creek Lake. In other words, at same chlorophyll a concentration, Fossil Creek Lake tends to have a higher Secchi depth than the Sheldon Lake.

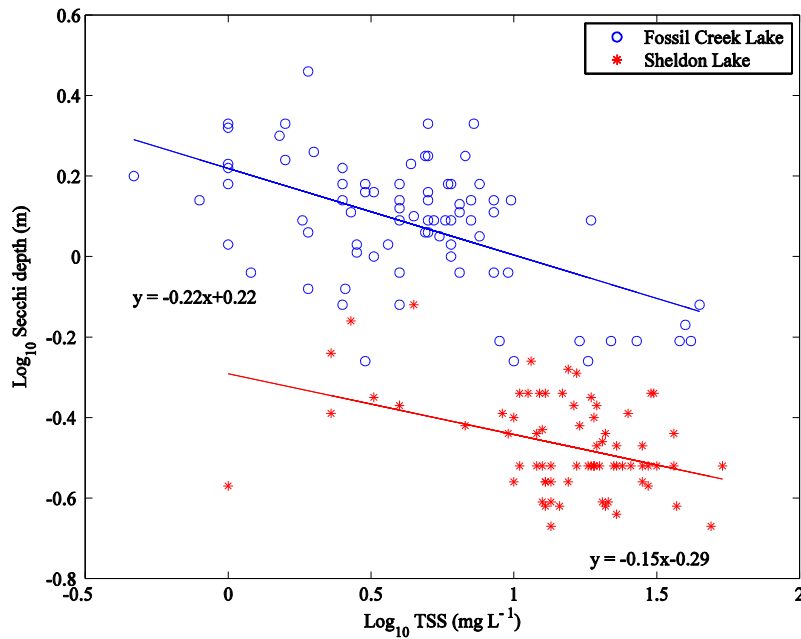


Figure 4-1 Relations between TSS and Secchi depth for Sheldon Lake (asterisks) and Fossil Creek Lake (open circles)

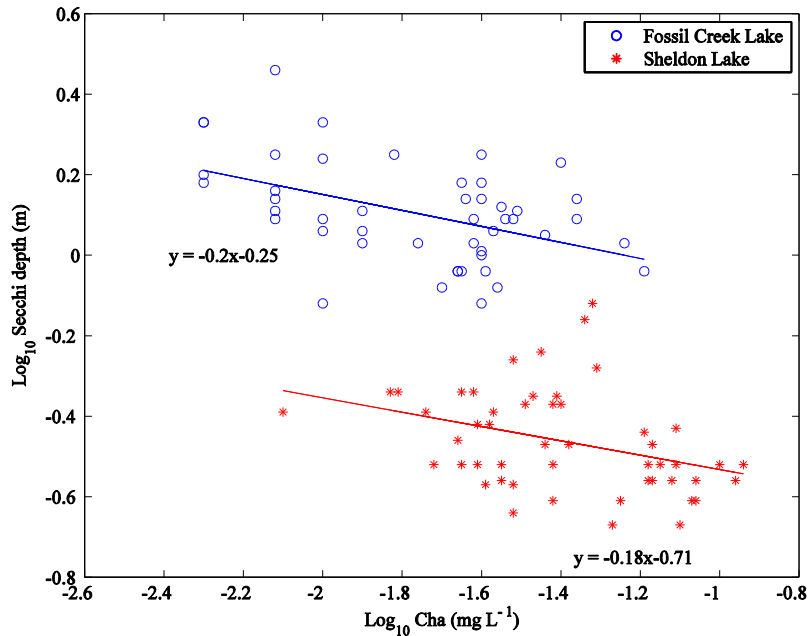


Figure 4-2 Relations between chlorophyll a and Secchi depth for Sheldon Lake (asterisks) and Fossil Creek Lake (open circles)

4.3.2 Water quality parameters related with TSS

To better understand the variability of total suspended solids in the two lakes, simple and multiple linear regression was used to evaluate relationships between monitored water quality parameters and total suspended solids. In Fossil Creek Lake, there are 6 water quality parameters which have significant positive relationships ($p < 0.05$) with total suspended solids (Table 4-3). These are IC, TOC, TN, hardness, alkalinity and chlorophyll a. Total organic carbon accounted for most of the variance in total suspended solids, which is approximately 42%. Lind and Davalos-Lind (1991) maintain that suspended solids provide surface for adsorption of dissolved organic materials, which indicates that total suspended solids are positively related to total organic carbon levels. Total nitrogen, explained 22% of the variation in total suspended solids. Alkalinity accounted for 19% of the variability, while hardness and chlorophyll a each accounted for 16% of the variability in total suspended solids. In contrast, around 11% of the variability in total suspended solids was explained by the linear regression on inorganic carbon (Table 4-3).

Multiple regression was used with IC, TOC and TN, and the correlation was significant ($p < 0.05$) with the relationship explaining 48% of the variability in total suspended solids.

Table 4-3 Water quality parameters and total suspended solids correlation coefficient square (R^2) and p value at Fossil Creek Lake and Sheldon Lake

Model	R^2	p
Fossil Creek Lake		
$\log_{10} \text{TSS} = 0.68 \log_{10} \text{IC} - 0.34$	0.11	2.00E-03
$\log_{10} \text{TSS} = 1.45 \log_{10} \text{TOC} - 0.87$	0.42	5.60E-11
$\log_{10} \text{TSS} = 0.86 \log_{10} \text{TN} + 0.64$	0.22	1.24E-05
$\log_{10} \text{TSS} = 1.60 \log_{10} \text{Hardness} - 4.12$	0.16	2.10E-04
$\log_{10} \text{TSS} = 0.95 \log_{10} \text{Alkalinity} - 1.37$	0.19	4.40E-05
$\log_{10} \text{TSS} = 0.40 \log_{10} \text{Chl-a} + 1.26$	0.16	8.15E-08
Sheldon Lake		
$\log_{10} \text{TSS} = 0.26 \log_{10} \text{RP} + 1.33$	0.09	8.00E-03
$\log_{10} \text{TSS} = 0.39 \log_{10} \text{TP} + 1.28$	0.06	3.00E-02
$\log_{10} \text{TSS} = 1.78 \log_{10} \text{Alkalinity} - 2.45$	0.15	7.00E-04

In Sheldon Lake, orthophosphate, total phosphorus and alkalinity have significant positive relationships ($p < 0.05$) with total suspended solids. However, orthophosphate only accounted for 9% of the variability in total suspended solids while total phosphorus accounted for 6%. When multiple regression was used with RP and TP, the correlation was significant ($p < 0.05$) with the relationship explaining 11% of the variability in total suspended solids. Alkalinity alone accounted for 15% of the variability in total suspended solids (Table 4-3).

In both lakes, alkalinity has a significant relationships with total suspended solids ($p < 0.05$) (Table 4-3), but the regression equations appeared different for each lake. To compare the total suspended solids-alkalinity relationships in the two lakes and examine this difference further, the total suspended solids and alkalinity data collected were plotted for both lakes (Figures 4-3). In Figure 4-3, all but three sites of Fossil Creek Lake data fall well below Sheldon Lake data, which indicates that the total suspended solids per unit of alkalinity is much less in Fossil Creek Lake

than in Sheldon Lake. Total suspended solids were negatively related with Secchi depth in both lakes (Table 4-3).

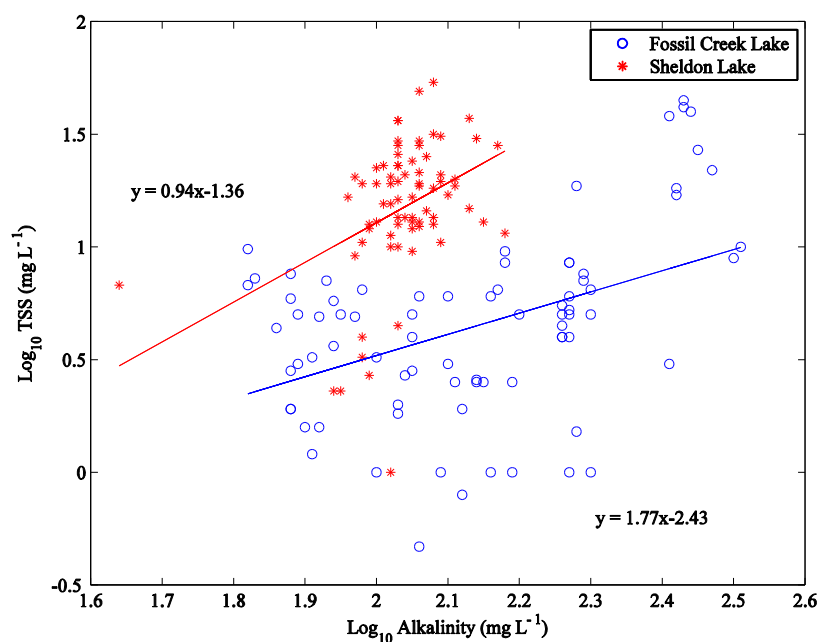


Figure 4-3 Relations between total suspended solids and alkalinity for Sheldon Lake (asterisks) and Fossil Creek Lake (open circles)

4.3.3 Water quality parameters related to chlorophyll a

Following similar logic, we also compared the models relating chlorophyll a to water quality parameters for the two lakes. For Fossil Creek Lake, IC, TOC, TSS, hardness, and alkalinity have a significantly correlation with chlorophyll a concentrations. All the parameters but alkalinity, are positive correlated with chlorophyll a concentrations. Hardness explained 39% of the chlorophyll a variation, followed by alkalinity, which was 22%. Total suspended solids, inorganic carbon and total organic carbon explained approximately 16%, 13% and 12% of the variability in chlorophyll a concentrations respectively. High concentrations of total suspended solids can cause light limitation and thus suppress chlorophyll concentrations (Hoyer et al., 2002). Multiple regression using IC and TOC explained 16% of the variation in chlorophyll a concentrations, whereas both hardness and alkalinity together accounted for 45% of the variation

in chlorophyll a. All of the five parameters together explained 64% of chlorophyll a variation. For Sheldon Lake, chlorophyll a concentrations were significantly positive correlated with nutrients, which are IC, TOC and TN ($p < 0.05$). TOC explained 22% of the variability in chlorophyll a, followed by IC (20%) and TN (16%). All of the three parameters together accounted for approximately 44% of the variability in chlorophyll a concentrations. Although nitrogen has been shown as the limiting nutrient in both lakes, the data presented in Table 4-4 suggest that carbon is the limiting nutrient in Fossil Creek, and carbon and nitrogen are both limiting in Sheldon Lake.

Table 4-4 Water quality parameters and chlorophyll a correlation coefficient square (R^2) and p value at Fossil Creek Lake and Sheldon Lake

Model	R^2	p
Fossil Creek Lake		
$\log_{10} \text{ Chl-a} = 0.56 \log_{10} \text{ IC} - 2.52$	0.13	1.40E-02
$\log_{10} \text{ Chl-a} = 0.54 \log_{10} \text{ TOC} - 2.31$	0.12	1.86E-02
$\log_{10} \text{ Chl-a} = 0.4 \log_{10} \text{ TSS} - 1.99$	0.16	4.80E-03
$\log_{10} \text{ Chl-a} = 1.71 \log_{10} \text{ Hardness} - 6.79$	0.39	2.59E-06
$\log_{10} \text{ Chl-a} = 0.91 \log_{10} \text{ Alkalinity} - 3.63$	0.22	8.61E-04
Sheldon Lake		
$\log_{10} \text{ Chl-a} = -1.53 \log_{10} \text{ IC} + 0.71$	0.20	1.12E-03
$\log_{10} \text{ Chl-a} = 0.70 \log_{10} \text{ TOC} - 2.06$	0.22	5.47E-04
$\log_{10} \text{ Chl-a} = 0.47 \log_{10} \text{ TN} - 1.40$	0.16	3.50E-03

Inorganic carbon and total organic carbon concentrations have significant relationships with chlorophyll a ($p < 0.05$) in both lakes (Table 4-4), but the regression equations appeared different for each lake, especially in equations of inorganic carbon. Inorganic carbon, total organic carbon, and chlorophyll a data collected were plotted for both lakes to compare the relationships (Figures 4-4, 4-5). In Figure 4-4, most of Fossil Creek Lake data fall well below Sheldon Lake data, which indicates that chlorophyll a levels in Fossil Creek Lake are less than they are in Sheldon Lake, and the chlorophyll a per unit of total organic carbon is higher in Sheldon Lake than the

one in Fossil Creek Lake. Inorganic carbon concentrations were negatively correlated with chlorophyll a levels in Sheldon Lake, which indicates that chlorophyll a concentrations increased with lower concentrations of inorganic carbon. However, chlorophyll a concentrations in Fossil Creek Lake are positively related with the inorganic carbon levels (Figure 4-5). When log IC is lower than 1.5 ($IC < 31.6 \text{ mg L}^{-1}$), most of chlorophyll a concentrations in Sheldon Lake are higher than those in Fossil Creek Lake. When Log IC is larger than 1.5 ($IC > 31.6 \text{ mg L}^{-1}$), chlorophyll a concentrations in Sheldon Lake are lower than those in Fossil Creek Lake.

There may be some other factors which might affect the chlorophyll a concentrations, such as flushing rate and zooplankton and shellfish population (Hoyer et al., 2002). High flushing rate of freshwater lakes washout or remove phytoplankton, resulting in low chlorophyll a concentrations in freshwater systems. Zooplankton and shellfish have been shown to decrease chlorophyll concentrations in lakes and estuaries (Quiros 1990; Meeuwig et al., 1998; and Hoyer et al., 2002)

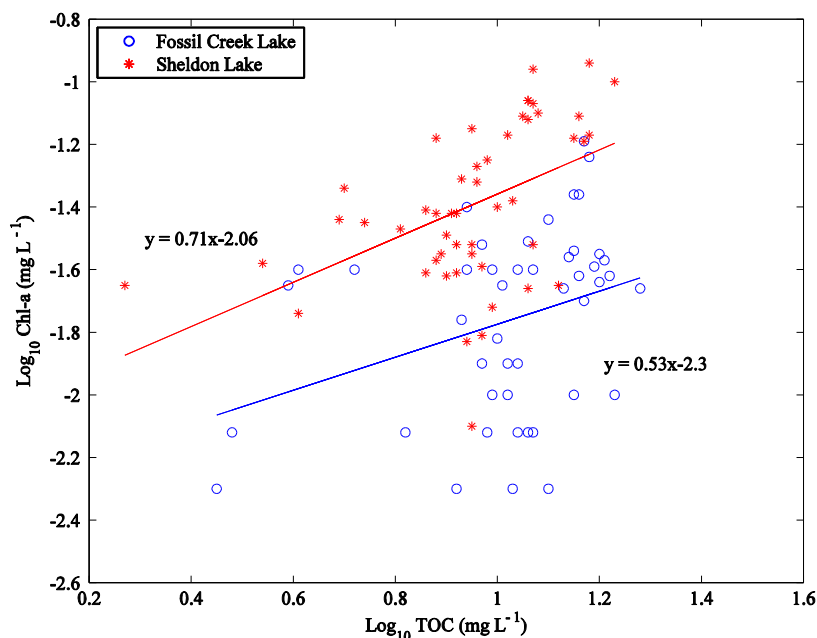


Figure 4-4 Relations between chlorophyll a and total organic carbon for Sheldon Lake (asterisks) and Fossil Creek Lake (open circles)

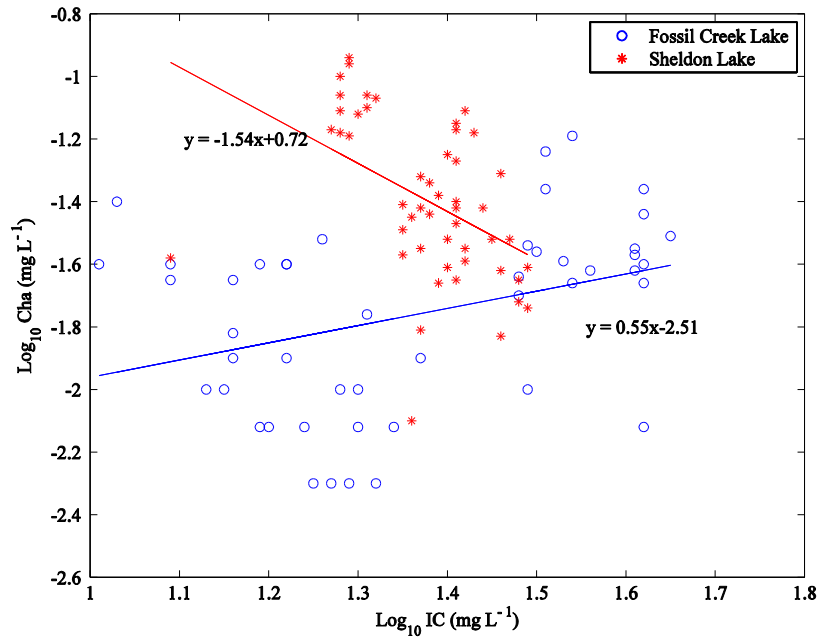


Figure 4-5 Relations between chlorophyll a and inorganic carbon for Sheldon Lake (asterisks) and Fossil Creek Lake (open circles)

4.4 Conclusions

In this study, we developed models relating water clarity and monitored water quality parameters in Fossil Creek Lake and Sheldon Lake respectively. The observations of other researchers usually pointed out that phosphorus, nitrogen and chlorophyll a are highly related to lake clarity. However, in Fossil Creek Lake, hardness accounted for the most variance in water clarity (40%), and alkalinity explained 35% of the variability in water clarity. Total nitrogen accounted for 39%, and the multiple regression analysis of nutrient levels (IC, TOC, TN and RP) together accounted for 46% of the variability in Secchi disk. Total suspended solids and chlorophyll a accounted for 31% and 22% of the variation in water clarity respectively in Fossil Creek Lake. All of these parameters were negatively correlated to lake clarity, which means the lower these values, the higher water clarity. The results in Fossil Creek Lake indicated that if the eutrophication condition were left without proper control, it will eventually negatively affect the lake clarity in the long term.

In Sheldon Lake, total suspended solids and chlorophyll a are the two parameters have significant negative correlation with Secchi depth. Approximately 16% and 13% of the variability in Secchi depth was explained by the linear regression on total suspended solids and chlorophyll a respectively, and 25% of the variability in Secchi depth was explained by using the two parameters together. Based on the results, decreasing total suspended solids levels and chlorophyll a concentrations should be the direct approaches to restore Sheldon Lake to a stable clear water state.

Because total suspended solids and chlorophyll a have significant relationships with Secchi disk ($p < 0.05$) in both lakes, comparison study was conducted on these two parameters in both lakes. Primary results show that the Secchi depth per unit of total suspended solids or chlorophyll a is much higher in Fossil Creek Lake than in Sheldon Lake. Furthermore, in both lakes, alkalinity has a significant relationship with total suspended solids, and the total suspended solids per unit of alkalinity is much less in Fossil Creek Lake than in Sheldon Lake. Chlorophyll a variation was significantly related to inorganic carbon and total organic carbon in both lakes. Chlorophyll a per unit of total organic carbon is higher in Sheldon Lake than in Fossil Creek Lake. However, when inorganic carbon is lower than 31.6 mg L^{-1} , most chlorophyll a concentrations in Sheldon Lake are higher than Fossil Creek Lake; when inorganic carbon is larger than 31.6 mg L^{-1} , chlorophyll a concentrations in Sheldon Lake are lower than Fossil Creek Lake.

Possible impact factors of water clarity, such as the phytoplankton species, zooplankton biomass, crayfish and fish community in the lake water, quantitative macrophytes, and lake flushing rate, etc were not investigated in this study due to the limitation of funding and still have to be determined in future research.

CHAPTER 5

CONCLUSIONS

5.1 Summary of hypotheses

1. *Placing aerators in the deepest locations of the lake will result in complete mixing due to water movement “downhill”, and will eliminate thermal stratification and improve near bottom water dissolved oxygen concentrations.*

Results from both Fossil Creek Lake (chapter 2) confirm this hypothesis. Results show that Fossil Creek Lake is destratified and well mixed through the entire water column after restoration by submerged aeration. The bottom water dissolved oxygen levels increased from less than 1 mg L^{-1} during the pre-aeration period to more than 4 mg L^{-1} , a level that suitable for most aquatic life. Furthermore, the differences of dissolved oxygen were not significant with the interaction of vertical depth and sampling sites after restoration.

2. *Submerged aeration will gradually decrease nutrient concentration in Fossil Creek Lake and may change the lake trophic status.*

The bottom levels of total phosphorus, orthophosphate and total nitrogen before restoration were usually 5 to 10 times higher than the surface levels. With the introduction of aeration, the vertical differences have been significantly reduced by the submerged aeration system (chapter 2). The whole lake concentrations of the three parameters after restoration are close to the surface values during pre-aeration period. However, the trophic state has remained the same as the pre-aeration period in Fossil Creek Lake.

3. *Chlorophyll a concentrations in Fossil Creek Lake will be suppressed by submerged complete aeration.*

Chlorophyll a concentrations during the pre-aeration year (2009) ranged from $8.44\text{--}55.14\text{ }\mu\text{g L}^{-1}$

and averaged $30.4 \mu\text{g L}^{-1}$. Chlorophyll a concentrations declined significantly ($p < 0.05$) during aeration, ranging from $9.12\text{--}42.93 \mu\text{g L}^{-1}$, and mean concentrations were $24.89 \mu\text{g L}^{-1}$ in 2010, and ranging from $11.13\text{--}48.43 \mu\text{g L}^{-1}$, and averaged at $24.48 \mu\text{g L}^{-1}$.

4. *Submerged aeration in Fossil Creek Lake and submerged aeration after dredging in Sheldon Lake may have different effects on nutrient levels, phytoplankton biomass and clarity.*

The concentrations of TN, TP and chlorophyll a were significantly reduced in both lakes by different restoration methods. The differences of Secchi depth in Fossil Creek Lake before and after aeration were insignificant ($p > 0.05$), while Secchi depth in Sheldon Lake decreased after aeration. Chlorophyll a and TSS concentrations were significantly higher in Sheldon Lake than in Fossil Creek Lake ($p < 0.05$). As a result, the Secchi disk transparency of Sheldon Lake, which rarely exceeded 0.5 m, was significantly lower than that of Fossil Creek Lake ($p < 0.05$) during the two-year study. The main reason for the decreased water transparency and clarity in Sheldon Lake after aeration is probably due to the loss of submerged macrophytes during dredging, which resulted in Sheldon Lake becoming a phytoplankton dominated lake. The resuspension of bottom sediment by aeration system in Sheldon Lake may be the other major reason.

The chlorophyll a and TSS concentrations, as well as TP in Sheldon Lake were significantly higher than those in Fossil Creek Lake. Both of the lakes are nitrogen limited eutrophic lakes. For aeration in both lakes, no positive effects on the limiting nutrient and eutrophic condition could be identified. Fossil Creek Lake is in macrophyte-dominated, clear water state before and after restoration, but Sheldon Lake condition was greenish brown and unsatisfactory even after restoration.

5. *The water clarity in both lakes is not only highly related to nitrogen, phosphorus and chlorophyll a, but also hardness and alkalinity.*

In Fossil Creek Lake, hardness accounted for the most variance (40%) in water clarity, followed by total nitrogen (39%) and alkalinity (35%), which are all negatively related with lake clarity. The multiple regression analysis of nutrient levels (IC, TOC, TN and RP) together accounted for 46% of the variability in Secchi depth. In Sheldon Lake, total suspended solids and chlorophyll a are the two parameters that have significant negative correlation with Secchi depth, which explained 25% of the variability. Hardness and alkalinity is not significantly related with the water clarity condition in Sheldon Lake.

5.2 Summary of findings

Submerged aeration is a technique intended to eliminate thermal stratification and improve bottom oxygen level. The primary strengths are simplicity and relatively low cost. The submerged aeration system in Fossil Creek Lake successfully eliminated most of its undesirable features, such as oxygen depletion, thermal stratification, and hydrogen sulfide odor and fish kills during lake overturns. In addition, it enhanced the macrophyte-dominated state and improved water clarity by suppressing the phytoplankton biomass. But aeration barely changed the limiting nutrient and eutrophic condition in either lake. Eutrophication generally promotes excessive plant growth and decay, favoring simple algae and plankton in many freshwater ecosystems. It often has two pronounced negative effects on the oxygen concentration and animal life in a lake when algal bloom becomes excessive. First of all, the excessive phytoplankton consumes a large amount of oxygen during dark phase respiration. The oxygen in the lake may drop to a level that causes stress or death in a population of fish, especially in hot summer months as the lake heats up and the reduces oxygen saturation levels. The second

negative effect of excessive bloom on a lake is a sudden death of phytoplankton population. If the phytoplankton biomass is suddenly killed, it will sink to the lake bottom and decompose rapidly, which could strip oxygen from the lake water to kill a fish population.

Natural or urban lakes with hydrogen sulfide concentrations at the level of 100 mg L^{-1} are rarely seen in North America. The mix of the entire water column would cause a remarkable decline in the dissolved oxygen level if there were not enough oxygen in the upper waters to be mixed or if the hypolimnetic water were rich in accumulated sulfide and organic matter. As a result, great care should be taken before initiating an aeration system in lakes with high hydrogen sulfide concentrations. It would be best to start up the system in the fall or spring when the lake is fully mixed and already aerated. Furthermore, one important factor during startup is to gradually increase the operating time of the system over a period of 4 to 8 weeks, and probably just 1/4 to 1/3 aerators at a time so that the lake has time to adjust to the new aeration system, the hydrogen sulfide odor is not too bad and the aquatic life has a place to take refuge.

Sediment dredging is one effective lake restoration technique to remove surface bottom layers rich in organic matters and nutrients and control their release or nutrient bioavailability. However, the process is expensive and aesthetically displeasing and it unavoidably affects the submerged aquatic vegetation and fish which in turn increases water turbidity. Because of the loss of aquatic vegetation through dredging, Sheldon Lake exhibited a turbid, phytoplankton-dominated state, rather than the clear, macrophyte-dominated state of Fossil Creek Lake. Due to the loss of submerged macrophytes, the eutrophic condition and bottom sediment suspended by wind, aerators and benthivorous fish, the lake water clarity as measured by Secchi disk transparency in

Sheldon Lake rarely exceeded 0.5m, which was significantly lower than Fossil Creek Lake with a Secchi disk transparency of more than 1m of most sampling dates.

5.3 Suggestions for future studies

In this study, it has been found that there are some areas requiring more research.

1. Phytoplankton is the major primary producer, which turns out an essential component of the trophic structure of freshwater ecosystems. Because of several factors as urban runoff, artificial mixing, and removal of bottom sediments, phytoplankton components of eutrophic, shallow lakes have frequently undergone dynamic changes in composition and abundance with important consequences for lake functioning and stability (Kalff, 2002). Seasonal succession of phytoplankton species in lakes reflects changes in physical and chemical characteristics of the water body. Monitoring seasonal changes in phytoplankton responses that is important to the long term water quality change can therefore contribute to effective management. Moreover, since different species have different preferences and tolerance ranges of various environmental factors, knowledge of the species composition of phytoplankton, and can be used in lake classifications and ecological status assessments (Willén et al. 1990).

2. The exchanges of nutrients, oxygen, and sulfide across the sediment-water interface are of significant ecological importance to shallow lakes. Nutrient regeneration or release in the sediment supplies a significant portion of phytoplankton and will further stimulating the cycle of algal growth, deposition of organic matter to sediments, and subsequent loss of oxygen. Nutrients and sulfide exchange rate between the sediment and water in Fossil Creek and Sheldon Lake need to be determined in future research in order to better understand the water and sediment interrelationship, to check the sediment oxidation/ reduction status, and to evaluate the

effectiveness of aeration at controlling nutrient release from sediments as well as the bottom sediment oxygen uptake rate.

3. Ecological responses, such as the phytoplankton species, zooplankton biomass, crayfish and fish community in the lake water, and quantitative macrophytes monitoring, that is important to the long term water quality changes, were not investigated in this study due to the funding limitation and still should be investigated in future research.

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APPENDIX A

Appendix A-1 Odor threshold

Concentration	Symptom
0.13 ppm	minimal perceptible odor
0.77 ppm	faint but perceptible odor
4.6 ppm	easily detectable moderate odor
27 ppm	strong unpleasant odor, but not intolerable

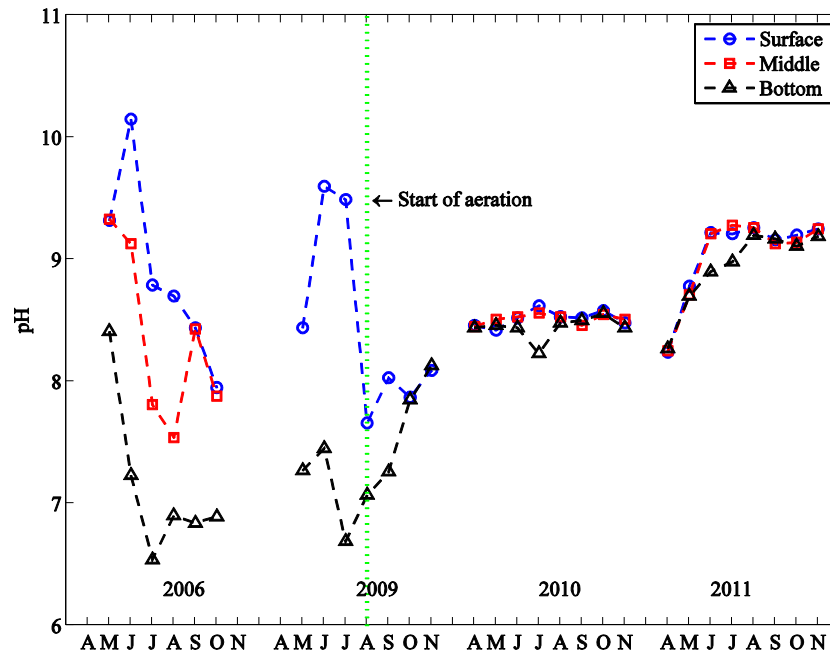
Source:<http://el.erdc.usace.army.mil/workshops/04jun-wots/kaluschue.pdf>

Appendix A-2 Typical exposure symptoms

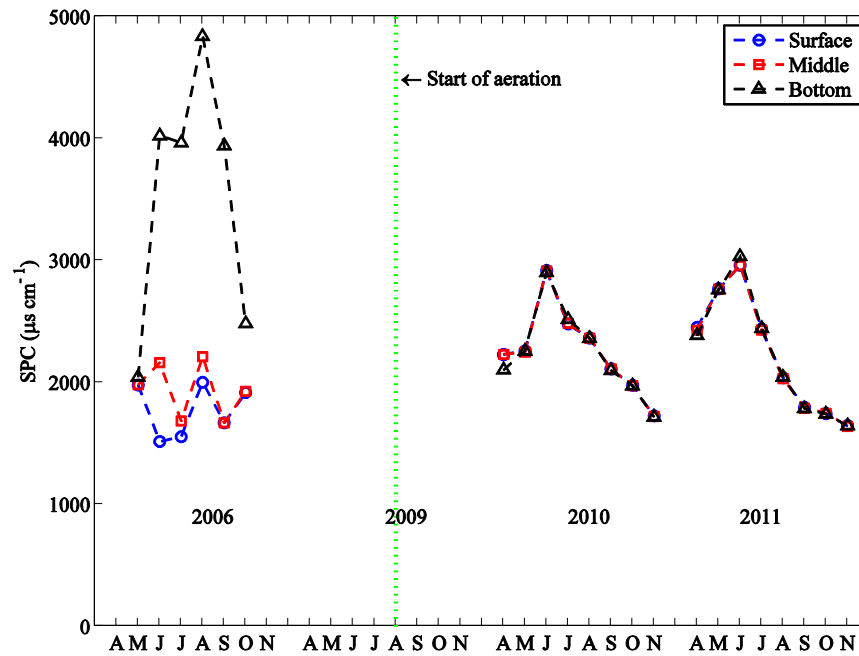
L O W	0 - 10 ppm	Irritation of the eyes, nose and throat
M O D	10 - 50 ppm	Headache Dizziness Nausea and vomiting Coughing and breathing difficulty
H I G H	50 - 200 ppm	Severe respiratory tract irritation Eye irritation / acute conjunctivitis Shock Convulsions Coma Death in severe cases

Source:<http://www.nfkci.info/Hydrogen Sulfide.htm>

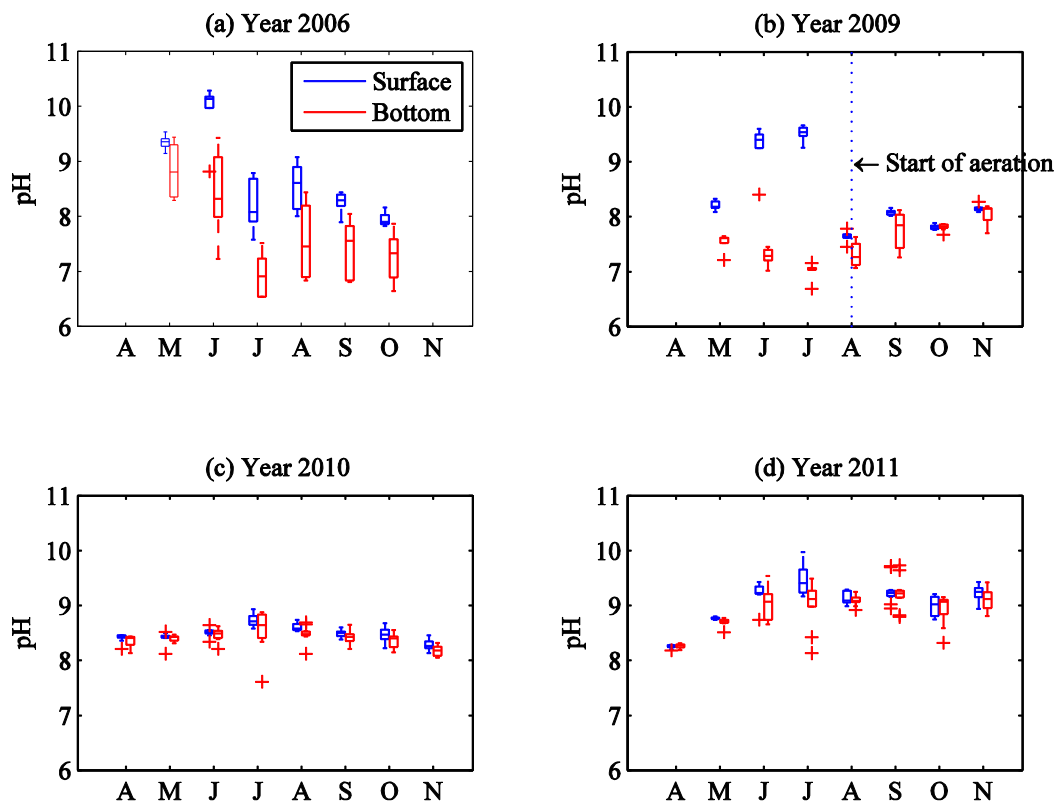
APPENDIX B



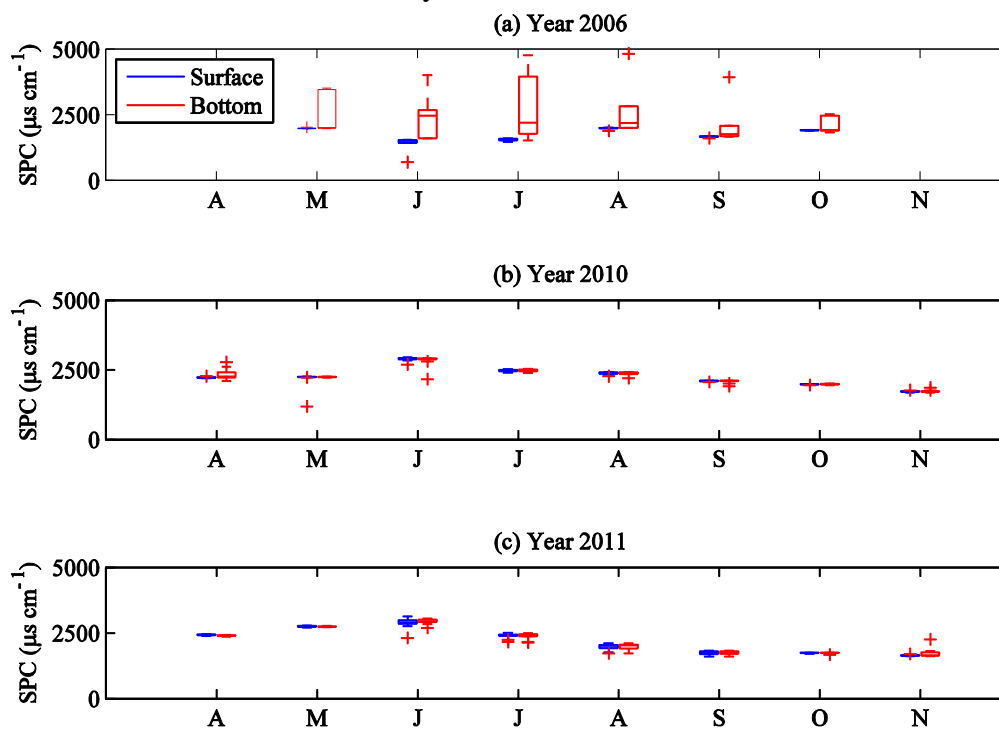
Appendix B-1 pH profile at surface, middle and bottom in sampling location 4. Start of aeration is indicated by a vertical line.



Appendix B-2 Specific conductance profile at surface, middle and bottom in sampling location 4. Start of aeration is indicated by a vertical line.



Appendix B-3 Surface and bottom pH from 11 sampling locations. Start of aeration is indicated by a vertical line.



Appendix B-4 Box plots of specific conductance spatial variation from 11 sampling locations at surface and bottom. Start of aeration is indicated by a vertical line.

APPENDIX C

Appendix C-1 Phosphorus loads from stormwater runoff in Sheldon Lake and Fossil Creek Lake in 2001 before restoration

Month	Intensity, in/month	Sheldon Lake			Fossil Creek		
		Q*, cfs	TP, mg/l	Load,kg/month	Q*, cfs	TP, mg/l	Load, Kg/month
January	0.59	0.08	0.32	1.99	0.04	0.32	1.01
February	0.26	0.04	0.33	0.84	0.02	0.36	0.47
March	0.98	0.14	0.35	3.61	0.07	0.35	1.84
April	2.54	0.36	0.33	8.93	0.19	0.33	4.54
May	3.49	0.48	0.93	34.06	0.25	0.93	17.31
June	0.41	0.06	1.82	7.86	0.03	0.82	1.80
July	0.89	0.12	1.10	10.31	0.06	0.41	1.95
August	1.07	0.30	4.36	98.22	0.15	0.86	9.85
September	0.89	0.13	0.24	2.25	0.06	0.74	3.52
October	0.28	0.04	0.35	1.03	0.02	0.35	0.52
November	0.86	0.12	1.66	15.06	0.06	0.66	3.04
December	0.07	0.01	0.35	0.26	0.00	0.35	0.13
Total, kg/year				184.42			45.99

*Q=CIA,

Where

C is the runoff coefficient, and is chosen at 0.53;

I is the rainfall intensity;

A is the contributing area

Appendix C-2 Phosphorus loads from stormwater runoff in Sheldon Lake and Fossil Creek Lake in 2010 after restoration

Month	Intensity, in/month	Sheldon Lake			Fossil Creek		
		Q, cfs	TP, mg/l	Load,kg/month	Q, cfs	TP, mg/l	Load, Kg/month
January	0.15	0.02	0.32	0.51	0.01	0.32	0.26
February	0.65	0.09	0.33	2.11	0.05	0.36	1.17
March	1.55	0.22	0.35	5.71	0.11	0.35	2.90
April	3.15	0.45	0.33	11.08	0.23	0.33	5.63
May	2.14	0.30	0.93	20.88	0.15	0.93	10.62
June	1.96	0.28	1.82	37.55	0.14	0.82	8.60
July	1.26	0.17	1.10	14.59	0.09	0.41	2.76
August	1.23	0.34	4.36	112.91	0.17	0.86	11.32
September	0.06	0.01	0.24	0.15	0.00	0.74	0.24
October	0.88	0.12	0.35	3.24	0.06	0.35	1.65
November	0.70	0.10	1.66	12.26	0.05	0.66	2.47
December	0.22	0.03	0.35	0.81	0.02	0.35	0.41
Total, kg/year				221.80			48.04

**Appendix C-3 Phosphorus loads from stormwater runoff in Sheldon Lake and Fossil Creek Lake
in 2011 after restoration**

Month	Intensity, in/month	Sheldon Lake			Fossil Creek		
		Q, cfs	TP, mg/l	Load,kg/mont h	Q, cfs	TP, mg/l	Load, Kg/month
January	0.29	0.04	0.32	0.98	0.02	0.32	0.50
Feburary	0.66	0.09	0.33	2.14	0.05	0.36	1.19
March	0.29	0.04	0.35	1.07	0.02	0.35	0.54
April	2.06	0.30	0.33	7.24	0.15	0.33	3.68
May	4.50	0.62	0.93	43.91	0.32	0.93	22.33
June	2.78	0.40	1.82	53.26	0.20	0.82	12.20
July	1.70	0.24	1.10	19.69	0.12	0.41	3.73
August	0.15	0.04	4.36	13.77	0.02	0.86	1.38
September	1.97	0.28	0.24	4.98	0.14	0.74	7.80
October	1.77	0.25	0.35	6.52	0.12	0.35	3.32
November	0.87	0.12	1.66	15.24	0.06	0.66	3.07
December	0.79	0.11	0.35	2.91	0.06	0.35	1.48
Total, kg/year				171.71			61.22

Appendix C- 4 Phosphorus input, output and in lake contents before and after restoration

Year	Lake	P _{in} , g/m ³	P _{out} , g/m ³	P _{lake} , g/m ³	Q, m ³ /yr	V, m ³	M _{in} , kg	M _{lake} , kg	M _{out} , kg
Pre-restoration	Sheldon	0.35	0.52	1.81	782598	133238	273.91	241.16	406.95
Pre-restoration	FCL	0.37	0.38	1.50	409559	68932	151.54	103.4	155.63
Year 2010	Sheldon	0.55	0.51	0.59	782598	133238	430.43	78.61	399.13
Year 2010	FCL	0.41	0.22	0.35	409559	68932	167.92	24.13	90.1
Year 2011	Sheldon	0.38	0.54	0.51	922190	133238	350.43	67.95	497.98
Year 2011	FCL	0.28	0.32	0.36	477819	68932	133.79	24.82	152.9