Review of St. Mary Lake Restoration Options

Prepared for:

Deborah Epps
Environmental Impact Assessment Biologist
Ministry of Environment
2080A Labieux Road
Nanaimo, BC.
V9T 6J9

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Report prepared by:

Ken I. Ashley, B.Sc., M.Sc., M.A.Sc., Ph.D.

Ken Ashley and Associates Ltd.,
1957 Westview Drive
North Vancouver, B.C.
Canada V7M 3B1

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Table of Contents

List of Figures

List of Tables

Executive Summary

Introduction
  Background
  General Characteristics
  Morphometry

Review of Lake Restoration Options
  Algicides
  Bacteria
  Destratification
  Dilutional flushing
  Diversion
  Dredging
  Food web manipulations
  Hypolimnetic aeration
  Hypolimnetic withdrawal
  P inactivation (alum, calcium and iron)
  Sediment oxidation
  Sound
  Watershed management and protection

Discussion

Literature Cited
List of Figures

Figure 1. St. Mary Lake on Salt Spring Island

Figure 2. Destratification system installed at Yellow Lake, BC

Figure 3. Mechanical surface aerators installed throughout Southern Interior British Columbia

Figure 4. Effect of increased flushing rate on critical P load for Coquitlam Reservoir

Figure 5. Reduction in total phosphorus in Lake Washington following nutrient diversion

Figure 6. Biomanipulation diagram indicating of top-down and bottom-up control various trophic levels

Figure 7. Full lift type of hypolimnetic aerator

Figure 8. Partial lift type of hypolimnetic aerator

Figure 9. Speece Cone type of hypolimnetic aerator

Figure 10. Hypolimnetic outlet siphon installed in Lake Ballinger, near Seattle, WA. Note the aerator on the siphon discharge

Figure 11. Schematic diagram of the Chain Lake, BC siphon installation

Figure 12. Schematic diagram of the “Riplox” system, showing the various mechanical components

Figure 13. Advertised sound coverage area for LG Sonic XXL Unit
List of Tables

Table 1. Morphometric Features of St. Mary Lake, Salt Spring Island

Table 2. Compendium of some common Algicide names

Table 3. Chemistry of copper sulfate

Table 4. Dredging options for Burnaby Lake, BC

Table 5. Total cost and cost per cubic meter for dredging options on Burnaby Lake

Table 6. Coverage area for algae

Table 7. Coverage area for blue-green algae

Table 8. Phosphorus budget for St. Mary Lake

Table 9. Summary of lake restoration options and techniques for St. Mary Lake
Executive Summary

St. Mary Lake has experienced cultural eutrophication due to human activities in the watershed, and water quality has deteriorated considerably since 1970. Taste and odour problems, low transparency, year-round cyanobacteria blooms and severe hypolimnetic oxygen depletion have negatively influenced the rainbow and cutthroat trout \textit{(Oncorhynchus mykiss and O. clarkii)} fishery and degraded the drinking water supply. A limnological assessment of St. Mary Lake attributed the decline in water quality to increased phosphorus loading from watershed development (e.g., septic tank discharges, road building and land clearing) to the point where the lake sediments had become the largest net source of phosphorus under anaerobic conditions.

A wide range of lake restoration techniques have been developed since the early 1970s, however, many of the techniques are not suitable on St. Mary Lake for a variety of reasons. The core activity of a water quality improvement plan must include a watershed management program to control point and non-point source nutrient loading. Without this approach, no lake restoration technique will be effective in the long term. Assuming an effective watershed management program is implemented, the appropriate water quality improvement techniques for St. Mary Lake are (1) hypolimnetic aeration, (2) food web biomanipulation and (3) hypolimnetic iron addition.

Hypolimnetic aeration is ideally suited to St. Mary Lake as internal loading is the single largest P source to St. Mary Lake, and the phosphorus in the lake is sensitive to changes in the oxidation-reduction potential. If the hypolimnion can be kept aerobic, the single largest P source can be significantly reduced, and water quality will improve accordingly. Hypolimnetic aeration conserves the cold water resource in the lake for potable water withdrawals and fisheries purposes. Food web biomanipulation should also be considered if large bodied zooplankton are being excessively grazed by planktivorous fish. Triploid cutthroat trout are an ideal top predator for use in BC lakes, and have demonstrated their effectiveness in controlling planktivorous fish in Wahleach Reservoir, near Hope. A hypolimnetic iron treatment, in combination with hypolimnetic aeration, should be considered if the sediment iron-phosphorus ratio is less than 15:1 due to pyrite formation in the profundal sediments.
Review of St. Mary Lake Restoration Options

Introduction

This report reviews the known lake restoration options and outlines their suitability for St. Mary Lake. St. Mary Lake is a key potable water reservoir for North Salt Spring Island, and has had a long history of water quality concerns since the 1970s (Nordin and McKeans (1983)). The Ministry of Environment, in cooperation with the North Salt Spring Waterworks District (NSSWD), installed and operated a hypolimnetic aeration system in St. Mary Lake for several years in the mid 1980s and early 1990s. The system was removed in 1994 due to structural problems. An updated model of a similar hypolimnetic aeration system is being considered for installation in 2007.

The purpose of this report is three-fold:

1. Review the available lake restoration techniques, and determine which options would be suitable for St. Mary Lake;

2. Analyze the techniques to determine if they could be implemented as stand-alone lake restoration options, or used in conjunction with the planned hypolimnetic aeration system to increase its effectiveness;

3. Provide a table that lists the advantages, disadvantages and possible options that could be used in conjunction with a hypolimnetic aeration system.

Background

St. Mary Lake is located in the small but distinctive Insular Lowland limnological region of BC (Northcote and Larkin, 1966). Lakes in this region are typically small (< 500 ha) with low mean depths. These lakes exhibit high surface temperatures, occasionally reaching 26 °C in midsummer, while bottom temperatures may reach 19 °C in shallower lakes that do not stratify. Total dissolved solids concentrations are usually above 100 mg L⁻¹, and over 200 mg L⁻¹ in some Gulf Island ponds. Due to the maritime climate, these lakes generally exhibit warm monomictic stratification (i.e., they are thermally stratified in the late spring, summer and early fall, and circulate freely during the rest of the year). However, brief periods of ice cover may form during the coldest winter months. Severe hypolimnetic oxygen depletion is typical for lakes in this region, and summer kills of fish have been reported in several lakes.

Phosphorus inputs from agricultural and residential development in this area of rapidly increasing population has accelerated cultural eutrophication, and water quality has deteriorated below Provincial and Federal standards for potable water and contact recreations in several lakes. Many rural areas, particularly the Gulf Islands, depend on
these lakes as their main supply of potable water. The combination of increasing population growth with decreasing water quality and availability has focused public and government attention on protecting and restoring key lakes in this small, but highly valued limnological region. Water quality, fisheries and contact recreation are considered of equal importance, with water quality becoming the dominant issue in water scare locations such as the Gulf Islands (Ashley and Nordin, 1999).

St. Mary Lake is a typical Insular Lowland lake that has experienced cultural eutrophication due to human activities in the watershed. St. Mary is a very important resource for Salt Spring Island as it supplies potable water for the north portion of the Island, and is the focus of considerable recreational activity. Several resorts are located around the lake perimeter, and recreational angling has been an important component of their operations.

The water quality of St. Mary Lake had deteriorated considerably since 1970. Taste and odour problems, low transparency, year-round cyanobacteria blooms and severe hypolimnetic oxygen depletion had eliminated most of the rainbow trout (Oncorhynchus mykiss) fishery and degraded the drinking water supply. Nordin and McKean (1983) conducted a thorough review of St. Mary Lake limnology, and attributed the decline in water quality to increased phosphorus loading from watershed development (septic tank discharges, road building and land clearing) to the point where the lake sediments had become a net source of phosphorus under anaerobic conditions.

Internal loading from the sediments was identified as the main source of phosphorus to the lake, with lesser amounts originating from septic tanks, groundwater, dust fall and natural watershed loading (Nordin and McKean, 1983). St. Mary Lake was selected as a study lake to evaluate the effectiveness of hypolimnetic aeration for improving water quality and increasing habitat for recreational fisheries (Ashley, 2000). As previously stated, a full lift hypolimnetic aerator was installed in St. Mary Lake in 1985-1986 by the Ministry of Environment, operated successfully until the early 1990s, and dismantled in 1994 due to structural corrosion of the inlet and outlet tubes.

**General Characteristics**

St. Mary Lake is now culturally eutrophic. It often experiences year-round algal blooms with resulting poor water quality and has a significant hypolimnetic oxygen deficit (Nordin and McKean, 1983). It is classified as a warm monomictic lake, as it thermally stratifies during the summer months, and circulates freely during the winter, and does not usually develop an ice cover. The fact that St. Mary Lake circulates all winter has likely prevented the lake from further water quality degradation, as oxygen is present throughout the water column and sediment-water interface for several months, thus allowing oxidation of a portion of the annual organic deposits.
The major natural vegetation cover of the St. Mary watershed is Douglas fir/western red cedar, with additional Douglas fir associated vegetation groups (Garry oak, Arbutus, Red alder and salal). Currently, considerable portions of the watershed have been developed and cleared for agricultural activities including forage crops and livestock, and lower density rural development is occurring within the watershed.

**Morphometry**

St. Mary is a medium sized lake with a surface area of 182 ha, and a maximum depth of 16.7 m (Figure 1). Additional morphometric features are as follows in Table 1.

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Figure 1. St. Mary Lake on Salt Spring Island.
Table 1. Morphometric Features of St. Mary Lake, Salt Spring Island.

<p>| | |</p>
<table>
<thead>
<tr>
<th></th>
<th></th>
</tr>
</thead>
<tbody>
<tr>
<td>Surface area</td>
<td>1.82 km$^2$</td>
</tr>
<tr>
<td>Drainage basin area</td>
<td></td>
</tr>
<tr>
<td>including lake</td>
<td>7.07 km$^2$</td>
</tr>
<tr>
<td>excluding lake</td>
<td>5.25 km$^2$</td>
</tr>
<tr>
<td>Maximum depth</td>
<td>16.7 m</td>
</tr>
<tr>
<td>Mean depth</td>
<td>8.8 m</td>
</tr>
<tr>
<td>Littoral area (&lt; 6 m)</td>
<td>30% of lake surface area (0.55 km$^2$)</td>
</tr>
<tr>
<td>Volume at 40.88 m asl</td>
<td>16,300,000 m$^3$</td>
</tr>
<tr>
<td>Volume at 40.0 m asl</td>
<td>14,709,500</td>
</tr>
<tr>
<td>Upper limit of thermocline</td>
<td>5 m</td>
</tr>
<tr>
<td>Lake level annual range (1979-81)</td>
<td>0.7 m (40.4 m – 41.4 m)</td>
</tr>
<tr>
<td>Mean calculated inflow (1979-81)</td>
<td>3,030,000 m$^3$</td>
</tr>
<tr>
<td>Mean calculated retention time (1979-81)</td>
<td>5.4 yr</td>
</tr>
<tr>
<td>Mean flushing rate (1979-81)</td>
<td>0.19 yr$^{-1}$</td>
</tr>
</tbody>
</table>

Review of Lake Restoration Options

Algicides

Algicides are chemical formulations that are applied to lakes in an attempt to control and/or kill algae. (Note: herbicides are specific chemical formulations designed to control and/or kill aquatic macrophytes). A variety of algicide formulations are available, under various trade names (Table 2). Copper is the active ingredient in the more common copper-based formulations (e.g., Cutrine Plus), and is an effective algicide that has been registered for use in potable water supplies (Cooke et al., 2005). The toxicity of copper formulations is from the cupric ion (Cu$^{+2}$), although some copper-hydroxy complexes may also be toxic (Cook et al., 2005) (Table 3). The cupric ion inhibits photosynthesis, algal phosphorus uptake, and nitrogen fixation (Cooke et al., 2005).

Table 2. Compendium of some common Algicide names

- Bethoxazin
- Copper sulfate
- Cybutryne
- Dichlone
- Dichlorophen
- Endothal
- Fentin
- Hydrated lime
- Nabam
- Quinoclamine
- Quinonamid
- Simazine
Table 3. Chemistry of copper sulfate

<table>
<thead>
<tr>
<th>STATUS:</th>
<th>ISO 765</th>
</tr>
</thead>
<tbody>
<tr>
<td>IUPAC:</td>
<td>copper(II) tetraoxosulfate</td>
</tr>
<tr>
<td></td>
<td>or copper(2+) tetraoxosulfate</td>
</tr>
<tr>
<td></td>
<td>or cupric sulfate</td>
</tr>
<tr>
<td>CAS:</td>
<td>sulfuric acid copper(2+) salt (1:1)</td>
</tr>
<tr>
<td>REG. NO.:</td>
<td>7758-98-7</td>
</tr>
<tr>
<td>FORMULA:</td>
<td>CuO₄S</td>
</tr>
<tr>
<td>ACTIVITY:</td>
<td>algicides</td>
</tr>
<tr>
<td></td>
<td>fungicides (copper fungicides)</td>
</tr>
<tr>
<td></td>
<td>herbicides (inorganic herbicides)</td>
</tr>
<tr>
<td></td>
<td>molluscicides</td>
</tr>
<tr>
<td>NOTES:</td>
<td>This substance is considered by the International Organization for Standardization not to require a common name. The parent acid is also considered not to require a common name, see sulfuric acid. In ISO 765-1976, the name given is “copper sulphate”, but ISO now requires the “f” spelling for sulfur and its compounds. This substance is a component of Bordeaux mixture, Burgundy mixture, and Cheshunt mixture.</td>
</tr>
<tr>
<td>STRUCTURE:</td>
<td><img src="" alt="CuO₄S" /></td>
</tr>
</tbody>
</table>

Algicides treat the symptoms of eutrophication without addressing the causal factors, hence are classified as a cosmetic treatment. Algicides may compound the eutrophication problem by killing algae and increased the loading of organic material to the hypolimnion, which further increased the oxygen demand and potential release of redox-sensitive metals and sediment bound phosphorus. Copper is often toxic to non-target organisms, including benthic invertebrates, fish and could create human health problems, and the non-target lethality of copper is greater in low alkalinity waters such as St. Mary Lake (Cook et al., 2005). Continued use of copper algicides may also result in unacceptable accumulations in sediments.

For example, Figure Eight Lake (Peace River district, Alberta) was treated with copper sulphate from 1980 to 1984 to suppress algal growth. A study in 1985 revealed the lake sediments were still toxic, and suppressed heterotrophic bacteria. Sulphate reducing bacteria were inhibited which blocked pyrite formation in the sediments and resulted in a
doubling of the sulphate concentration in the water column. Water column concentrations of copper exceeded levels required to kill sensitive invertebrates, hence the copper was highly bioavailable. The study concluded the long term toxicity of copper can interfere with fisheries in Figure Eight Lake (Prepas et al., 1987).

Additional analysis of Figure Eight Lake revealed that the depression of sulphate reducing bacteria had a significant impact on the biogeochemical cycling in the lake. Sulphate reducing bacteria produce sulfides which are retained within lake sediments as pyrite (i.e., FeS$_2$). Pyrite formation removes sulphate and hydrogen ions from the water column, and increases the alkalinity. In Figure Eight Lake, the addition of copper to suppress algae growth drastically inhibited the formation of pyrite, and the copper was toxic to sulphate reducing bacteria and other bacteria in the lake. The disappearance of amphipods post-treatment indicated significant alteration of the food web in the lake, and the authors concluded the use of copper sulphate as an algicide should be carefully reconsidered (Manning et al., 1987). In summary, algicides are not recommended for use in St. Mary Lake.

**Bacteria**

Some commercial products are available which claim the addition of dried bacteria and trace elements can markedly improve water quality conditions in small ponds. The trade name of these products includes Aquatron™, LymnoZyme™ and Waste and Sludge Reducer™. These products are advertised to contain “aerobic, facultative anaerobic bacteria and trace elements designed to enhance water quality and consume excess nutrients in pond environments”. There has been little information published in the scientific literature to assess the effectiveness of these treatments. The claims made by the manufacturers should be viewed with some caution:

http://www.pointfour.com/Products/Microbes/aquatron.html

http://www.pointfour.com/Products/Microbes/lymnozyme.html


Some bacteria do possess unique properties, especially those which have evolved in hot springs and meromictic lakes. These unique bacteria are often being explored to determine their suitability for specific waste treatment or industrial applications. For example, the *Halomonas campisalis* bacterium which was discovered in meromictic Soap Lake, WA is being studied to determine if it can be used to remove nitrate contaminants from fertilizer and explosive manufacturing plants. In the case of St. Mary Lake, it is unlikely that any bacteria could be purchased that are not already present in the lake, hence the probability of improving water quality through bacterial additions are minimal.
**Destratification**

Destratification is the most widely used procedure for circulating thermally stratified bodies of water. This technique increases dissolved oxygen in bottom waters by reducing thermal gradients and attempting to homogenize the entire water mass (Dunst et al., 1974). Destratification was first used in 1919 (Scott and Foley, 1919), and is typically accomplished by mixing cooler, more dense (and often anoxic) hypolimnetic water with warmer, less dense epilimnetic water. The bottom water absorbs oxygen and heat, and releases dissolved gases above equilibrium concentrations before sinking to a new equilibrium depth. Entire bodies of water can often be circulated from a single site and eventually become isothermal if the system is appropriately sized and the basin has single point of maximum depth.

The most common destratification method involves the injection of compressed air through perforated pipes or diffusers located neat the lake bottom (Figure 2). This is the most efficient destratification technique as the rising air bubbles entrain bottom waters to the surface where turbulent mixing is induced. Variations of this method include the Aero-Hydraulic Cannon. This is a low-head-, high-volume positive displacement pump which operates via periodic ejection of large air bubbles, which entrains water en route to the surface (Toetz et al., 1972).

![Figure 2. Destratification system installed at Yellow Lake, BC.](image)

Mechanical pumping has also been employed in several destratification projects. Hooper et al. (1952) destratified a small Michigan lake by pumping hypolimnetic water to the
surface, and Summerfelt et al. (1976) destratified a eutrophic reservoir by pumping epilimnetic water to the bottom via large axial flow pumps. Ridley et al. (1966) constructed several large angled jets in the reservoir floor and used the resulting jets of water to destratify a large aboveground reservoir that supplied potable water to the London, England water supply system. Many small lakes in B.C have been circulated with floating mechanical surface aerators (Figure 3) (Ashley and Nordin, 1999).

![Figure 3. Mechanical surface aerators installed throughout Southern Interior British Columbia (Ashley and Nordin, 1999).](image)

The principle disadvantage of destratification is greatly increased heat budgets. This is a major concern if one wishes to retain cooler hypolimnetic water for various domestic or industrial purposes or to maintain cold water fish habitat. Thermal gradients are greatly reduced and the entire water mass usually approaches the normal surface temperatures. This can be a serious problem during warm summer months as it eliminates cold water fish habitat in temperate lakes and reservoirs, and increases sediment oxygen demand. In addition, circulation currents may transport nutrients released from anaerobic sediments to the euphotic zone and stimulate primary production, if the system is undersized.

In Canada, an aesthetic objective of 15 °C has been established for the temperature of drinking water. The Health Canada rational is as follows:

1. The importance of temperature as a determinant of water quality is derived mainly from its relationship with other water quality parameters. Most of these relationships have a bearing on the aesthetic aspects of water quality; some are indirectly related to health;

2. The palatability of drinking water is to some extent dependent on temperature. The figure of 19 °C is often cited as a “limit” above which most consumers complain. At temperatures above 15 °C, the growth of nuisance organisms in the distribution system becomes a problem and could lead to the development of unpleasant tastes and odours. The effect of low temperature on water treatment
processes is controlled by altering the amounts of chemicals used in treatment; low temperature is not a barrier to the production of water of an acceptable quality;

3. The aesthetic objective for temperature is therefore 15 °C. Maintenance of the water temperature at or below this level offers several additional advantages. Low temperature aids in the retention of a chlorine residual by reducing the rates of reaction leading to hypochlorous acid removal; economic losses due to corrosion are reduced at low temperature; and cool water discourages the use of alternative sources that may be injurious to health:

http://www.hc-sc.gc.ca/ewh-semt/pubs/water-eau/doc_sup-appui/temperature/index_e.html#1

Given the importance of cool water for drinking water supply and salmonid habitat, destratification is not a suitable restoration technique for St. Mary Lake.

**Dilutional flushing**

Dilutional flushing involves the addition of low nutrient water to reduce the concentration of nutrients and planktonic algae in a lake. This technique physically removes algal biomass and nutrients from a lake, and exports them downstream. Dilutional flushing can be approached by two methods: pumping water out of the lake, thus permitting re-fill with (hopefully) nutrient poor ground water, or by routing additional quantities of nutrient poor surface waters into the lake (Dunst et al., 1974).

Pumping water out of St. Mary Lake would not be feasible, as a full lake level is required for potable water withdrawals, recreational use and for fish habitat. This technique would be most effective during the summer months when recreational and consumptive use are at their seasonal maximum. However, salmonids require a cold water refuge from the higher epilimnetic water temperatures, hence it is not realistic to consider this approach for St. Mary Lake. In addition, hydrology studies would have to be done in advance to determine the nutrient concentrations in the ground water.

Chain Lake (43.7 ha, $Z_{max} = 7.9$ m) near Princeton BC is the site of a dilutional flushing experiment intended to improve poor water quality. A 2 km long water diversion was built in 1968 to flush the lake with nutrient poor subalpine water from Shinish Creek. While sound in principle, the project has been ineffective due to the asynchrony of the water flow, the location of the diversion inflow too close to the lake’s outlet, and the magnitude of sediment phosphorus release (Murphy, 1985). Internal loading released about 78% of the lake’s phosphorus supply during summer months when the water availability, hence effectiveness of the flushing, was lowest. The authors concluded a larger diversion would suppress algae growth but may increase the growth of macrophytes, and recommend proceeding with a sediment dredging/hypolimnetic withdrawal experiment to increase the effectiveness of the Shinish Creek dilutional flushing (Murphy et al., 1987).
The concept of routing large quantities of nutrient poor water through St. Mary Lake is not feasible as there is no large source of low nutrient water available on Salt Spring Island. The theory of dilutional flushing is sound, as a reduction in lake retention time can reduce the trophic state of a lake, as demonstrated by the Vollenwieder loading equation:

\[ L_p = P \times z \times \frac{(1+T_w^{1/2})}{T_w} \]

where:
- \( L_p \) = annual P load (mg P m\(^{-2}\) yr\(^{-1}\))
- \( P \) = spring overturn phosphorus concentration (mg m\(^{-3}\))
- \( T_w \) = water residence time (years)
- \( z \) = mean depth (m)

Using Coquitlam Reservoir as an example, the effect of increased flushing rate/reduced residence time allows a water body to “tolerate” higher phosphorus loading rates before developing eutrophic symptoms (Figure 4). Regardless, this is not a realistic option for St. Mary Lake due to a shortage of large quantities of low nutrient water on Salt Spring Island.

![Critical Lake P Load](image)

**Figure 4.** Effect of increased flushing rate on critical P load for Coquitlam Reservoir.

**Diversion**

Nutrient diversion is defined as the rerouting of nutrient rich waters outside of a lake’s drainage basin. This technique can be effective only if the environment which receives the additional nutrients can assimilate the extra nutrient load without itself developing symptoms of eutrophication.

One of the most famous cases of diversion occurred in WA State, in which Metro Seattle diverted Seattle sewage around Lake Washington from 1964 to 1967 (Figure 5). The
subsequent recovery is well known in limnology, and the rapid recovery has been attributed to the relatively great depth (64 m max. depth), the short residence time (0.4/yr), relatively brief history of enrichment, and the fact that 88% of the nutrient loading was diverted (Cook et al., 2005).

Figure 5. Reduction in total phosphorus in Lake Washington following nutrient diversion.

This technique is not applicable to St. Mary Lake as the main source of phosphorus to the lake is internal loading from the sediments, with lesser amounts originating from septic tanks, groundwater, dust fall and natural watershed loading (Nordin and McKean, 1983). In addition, the lake’s water is highly valued as a potable water source and for recreational activities; hence diversion without a high quality supply of replacement water is not reasonable.

Dredging

Lake dredging can be an effective technique for restoring lakes, and is often recommended for deepening shallow lakes which have extensive macrophyte problems (e.g., Quamichan Lake). The main advantages of dredging are the physical removal of nutrient rich sediments, and increase in lake depth. The obvious drawbacks are the high costs, and the often insurmountable problem of how and where to dispose of the potentially huge volumes of fluid lake sediments, some of which may be contaminated.
Relatively few lakes have been dredged in BC because of these obstacles. Chain Lake, near Princeton, was partially dredged in the 1987 and 1988 as part of a hypolimnetic withdrawal project (Macdonald et al., 2004). Only a small area of the lake was dredged, yet this required a dredge disposal field approximately the size of a football field. Compounding the problem, the iron sulfides which had formed in the lake sediments began to oxidize on exposure to the air, and the dredged material had to be limed afterwards to prevent acid runoff.

The City of Burnaby has also embarked on a partial dredging program on Burnaby Lake. Portions of the lake sediments were contaminated with metals and polycyclic aromatic hydrocarbons (PAHs), and environmental assessment studies were required in advance of the dredging program to develop an appropriate mitigation strategy. Four options were considered, ranging from 262,000 to 563,000 m$^3$ (Table 4):

http://www.city.burnaby.bc.ca/cityhall/departments/engnrn/engnrn_whtshp/engnrn_whts hp_brnbyl/engnrn_whtshp_brnbyl_envfinal/engnrn_whtshp_brnbyl_envfinal_a.html

Table 4. Dredging options for Burnaby Lake, BC.

<table>
<thead>
<tr>
<th>Option</th>
<th>Dredge Volume (m$^3$)</th>
</tr>
</thead>
<tbody>
<tr>
<td>2</td>
<td>262,000</td>
</tr>
<tr>
<td>3</td>
<td>368,000</td>
</tr>
<tr>
<td>4a</td>
<td>403,000</td>
</tr>
<tr>
<td>4b</td>
<td>563,000</td>
</tr>
</tbody>
</table>

The total costs ranged from $60 to $71 per cubic meter, and total costs ranged from 18.6 million to $34.5 million (Table 5).

Table 5. Total cost and cost per cubic meter for dredging options on Burnaby Lake.

<table>
<thead>
<tr>
<th>Item</th>
<th>Option 2</th>
<th>Option 3</th>
<th>Option 4a</th>
<th>Option 4b</th>
</tr>
</thead>
<tbody>
<tr>
<td>Scenario 1</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Total Cost (Million)</td>
<td>$18.6</td>
<td>$24.5</td>
<td>$26.4</td>
<td>$33.8</td>
</tr>
<tr>
<td>Unit Cost / m$^3$</td>
<td>$71</td>
<td>$67</td>
<td>$65</td>
<td>$60</td>
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<tr>
<td>Scenario 2</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Total Cost (Million)</td>
<td>$20.6</td>
<td>$25.2</td>
<td>$27.1</td>
<td>$34.5</td>
</tr>
<tr>
<td>Unit Cost / m$^3$</td>
<td>$79</td>
<td>$68</td>
<td>$67</td>
<td>$61</td>
</tr>
</tbody>
</table>

Given the location of St. Mary Lake on Salt Spring Island, the high costs of dredging, and the difficulties in locating adequate dredge disposals sites, it is unlikely that this option would be technically feasible or economically viable for St. Mary Lake.
**Food web manipulations**

Food web manipulation, also known as “biomanipulation”, has been successfully used to improve water quality in numerous lakes. The technique relies on the phytoplankton filtering capability of large zooplankton populations, mainly *Daphnia* sp., which are enhanced by reducing planktivorous fish populations, or by increasing predation on planktivores through stocking of piscivorous fish. The technique is relatively inexpensive, but requires constant monitoring of the ratio between planktivorous and piscivorous fish, and is most effective in lakes which have already undergone reductions in P loading (Cook et al., 2005).

The concept of biomanipulation arose from a series of “top-down bottom-up” models which were proposed to explain trophic level interactions. In theory, large populations of piscivorous fish exert significant predation control on the small planktivorous fish, which in turn reduces the predation rate on *Daphnia* sp., which allows the large-bodied *Daphnia* populations to increase in number and graze down the edible algae thus increasing the transparency of the water (see Figure 6). If insufficient numbers of predatory fish are present, the small plantivorous fish can increase in number and graze down the large-bodied zooplankton, which reduces the grazing rate on algae, which decreases the transparency of the water.

These models were based on enclosure experiments, and multi-year studies of various oligotrophic and eutrophic lakes. Numerous regression models of total phosphorus, chlorophyll and fish biomass indicated that bottom-up forces (i.e. nutrients) had the strongest effect at trophic levels nearest the resources (Cook et al., 2005). In contrast, top-down forces had the strongest effect near the piscivore level. The predictions indicated that bottom-up control becomes increasingly important relative to top-down control in eutrophic lakes. Consequently, piscivores are less likely to exert a cascading effect on algae in the most productive lakes.

In reality, lakes are strongly influenced by year to year variations in precipitation, water budget and nutrient loading, and by climate and variable fish recruitment years. These stochastic events in turn affect both top and bottom trophic levels, hence the effects often lag between years and it is difficult to decipher the signal amongst the environmental noise. Most of the biomanipulation studies that demonstrated significant effects were in shallow eutrophic lakes (e.g., 3-6 m).

A few biomanipulation studies have been conducted in deeper lakes (e.g., Lake Mendota, Wisconsin), however, these lakes had classic planktivore populations (i.e. cisco and yellow perch (*Coregonus artedi* and *Perca flavescens*) and piscivore (i.e., northern pike and walleye (*Esox lucius* and *Stizostedion vitreum*)) populations. Biomanipulation is less effective in deep lakes, and requires a reduction in nutrient loading to be most effective.
Figure 6. Biomanipulation diagram indicating of top-down and bottom-up control various trophic levels (from Cook et al., 2005).

An innovative application of biomanipulation is being used in BC at Wahleach Reservoir (4,100 ha, $Z_{\text{max}} = 18.3$ m; near Hope, BC) to control an illegally introduced population of sticklebacks ($Gasterosteus aculeatus$) which were competing with stocked kokanee ($Oncorhynchus nerka$) for the available limnetic zooplankton. In this unique application, inorganic nutrients were applied to stimulate algae growth, and large sterile piscivores ($Oncorhynchus clarkii$) were introduced to control planktivores, in an attempt to remove the unwanted sticklebacks, and divert the large bodied zooplankton to the recreational fishery via kokanee. This is the first time biomanipulation has been conducted in combination with nutrient enrichment, and the experiment has been highly successful,
demonstrating the effectiveness of biomanipulation under the correct circumstances. *(Perrin et al., 2006).*

In conclusion, biomanipulation has been successful in some lakes, and it could potentially be used as a component of the St. Mary Lake water quality management program. An analysis of the St. Mary Lake fish population should be conducted to confirm the species composition of planktivores. If the planktivores are numerous and are suitable prey for piscivores, sterile cutthroat trout should be introduced to reduce the populations of planktivores. It is important to simultaneously enact catch and release regulations for the sterile piscivores, as they grow to large sizes, and can attract significant angling effort. However, biomanipulation is not a panacea for eutrophication management, and cannot be used as a substitute for the factors that caused eutrophication of the lake in the first place.

**Hypolimnetic aeration**

Hypolimnetic aeration was initially developed in post-war Switzerland as an innovative process to improve the water quality of Lake Bret, which is the water supply for Lausanne *(Mercier and Perret, 1949).* Water was mechanically pumped to a shore based splash basin where it was aerated and allowed to return by gravity flow through a pipe to the hypolimnion. The concept generated little interest for nearly twenty years, then re-emerged in the 1960’s due to the pioneering research efforts of Heinz Bernhardt in West Germany, and Richard Speece and Arlo Fast in the USA. In these systems, water was airlifted to the surface and oxygenated water returned to the hypolimnion via return tubes. Reports on the selective aeration capabilities of hypolimnetic aeration were published in the primary literature *(Bernhardt, 1967; Speece, 1971; Fast, 1971)* and the process attracted attention from various professional disciplines.

The interdisciplinary field of lake restoration blossomed in the 1970’s as limnologists and engineers developed new methods for restoring eutrophic lakes and reservoirs. Lake restoration refers to “… the manipulation of a lake ecosystem to effect an in-lake improvement in degraded, or undesirable conditions.” *(Dunst et al., 1974).* Coincidentally, hypolimnetic aeration had matured to a stage where it was viewed as suitable technology for the rapidly developing field of lake restoration. Hypolimnetic aeration systems were soon installed in several Western European countries *(Bernhardt, 1974; Verner, 1984), Canada (Ashley, 1983; McQueen and Lean, 1984; Ashley, 1985; McQueen and Lean, 1986; Ashley, 1987)* and the United States *(Fast and Lorenzen, 1976; Fast et al., 1976).* By the late 1980’s, hypolimnetic aeration was recognized by perceptive environmental engineers and limnologists as a valuable, multi-purpose water quality restoration technique.
For the purposes of this discussion, hypolimnetic aeration systems are grouped into three general categories: (1) full or partial lift with compressed air; (2) full or partial lift with pure oxygen or oxygen supplementation and (3) DBCA or Speece Cone aerators.

Types of Hypolimnetic Aerators

Full or Partial Lift Hypolimnetic Aeration with Compressed Air

A variety of hypolimnetic aerators designs have been proposed and at least 13 designs subjected to full scale testing (Fast and Lorenzen, 1976). However, the conventional full lift “Bernhardt” hypolimnetic aerator (Figure 7) has emerged as one of the more widely used designs due to its relatively simple design and availability of published data on site specific oxygen transfer and energy efficiency (Lorenzen and Fast, 1977; Taggart and McQueen, 1982; Ashley, 1985; Ashley et al., 1987; Little, 1995). Several design improvements have been suggested for full lift aerators (Lorenzen and Fast, 1977; Taggart and McQueen, 1982; Ashley, 1989, Ashley, 2000), however, relatively few have been subject to thorough laboratory and field testing.

Figure 7. Full lift type of hypolimnetic aerator.

Three design parameters relevant to full lift designs that have undergone limited laboratory and field trials include (1) depth of air injection; (2) separator box surface exchange area and
(3) diffuser orifice diameter. Varying the surface exchange area of the separator box was found to have no effect on induced water velocity, oxygen input, daily \( \text{O}_2 \) load, aeration efficiency \((\text{AE}; \text{g} \text{O}_2 \text{kWh}^{-1})\) and oxygen transfer efficiency \((\text{TE}; \%))\) (Ashley and Hall, 1990). The effect of diffuser depth on oxygen input was significant, however the effect was less than expected; but orifice diameter did exert a significant influence on oxygen input, daily oxygen load, TE and AE (Ashley et al., 1990). The importance of diffuser design was clearly demonstrated by a diffuser retrofit on a large Bernhardt style full lift hypolimnetic aerator, where the original 3,175 \( \mu \) diameter diffusers were replaced with 140 \( \mu \) diameter fine bubble silica glass diffusers. This increased the overall average daily oxygen input to St. Mary Lake (Salt Spring Island, B.C.) from 311 kg day\(^{-1}\) to 512 kg day\(^{-1}\) with no corresponding increase in operational costs (Ashley et al., 1990).

Partial lift hypolimnetic aerators have not experienced the widespread application of full lift systems, although the concept is technically sound (Figure 8). One reason may be the “overselling” of their capabilities relative to other designs of hypolimnetic aerators. The submerged design approach is particularly useful in high boat traffic areas and ice-prone regions, but it is more difficult to conduct routine inspections and performance monitoring. In extreme cases, undersized partial lift systems may have to be replaced, as they are less amenable to retrofitting than full lift designs. This was the unfortunate situation in Medical Lake, Washington where a partial lift hypolimnetic aeration system, with a coarse bubble diffuser, was unable to meet the hypolimnetic oxygen demand. After retrofitting with a smaller 600 \( \mu \) orifice diameter diffuser and a higher output compressor, the partial lift unit was still undersized and then experienced complete structural failure. It was replaced with two Bernhardt style full lift hypolimnetic aerators, which successfully oxygenated the hypolimnion and significantly improved the water quality of Medical Lake (Soltero et al., 1994).

Figure 8. Partial lift type of hypolimnetic aerator.
Full or Partial Lift Hypolimnetic Aeration with Pure Oxygen or Oxygen Supplementation

Pure oxygen has been used in several installations to increase the amount of oxygen transferred to the hypolimnion of eutrophic lakes and reservoirs. Although technically classified as a mechanical pump design of hypolimnetic aeration, Fast and Lorenzen (1976) describe two examples of side stream pumping (SSP) where pure oxygen was added to water pumped from a quarry lake and returned to the hypolimnion via a high pressure discharge line. This method increased the hypolimnetic oxygen concentration in Ottoville Quarry, Ohio to 21.5 mg·L⁻¹ which was the highest hypolimnetic oxygen concentration recorded to date in a stratified water body (Fast et al., 1977). Smith et al., (1975) used supplemental oxygen in Mirror Lake, Wisconsin to overcome the high rate of hypolimnetic oxygen demand, and measured oxygen concentrations in the full lift separator box of 13.4 mg·L⁻¹ on pure oxygen and 8.6 mg·L⁻¹, using a blend of 0.45 m³·min⁻¹ compressed air and 0.16 m³·min⁻¹ liquid oxygen.

Prepas et al. (1997) successfully used liquid oxygen to oxygenate the north basin of naturally eutrophic Amisk Lake, Alberta using a Deep Oxygen Bubble Injection (DOBI) system originally proposed by Speece (1971). This system injected up to 1.3 t·day⁻¹ of fine oxygen bubbles (20-1,500 µ diameter) at 33 m and was able to maintain hypolimnetic oxygen concentrations >1.7 mg·L⁻¹ in summer and >5 mg·L⁻¹ in winter. Gemza (1997) used an innovative full lift aerator design with an electric impeller in the separator box to increase water flow and oxygen transfer. Using Pressure Swing Adsorption (i.e., PSA) generated oxygen, this design oxygenated anoxic hypolimnetic water up to 8.0 - 11.5 mg·L⁻¹, and was instrumental in improving the water quality in Heart Lake and Whittaker Lake in Southern Ontario. The main limitations and concerns with the use of liquid oxygen are the high operating costs (Fast et al., 1976), availability in remote locations, and site storage/security issues. However, the development of PSA oxygen generation has changed the economics and storage/security issues of using oxygen, and increased the opportunities for innovative application in hypolimnetic aeration (Speece et al., 1990).

Downflow Bubble Contact Aeration

A promising new type of hypolimnetic aerator is the “Speece Cone” design (Figure 9). This concept was originally proposed in 1971 as a Downflow Bubble Contact Aerator (DBCA) with an open cone (Speece, 1971), then re-designed in 1990-91 with a closed cone and field tested for the first time in 1992 in Newman Lake, Washington (Doke et al., 1995). An innovative design of hypolimnetic aerator was necessary as Newman Lake is large (490 ha) but quite shallow, with a maximum depth of 9.1 m and a mean depth of only 5.8 m, which was too shallow for conventional partial or full lift hypolimnetic aerator designs. The Speece Cone installed in Newman Lake was 2.8 m in diameter, 5.5 m tall, with a 45 kW submerged axial flow pump for water circulation and two 37 kW air compressors supplying compressed air to two Pressure Swing Adsorption (PSA) on-site oxygen generation units.
The system was designed to supply 1,361 kg·d\(^{-1}\) of oxygen to the hypolimnion through a specially designed diffuser to avoid unintentional destratification and sediment disturbance (pers. comm., G. Lawrence, UBC, Vancouver, B.C.).

The system has performed extraordinarily well to date, with measured oxygen concentrations in the outlet pipe >30 mg·L\(^{-1}\) despite being located in only 8.7 m of water. The system increased average summer hypolimnetic oxygen concentrations to 5.5 mg·L\(^{-1}\) in 1992; however, thermal stratification was less stable due to two severe storm events (Thomas et al., 1994). An even larger Speece Cone (7 m in height), capable of supplying 8,000 kg·d\(^{-1}\) of oxygen, was installed and operated in Camanche Reservoir, California in 1993 and 1994 to improve water quality and prevent periodic fish kills in a salmonid hatchery that was relying on hypolimnetic water discharged from the anoxic hypolimnion. Unpublished reports indicate this system has also performed extremely well to date (pers. comm., A. Horne, U. California at Berkeley).

![Figure 9. Speece Cone type of hypolimnetic aerator.](image)

**Hypolimnetic withdrawal**

Hypolimnetic withdrawal is a technique used in stratified lakes to remove hypolimnetic nutrients via a deep pipe situated in the hypolimnion and discharging via an outlet below lake level, so the device operates as a siphon (Figure 10). Also know as an “Olszewski tube” after its Polish inventor, this technique has been used in numerous lake restoration projects throughout Western Europe and North America, and is one of the less expensive and minimally intrusive restoration techniques (Cook et al., 2005). Hypolimnetic retention time is reduced, which reduces the time available for anoxic conditions to
develop, and high concentrations of phosphorus and reduced compounds are exported in the discharged water.

The technique has proven effective, but has a number of deleterious effects in the receiving stream if not properly managed. The discharged water is low in oxygen, hence an oxygen sag will occur in the receiving stream, so an aeration system should be added to the outlet of the siphon. High concentrations of nutrients are released into the stream which usually results in excessive algal growth. Hydrogen sulphide gas and reduced metals are also discharged into the receiving stream, which can have deleterious effects on the stream biota.

Figure 10. Hypolimnetic outlet siphon installed in Lake Ballinger, near Seattle, WA. Note the aerator on the siphon discharge.

A hypolimnetic outlet siphon was installed in Chain Lake, BC in August 1993 in an attempt to improve the performance of the existing dilutional flushing efforts. The system consisted of a 150 m length of 45 cm pipe, reaching from the 6.2 m depth just outside of a dredged trench in the deepest section of the lake, and connected to a pre-existing culvert pipe through the outlet dam. The driving force that moves water though the pipe is the elevation differential between the lake and the downstream discharge point. The outlet end of the pipe was fitted with a splash fountain to increase oxygen
concentration in the water before it entered Hayes Creek. The system drains the water column below 6.5 m every 100 days, and drains the deepest region of the lake every 5-10 days. The system has operated for 76-156 days per year since 1994, and exported ~60 kg yr\(^{-1}\) of total phosphorus at an average discharge flow rate of 50 L s\(^{-1}\) (Macdonald et al., 2004). Minimal pre-siphon data exist, hence the effectiveness of the siphon system relied mainly on Secchi disk data collected by local residents. A statistical analysis of the data (Mann-Kendall test) indicates a 90%-plus probability that Chain Lake Secchi depth has increased in June and August following start-up of the siphon system. Oxygen depressions and elevated concentrations of total phosphorus, ammonia, iron and manganese were recorded downstream of the discharge fountain, however, no fish kills have been recorded in Hayes Creek in 9 years of operation (Macdonald et. al., 2004).

Figure 11. Schematic diagram of the Chain Lake, BC siphon installation.

In terms of applicability to St. Mary Lake, a hydraulic water, thermal and nutrient budget model would be required to determine the percentage of the annual phosphorus load which could potentially be exported via this technique. If the normal stream discharge from St. Mary Lake is quite low in the summer and fall months, it is uncertain that this technique would be effective, as there would be insufficient stream discharge to significantly dilute hypolimnetic phosphorus concentrations. If salmonids are present in the outlet stream, they could be negatively influenced by the discharge of water that is low in dissolved oxygen, and which could contain elevated concentrations of iron, manganese and hydrogen sulphide. Given the scarcity of high quality potable water on
Salt Spring Island, this would reduce the effectiveness of this technique. The hydraulic residence time of St. Mary Lake was estimated as 5.4 years during 1979-1981, hence this is not a rapidly flushed system (Nordin and McLean, 1983). The volume of the hypolimnion (5-16.5 m; 6,843,000 m$^3$) is considerably less than the whole lake volume (16,300,000 m$^3$), hence hypolimnetic residence time would be shorter; however, an updated nutrient and water budget model would still be required to assess the potential efficacy of this technique.

**P inactivation (alum, calcium and iron)**

**Aluminum salts**

Phosphorus inactivation is one of the most widely used lake restoration techniques, and has been applied to hundreds of eutrophic lakes throughout Western Europe and the United States. The purpose is to reduce a lake’s nutrient content by removing P from the water column, and by retarding the release of P from sediments. P inactivation is most effective in lakes that have lower flushing rates, minimal macrophyte cover and receive a significant portion of their annual P supply from internal loading. In addition, lakes that have not responded to alternative lake restoration techniques because their sediments are enriched from a history of high external loading are typically considered as candidates for P inactivation. Some stratified lakes have demonstrated reduced internal loading for 12-15 years following treatment with aluminum salts (Cook et al., 2005).

Aluminum salts (e.g., alum or sodium aluminate) are the most widely used P inactivation chemical, followed by iron salts, and calcium compounds. When alum or aluminum salts are added to water, they dissociate to form aluminum ions, which then hydrate, and through a series of hydrolysis reactions, form colloidal aluminum hydroxide floc (Equations 1 – 3):

$$\text{Al}^{+3} + \text{H}_2\text{O} \leftrightarrow \text{Al(OH)}^{+2} + \text{H}^+$$

(1)

$$\text{Al(OH)}^{+2} + \text{H}_2\text{O} \leftrightarrow \text{Al(OH)}_2^+ + \text{H}^+$$

(2)

$$\text{Al(OH)}_2^+ + \text{H}_2\text{O} \leftrightarrow \text{Al(OH)}_3 (s) + \text{H}^+$$

(3)

The aluminum hydroxide floc quickly settles to the lake sediment and sorbs P in the water column en route, and continues to sorb and retain P within the floc lattice structure, even under reducing sediment conditions (Cook et al., 2005). Alum is one of the most widely used chemicals in the potable water treatment industry, and has been used for over 200 years to remove suspended sediment in water treatment plants.

The pH of the water body determines which hydrolysis products will form, and their solubility. In the pH range of 6-8, Al(OH)$_3$ dominates. At pH 4 to 6, various soluble intermediate forms occur, and at pH less than 4, hydrated and soluble Al$^{+3}$ dominates.
(Cook et al., 2005). Above pH 8, the aluminum hydroxide becomes increasingly prevalent as aluminate ion, which has increasing solubility and can lead to release of P which was previously sorbed to an aluminum salt (Equation 4):

\[
\text{(4)} \quad \text{Al(OH)}_3 \ (s) + \text{H}_2\text{O} \leftrightarrow \text{Al(OH)}_4^- + \text{H}^+
\]

The properties of Al(OH)₃ that are most useful in lake restoration are its apparent low toxicity to lake biota, its ability to sorb large amounts of particulate and soluble P, and the binding of P to the floc (Cook et al., 2005). In contrast to iron, which is redox sensitive, anoxic or anaerobic sediment conditions do not solubilize P from the settled floc and allow P release, although P may be released if high pH occurs.

Hydrogen ions are liberated when an aluminum salt is added to water, and H⁺ increases in proportion to the decline in alkalinity. In lakes such as St. Mary which have low alkalinity (~ 30 mg L⁻¹; Nordin and McKean, 1983), alum treatment may cause a significant decline in pH at low or moderate doses, potentially leading to increasing concentrations of toxic, soluble aluminum forms, including Al(OH)₂⁺ and Al⁺³. This limits the concentration of alum which can be added, so various buffers are usually added to the alum slurry as it is being added. These include sodium hydroxide, calcium hydroxide and sodium carbonate, which all increase the cost and logistical requirements for whole-lake treatments.

The toxicity of aluminum is complicated by the variable solubility of several forms, as shown in Equations 1-4. The effect of alum treatment on invertebrates is variable, ranging from significant declines in benthic invertebrate density and species richness in soft water lakes, to no change or increases in benthic invertebrate abundance in hardwater lakes. The toxic effect of monomeric Al on fish in the pH range 4.5-5.5 is thought to be due to interactions of cationic species at the gill surface, which causes respiratory and ion regulatory dysfunction (Cook et al., 2005). Exposure of smallmouth bass (Micropterus dolomieu) to low doses of Al(OH)₃(s) at pH 7.3 to 7.5 caused significantly reduced activity in 30 day continuous exposures. In softwater Liberty Lake, near Spokane WA, an in situ study conducted on rainbow trout (Oncorhynchus mykiss) examined the effect of low does of Al (i.e. 0.5 mg Al L⁻¹). No mortalities were observed during the treatment, however, over the ensuing 5 weeks of observation, trout in treated area cages exhibited increased mortality relative to the control cages, although no gill hyperplasia was noted (Cook et al., 2005).

Calcium

Calcium has been occasionally used to inactivate P in lakes by mimicking the “whiting” reaction that occurs naturally in hardwater lakes during periods of intense photosynthetic
activity (Equation 5). The resulting calcium carbonate precipitate sorbs P, especially when the pH exceeds 9.0.

\[
\text{(5)} \quad \text{Ca(HCO}_3\text{)}_2 \leftrightarrow \text{CaCO}_3 \text{(s)} + \text{H}_2\text{O} + \text{CO}_2
\]

The first whole-lake treatment of this concept was conducted at hardwater Frisken Lake (33.8 ha, Z_{max} = 12 m), near Kamloops, BC. A total of 23 metric tonnes of Ca(OH)\textsubscript{2} (slaked lime) was added to the epilimnion in the summer of 1983, and 16 metric tonnes were added the following spring. P precipitation from the epilimnion was significant, and noticeable increases in transparency followed, including removal of blue green algae (Aphanizomenon flos-aquae). However, the calcite precipitate re-dissolved in the lower pH hypolimnion, reducing the effectiveness of this treatment (Murphy et al., 1988).

Iron salts

Iron has also been used on occasion to remove P from the water column. Ferrous (Fe\textsuperscript{2+}) iron is oxidized to ferric (Fe\textsuperscript{3+}) iron by reacting with dissolved oxygen in the water column, where it forms an insoluble precipitate which sorbs P as it settles through the water column (Equation 6):

\[
\text{(6)} \quad \text{Fe}^{2+} + \frac{1}{4} \text{O}_2 + 2\text{OH}^- + \frac{1}{2} \text{H}_2\text{O} \leftrightarrow \text{Fe(OH)}_3 \text{(s)}
\]

FePO\textsubscript{4} also forms, but the primary mechanism of P removal from the water column and retention in the sediments is sorption by Fe(OH)_3, and is most effective between pH 5 to 7 (Cook et al., 2005). The principle concern with this technique is the redox chemistry of iron, which reverts to Fe\textsuperscript{2+} in anoxic sediments, and releases the previously sorbed P.

A series of iron treatments were conducted in Black Lake (4 ha, Z_{max} = 9.0) near Kaledon, BC in the early 1990s. A total of 280 kgs of Fe\textsubscript{2}(SO\textsubscript{4})\textsubscript{3} was added in June 1990, 172 kgs of FeCl\textsubscript{3} added in June 1991, and 286 kg of FeCl\textsubscript{3} applied in May, 1992. The 1990 application was poured into the lake in the propeller wash of a small boat, whereas the 1991 and 1992 applications were done using a peristaltic pump. This resulted in whole lake concentrations of 1.97 mg, 1.21 and 2.01 mg L\textsuperscript{-1} in 1990, 1991 and 1992 respectively.

The 1990 treatment precipitated P from the epilimnion, but also stimulated an intense Aphanizomenon bloom as the algae had been iron limited. This increased the lake’s pH to 9-10, which triggered a calcite event, which further reduced the epilimnetic P concentration to 20-30 ug L\textsuperscript{-1}. The iron and calcite were released in the anoxic hypolimnria, and soluble reactive P concentrations increased to > 500 ug L\textsuperscript{-1}. The 1991 iron concentrations were too low (i.e, 1.21 mg L\textsuperscript{-1}) to cause a significant precipitation of
P. The 1992 application reduced surface P to 35-45 ug L\(^{-1}\), and presented algal blooms for the remainder of the summer (Hall et al., 1994).

In summary, aluminum salts can be used as a stand-alone treatment to remove P from the water column and permanently bind P in the sediments. The Al(OH)\(_3\) (s) complex is apparently resistant to redox changes. However, the additional of aluminum salts produces H\(^+\) ions and pH will decrease at a rate dictated by the lake’s alkalinity and Al dose. This can lead to high concentrations of soluble and potentially toxic aluminum species, and unless the lake is well buffered, or buffers are added, the use of aluminum salts may not be appropriate.

Sorption of P to calcium can lead to significant removals, but only in hard water lakes where the calcium carbonate precipitation reaction can be triggered, but most of the P will be released in the acidic pH hypolimnetic waters. Iron additions can also remove P from the water column, but the redox sensitivity limits its application to those instances where the hypolimnion is aerobic, or is used in combination with an aeration system to maintain oxidizing conditions in the hypolimnion. In some cases where the sediment ratio of Fe to P has declined below 15:1 due to pyrite formation, iron addition, in combination with hypolimnmonic aeration, could be a potentially effective treatment combination.

**Sediment oxidation**

Sediment oxidation is a rarely used *in situ* restoration technique that was pioneered in Sweden by Dr. William Ripl. The technique involves the injection of a series of chemicals directly into the lake sediment using a “harrow” type device with an air injection system (Figure 12). The initial treatment usually involves ferric chloride (FeCl\(_3\)) to remove hydrogen sulphide and form ferric hydroxide (Fe(OH)\(_3\)). The next step involves injection of lime (Ca(OH)\(_2\)) to increase the pH of the sediments to an optimum range and encourage microbial denitrification. The final step involves injection of calcium nitrate (Ca(NO\(_3\))\(_2\)) to act as an electron acceptor. Ferric chloride and lime additions may be unnecessary in some cases. This technique treats the top 15 to 20 cm of sediment, and is best suited for small stratified lakes that have anaerobic sediments, high interstitial P concentrations, and where iron oxidation-reduction reactions control P exchange between sediments and the overlying water (Cook et al., 2005).

In Lake Lillesjön, Sweden, this treatment reduced the interstitial P content in the upper 20 cm of sediment by 70-85%. Nitrogen was lost through denitrification and release of nitrogen gas, and recycling of P and N was reduced by 80-90%. Sediment oxygen demand was reduced by ~30% (Ripl and Lindmark, 1978). This technique has also been applied in Lake Trekanten (Sweden), White Lough (Ireland) and Long Lake (Minnesota). The results have been somewhat successful, however, the logistical difficulties associated with negotiating a harrow around the lake bottom sediments and the cost (i.e., $469,000 in 2002 $US for 87 ha Lake Trekanten) have limited the use of this innovative P inactivation technique.
Figure 12. Schematic diagram of the “Riplox” system, showing the various mechanical components.

**Sound**

A little known technique for controlling algae is the use of ultrasound. This technique is not discussed in any recent lake restoration literature, however, the company manufacturing the makes positive claims about their product. The manufacturer, SonicSolutions™, provide the following statements in their advertising literature and web-site:

“The use of ultrasound for controlling algae has been known for some time. However, the practical use of such technology is relatively recent and utilises the resonance effects of ultrasonic waves on the algae cell. The ultrasonic waves are derived from the creation of certain sound vibrations with periodic interruptions. A submerged transducer that is
specifically designed and purpose built to be small and water resistant generates the ultrasonic vibrations. These shock waves are directed at the vacuole of the algae. Initial observations show that the shock waves probably weaken the cell membranes causing the algae to collapse in on themselves and sink out of suspension.

This new approach is environmentally friendly, cost effective and uses no chemicals. These ultrasonic vibrations, which are inaudible to people, are no threat to human beings, animals or fish.

These ultrasonic devices are used in horticulture, aquaculture, potable and wastewater applications. The transducers used are capable of emitting ultrasonic vibrations up to a range of 200 meters, covering a radius of 20 m.”

The company provides the following coverage areas for their various LG Sonic Units (Tables 6 and 7):

**Table 6. Coverage area for algae.**

<table>
<thead>
<tr>
<th>LG Sonic Model</th>
<th>Sq. meters</th>
<th>Range (m)</th>
<th>Power (watts)</th>
<th>Cable length (m)</th>
</tr>
</thead>
<tbody>
<tr>
<td>XXL</td>
<td>18,082</td>
<td>150</td>
<td>12</td>
<td>17</td>
</tr>
<tr>
<td>XL</td>
<td>8,036</td>
<td>100</td>
<td>10</td>
<td>17</td>
</tr>
<tr>
<td>Tank</td>
<td>723</td>
<td>30</td>
<td>9</td>
<td>10</td>
</tr>
<tr>
<td>Pool*</td>
<td>2,009</td>
<td>50</td>
<td>9</td>
<td>10</td>
</tr>
<tr>
<td>SSS</td>
<td>80</td>
<td>10</td>
<td>8</td>
<td>10</td>
</tr>
</tbody>
</table>

* Note: pool radius and coverage area assumes chlorinated water.

**Table 7. Coverage area for blue-green algae.**

<table>
<thead>
<tr>
<th>LG Sonic Model</th>
<th>Sq. meters</th>
<th>Range (m)</th>
<th>Power (watts)</th>
<th>Cable length (m)</th>
</tr>
</thead>
<tbody>
<tr>
<td>XXL</td>
<td>128,584</td>
<td>400</td>
<td>12</td>
<td>17</td>
</tr>
<tr>
<td>XL</td>
<td>57,148</td>
<td>267</td>
<td>10</td>
<td>17</td>
</tr>
<tr>
<td>Tank</td>
<td>5,143</td>
<td>80</td>
<td>9</td>
<td>10</td>
</tr>
<tr>
<td>Pool**</td>
<td>5,143</td>
<td>80</td>
<td>9</td>
<td>10</td>
</tr>
<tr>
<td>SSS</td>
<td>571</td>
<td>27</td>
<td>8</td>
<td>10</td>
</tr>
</tbody>
</table>

** Note: pool radius if used in non-chlorinated system is same as tank.

The manufacturer web-site is: [www.sonicsolutionsllc.com](http://www.sonicsolutionsllc.com)

The Canadian Distributor of this product ([www.CanadianPond.ca](http://www.CanadianPond.ca)) has provided a cost estimate to treat Bouchie Lake, a highly eutrophic 134 ha lake located 10 km northwest of Quesnel. Their quote, dated May 17, 2007 indicates a total of seven XXL Ultrasonic algae killer units with low profile floats would be required, using 25 watts per unit, with a cable length of 187 m. The units would operate on 120 volt AC current. The cost of their proposal, including GST and PST, was $19, 616 CDN.
If there is any credibility to their claims, this would be a very cost-effective method for controlling blue-green algae. Unfortunately, no peer reviewed information could be located to verify the accuracy of their claims, hence it is not possible to evaluate this technique at the present time. The logistics of installing and maintaining an AC electric cable network on a lake the size of St. Mary (or Bouchie Lake) would not be a trivial task, hence this cost estimate is likely to be significantly less than the actual installed cost.

Needless to say, this is an intriguing technology. Hopefully some studies will emerge in the peer-reviewed literature and provide independent verification of the technology.

Figure 13. Advertised sound coverage area for LG Sonic XXL Unit.
**Watershed management and protection**

One of the most important long term water quality improvement strategies for eutrophic lakes is watershed protection and management to minimize point and non-point source nutrient loading. While various lake restoration techniques, especially ambitious programs like lake dredging, alum treatment and hypolimnetic aeration can create significant short term improvements in water quality, their long term effectiveness will be compromised if watershed nutrient loading is not effectively managed, and ultimately reduced.

*Nordin and McKean* (1983) conducted a detailed assessment of the St. Mary Lake watershed, and concluded the largest phosphorus source was internal loading (58-71%), followed by septic tanks (24-36%), stream inputs (3-5%) and groundwater (1%) (*Table 8*):

**Table 8. Phosphorus budget for St. Mary Lake.**

<table>
<thead>
<tr>
<th>P sources</th>
<th>1979-80 P load (kg)</th>
<th>1980-81 P load (kg)</th>
<th>1981-82 P load (kg)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Internal loading</td>
<td>465 (58%)</td>
<td>560 (62%)</td>
<td>850 (71%)</td>
</tr>
<tr>
<td>Septic inputs</td>
<td>290 (36%)</td>
<td>290 (32%)</td>
<td>290 (24%)</td>
</tr>
<tr>
<td>Streams</td>
<td>40 (5%)</td>
<td>40 (5%)</td>
<td>40 (3%)</td>
</tr>
<tr>
<td>Groundwater</td>
<td>10 (1%)</td>
<td>10 (1%)</td>
<td>10 (1%)</td>
</tr>
<tr>
<td>Total</td>
<td>805</td>
<td>900</td>
<td>1,190</td>
</tr>
</tbody>
</table>

Since internal loading is the largest source, any restoration strategies designed to improve the water quality in St. Mary lake need to address this P source. The next largest P source is septic tank inputs, hence a watershed management plan must be adopted which reduces the watershed loading of phosphorus into St. Mary Lake. This would involve an entire suite of watershed management activities ranging from resident lifestyle changes, rainwater management, lawn maintenance to installing pump and haul type of septic systems to prevent the P from reaching the lake.

Unfortunately, many districts avoid implementing watershed management plans because they are not a “quick fix” like many of the other lake restoration techniques, and involve considerable dialogue with watershed residents, some of whom may not share the same view as the lake managers, nor have the financial resources to undertake engineering modifications to their septic systems. These issues are best addressed through formation of a watershed management council and public education/outreach programs to explain the linkages between human activity in the watershed, nutrient sources and the resulting water quality in the lake, and the subsequent costs of treating the water to Canadian drinking water standards.
An additional impediment is the time lag between implementing watershed management plans and observing improvements in lake water quality. As a result, watershed management programs often get relegated to the “back burner” while more urgent issues get addressed. Obviously some higher priority issues will occur that require immediate attention, however, failure to enact watershed nutrient management practices will ultimately result in increased future lake restoration costs. In the case of St. Mary Lake, it is unlikely that any lake restoration technique will be successful over the long term without controlling watershed nutrient loading, as additional growth pressures in the watershed will increase watershed nutrient loading, which will eventually negate the benefits of any existing lake restoration programs.

Discussion

The preceding review of lake restoration techniques is intended to provide a general overview of the range of available techniques, and includes some initial comments on the suitability of the available techniques to St. Mary Lake. The following table summarizes the aforementioned information, and lists the advantages, disadvantages and identifies techniques that could be used as stand-alone treatments, or used in conjunction with a hypolimnetic aeration system (Table 9):

Table 9. Summary of lake restoration options and techniques for St. Mary Lake.

<table>
<thead>
<tr>
<th>Technique</th>
<th>Advantages</th>
<th>Disadvantages</th>
<th>Suitability as stand-alone technique</th>
<th>Compatibility with hypolimnetic aeration</th>
</tr>
</thead>
<tbody>
<tr>
<td>Algicides</td>
<td>Rapid effect, inexpensive, easy to apply</td>
<td>Cosmetic treatment, increases organic loading, non-target toxicity, toxic accumulation in sediments</td>
<td>Not suitable for St. Mary Lake</td>
<td>No</td>
</tr>
<tr>
<td>Bacteria</td>
<td>Inexpensive, easy to apply</td>
<td>Unknown effectiveness</td>
<td>Not suitable for St. Mary Lake</td>
<td>Unknown</td>
</tr>
<tr>
<td>Destratification</td>
<td>Rapid effect, less expensive than hypolimnetic aeration</td>
<td>May trigger algae bloom, loss of cold water in hypolimnion, will cause increased bacterial re-growth in water distribution system, requires ongoing maintenance</td>
<td>Not suitable for St. Mary Lake</td>
<td>No</td>
</tr>
<tr>
<td>Dilution and flushing</td>
<td>Low energy costs</td>
<td>Loss of cold water in hypolimnion, will cause increased</td>
<td>Not feasible for St. Mary Lake due to water scarcity</td>
<td>No</td>
</tr>
<tr>
<td>Method</td>
<td>Cost</td>
<td>Effect</td>
<td>Feasibility</td>
<td></td>
</tr>
<tr>
<td>-----------------------------</td>
<td>-----------------------</td>
<td>---------------------------------------------</td>
<td>----------------------------------------------------------------------------</td>
<td></td>
</tr>
<tr>
<td><strong>Diversion</strong></td>
<td>Low energy costs</td>
<td>Not effective in lakes with high internal nutrient loading, requires large volumes of low nutrient water</td>
<td>Not feasible for St. Mary Lake due to water scarcity</td>
<td></td>
</tr>
<tr>
<td><strong>Dredging</strong></td>
<td>Physical removal of nutrient rich sediments, deepening of lake</td>
<td>Very high cost, requires large area for sediment disposal, sediments may be contaminated</td>
<td>Yes</td>
<td></td>
</tr>
<tr>
<td><strong>Food web manipulation</strong></td>
<td>Low cost</td>
<td>Requires ongoing management, results can be unpredictable, requires sterile triploid Cutthroat trout as piscivores</td>
<td>Yes</td>
<td></td>
</tr>
<tr>
<td><strong>Hypolimnetic aeration</strong></td>
<td>Proven technology, retains cold water hypolimnion</td>
<td>Expensive, requires ongoing maintenance, physical presence in lake</td>
<td>Yes</td>
<td></td>
</tr>
<tr>
<td><strong>Hypolimnetic withdrawal</strong></td>
<td>Proven technology, low energy costs, exports nutrients from system</td>
<td>Moderately expensive to install, negative impacts on receiving stream (oxygen sag, heavy metals, eutrophication), loss of cold water hypolimnion</td>
<td>Not feasible for St. Mary Lake due to water scarcity</td>
<td></td>
</tr>
<tr>
<td><strong>P inactivation – alum</strong></td>
<td>Rapid effect, long term binding in the sediment, redox insensitive</td>
<td>Expensive, cosmetic treatment, non-target toxicity</td>
<td>Yes</td>
<td></td>
</tr>
<tr>
<td><strong>P inactivation – calcium</strong></td>
<td>Rapid effect</td>
<td>Not applicable to soft water lakes</td>
<td>No</td>
<td></td>
</tr>
<tr>
<td><strong>P inactivation – iron</strong></td>
<td>Rapid effect</td>
<td>Not suitable for lakes with anoxic hypolimnion, redox</td>
<td>No</td>
<td></td>
</tr>
</tbody>
</table>
A wide range of lake restoration techniques have been developed since the early 1970s, however, many of the techniques are not suitable for St. Mary Lake for a variety of reasons. Algicides, which are heavily marketed in the United States, are rarely used in British Columbia for obvious reasons. They are a purely cosmetic treatment at best, and may be toxic to the lake’s biota. Several of the proven techniques (e.g., dilutional flushing and diversion) require large volumes of low nutrient water, hence are not feasible on Salt Spring Island. Since St. Mary Lake is a potable water source, and cold water salmonids are valued, techniques that eliminate the cold water hypolimnion, such as destratification and hypolimnetic withdrawal, are not suitable. Some of the techniques have unknown effectiveness (e.g., bacteria and sound). It is difficult to imagine that any species of bacteria capable of improving water quality in St. Mary Lake is missing from the lake’s ecosystem.

The ultrasound concept, while intriguing, just seems to be too good to be true, and it would be logistically difficult to install and maintain a network of 120 volt AC wiring in the lake, given the voltage drop that occurs over relatively short lengths of AC cable. Lake dredging is an effective lake restoration technique; however, the very high costs and extensive land area required for sediment disposal make this an impractical solution for St. Mary Lake. In-situ sediment oxidation would be equally complex, expensive and logistically challenging for St. Mary Lake. Alum treatment, which is widely used in the United States, is not viewed with the same enthusiasm in Canada, particularly in potable water lakes with recreational fisheries. Calcium treatment would be ineffective due to the low alkalinity and pH of St. Mary Lake, whereas iron treatment would not be suitable as a stand-alone treatment due to its redox sensitive nature.

In reality, only a limited sub-set of lake restoration techniques are suitable for St. Mary Lake. The core activity of a water quality improvement program must include a watershed management program to control point and non-point source nutrient loading. Without this approach, no lake restoration technique will be effective in the long term. Assuming an effective watershed management program is implemented, the suitable water quality improvement techniques for St. Mary Lake are (1) hypolimnetic aeration, (2) food web biomanipulation, and (3) hypolimnetic iron addition.
Hypolimnetic aeration is well suited to St. Mary Lake as internal loading is the single largest P source to St. Mary Lake, and the phosphorus in the lake is sensitive to changes in the oxidation-reduction potential. If the hypolimnion can be kept aerobic the single largest P source can be significantly reduced, and water quality will improve accordingly. Hypolimnetic aeration conserves the cold water resource in the lake for potable water withdrawals and fisheries purposes.

Hypolimnetic aeration has a proven performance record in St. Mary Lake, and during the late 1980s and early 1990s it was responsible for a noticeable improvement in lake water quality and trout angling success. SCUBA divers noticed the profundal sediments had a reddish brown oxide crust, exactly as predicted from oxidation-reduction chemistry. A new hypolimnetic aeration system, which incorporates advances in oxygen transfer technology developed during the 1990s, is the logical core option for in-lake treatment of water quality in St. Mary Lake.

Food web biomanipulation should also be considered if large bodied zooplankton are being excessively grazed by planktivorous fish. Triploid cutthroat trout are an ideal top predator for use in BC lakes, and have demonstrated their effectiveness in controlling planktivorous fish in Wahleach Reservoir, near Hope (Perrin et al., 2006).

Iron additions to the hypolimnion, in combination with hypolimnetic aeration, should be considered if the sediment iron-phosphorus ratio is less than 15:1 due to pyrite formation. This would increase the sediment binding capacity for phosphorus and increase the effectiveness of a hypolimnetic aeration system to retain P in the profundal sediments. Since iron is redox sensitive, this technique cannot be used as a stand-alone treatment, and should only be used in combination with hypolimnetic aeration.
Literature Cited


