Little St. Germain Lake Aluminum Sulfate Treatment Proposal

Prepared for
Little St. Germain Lake Protection and Rehabilitation District

March 2009

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BARR

4700 West 77th Street
Minneapolis, MN 55435-4803
Phone: (952) 832-2600
Fax: (952) 832-2601

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Table of Contents

List of Tables

List of Figures

List of Appendices

1.0 Executive Summary

At the request of the Wisconsin Department of Natural Resources, an evaluation has been conducted to examine the need, expected benefits, and other potential consequences of using alum to treat the bottom sediments of Little St. Germain Lake. There has been extensive work conducted on Little St. Germain Lake, largely because water quality in the lake is in the eutrophic range and appears to have worsened over the last two decades. Seasonal trends in water quality show that degradation occurs during the summer when phosphorus contributions from inflows are lower but internal phosphorus loading is elevated. The degraded water quality has negative impacts on aesthetics, fish populations, and macrophytes leading to lower enjoyment of the lake by residents and others who use the lake for these purposes.

Recent studies have focused on methods to reduce external or internal phosphorus loads in order to improve water quality in the lake. A study conducted for the Little St. Germain Lake Protection and Rehabilitation District examined the feasibility of treating inflow from Muskellunge Creek with alum to improve water clarity in the lake. Muskellunge Creek, the main inlet to the lake, enters in the East Bay. Inflows from the creek do affect water quality but the majority of phosphorus loading occurs during the early part of the growing season and modeling results indicate that treating the inflow would provide limited benefit in late summer when water quality is usually the most degraded.

Due to the high cost of inflow alum treatment, an additional study was conducted to determine the benefit of applying alum directly to the lake to control internal phosphorus loading. Modeling showed that decreasing internal phosphorus loading would reduce surface water phosphorus in the lake by a greater amount than inflow treatment, especially in the later part of the summer.

Alum treatment was decided upon as the most economical solution to improve water quality based on the cost and expected benefits. Chemical modeling was conducted using a USGS program called PHREEQ to evaluate the effect of different alum doses on pH in the lake. Because of the low alkalinity of the lake, the dose needed to inhibit internal phosphorus loading would have to be split and applied for several successive years. It is recommended that the treatment be split into three equal phases (doses) to minimize the use of costly buffers. It is possible that two treatments (conducted in successive years) will be adequate to inhibit internal loading, however, sediment coring is recommended after the second treatment along with ongoing water quality monitoring to determine the potential need for the third treatment. This approach will also allow for adaptive management based on monitoring after the initial phase(s) of treatment. The alum dose and expected cost for alum application in each phase are shown in Table EX-1. Additional monitoring and contract bidding costs of $7,750 will be needed for each phase of treatment beyond the first treatment conducted in 2009.

Table EX-1. Alum dose and expected cost of treatment (per phase). Cost includes lime addition.

Treatment Zone	Alum Applied (gallons)
East	57250
Lower East	66690
Total	123940
Treatment cost per phase	**$ 202,000**

Potential effects of alum treatment on benthic invertebrates, fish, and macrophytes were considered in this proposal. A review of published studies on the potential effects of alum treatment show that effects on aquatic life are limited as long as the pH of lake water during treatment remains above 6. Due to the expected increase in water clarity, more lake bottom area will be available for plant colonization but studies have shown that the diversity or health of a macrophyte community increases with an increase in water clarity.

Post-treatment monitoring will be conducted to assess the effectiveness and longevity of the treatment with regard to controlling internal phosphorus loading and improving water quality in Little St. Germain Lake. It is expected that the water quality benefits of the alum treatment will last for a minimum of 10 years (case studies have demonstrated that improvement can last for approximately 5 to 20 years). It can be expected that the proposed alum treatment will have greater longevity compared to past treatments on other lakes because of improved alum dosing procedures.

2.0 Historical Conditions

Little St Germain Lake is located in Vilas County, WI in the town of St. Germain. The lake is highly sought after for recreational and other activities which include fishing, waterskiing, swimming, and boating.

2.1 Land Use

Settlers first arrived in the town of St. Germain in 1903. Since around that time, logging, recreation, and a small amount of farming and development have dominated the direct watershed areas around the lake. Current land uses are shown in Figure 1.

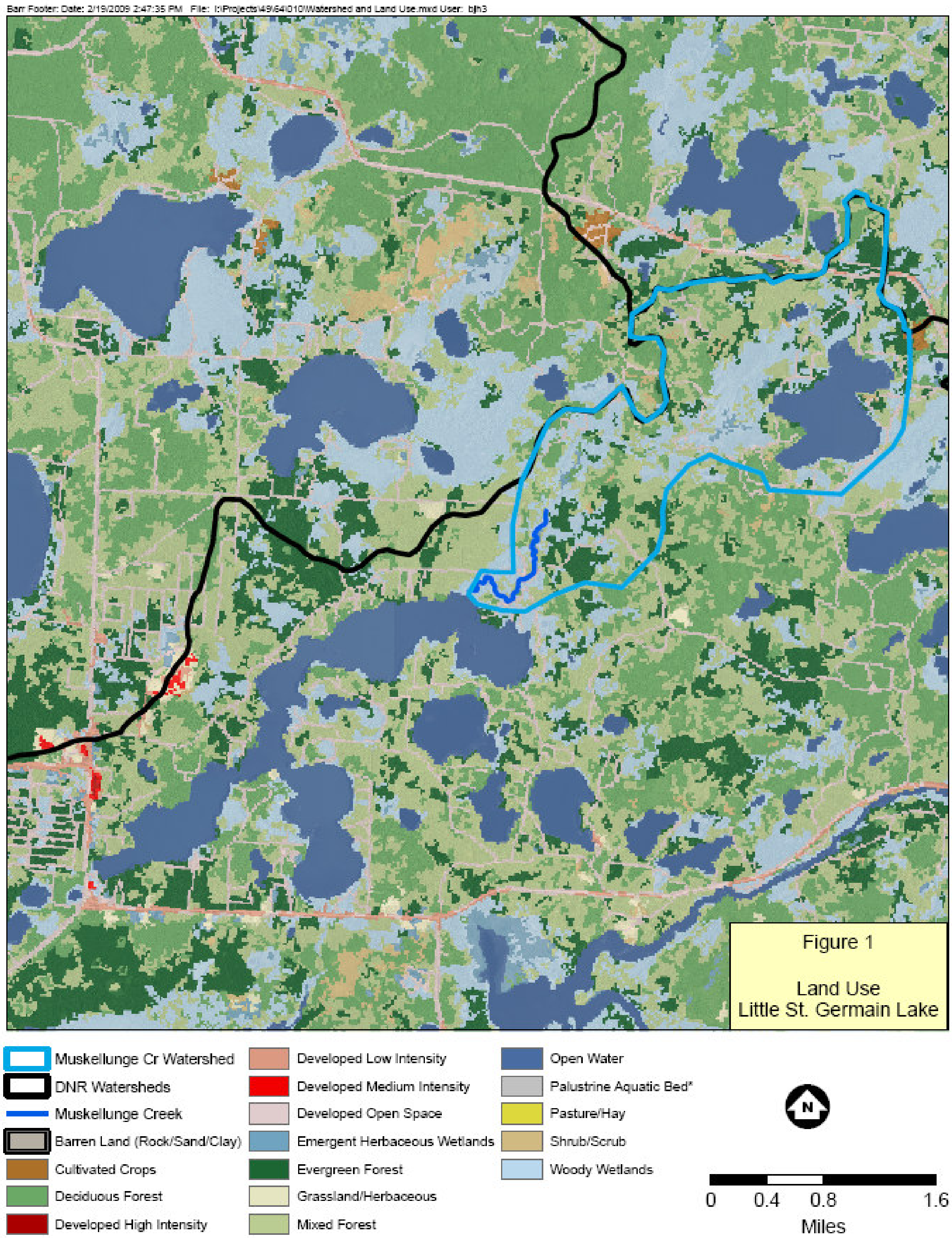


Figure 1. Land use and watershed areas around Little St. Germain Lake.

2.2 Fishery and Aquatic Habitat (Macrophytes)

2.2.1 Fish Data

Data for the Little St. Germain Lake fishery was provided by the WIDNR (Figure 2). When comparing 1992 to 1997 and 2007, the muskie population appears to have dropped in the lake. After 2000, however, the amount of fish stocked bi-yearly was reduced from approximately 2000 to 490.

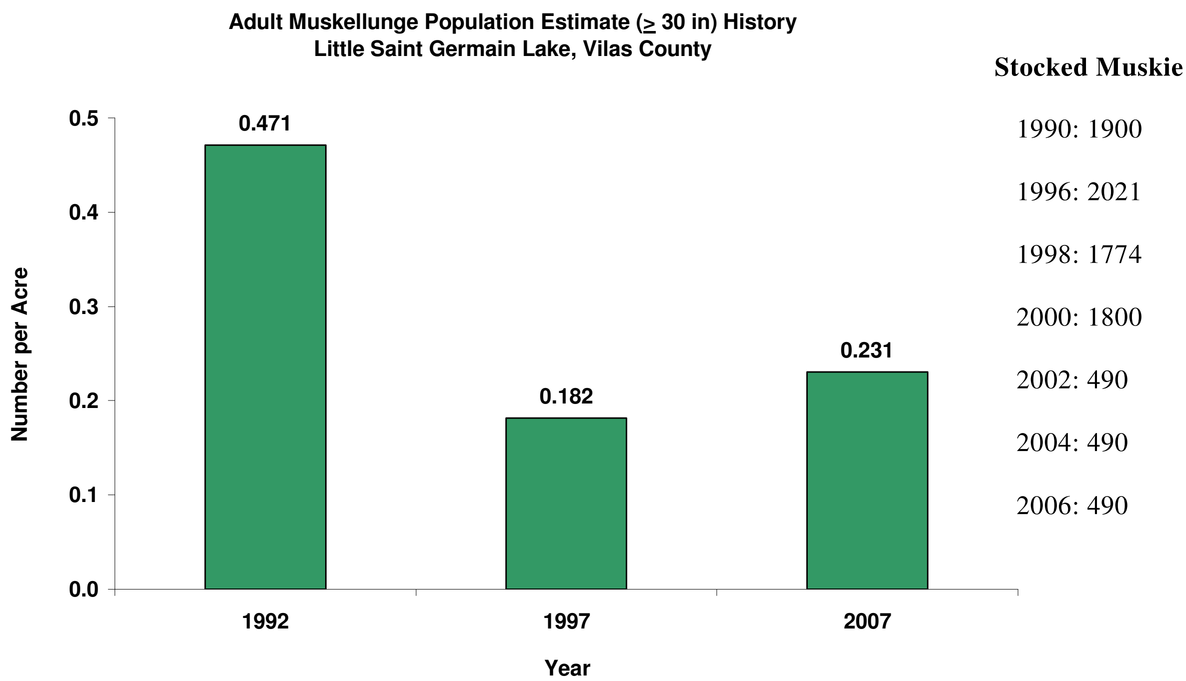


Figure 2. Muskellunge population and stocking data for Little St. Germain Lake.

2.2.2 Macrophyte Treatment Data

Little St. Germain Lake has populations of both Eurasian watermilfoil (EWM) and curlyleaf pondweed (CLP). Both of these plants can negatively affect the recreational and/or water quality of a lake. Both tend to dominate the aquatic plant community over time, lead to lower diversity within the system, interfere with recreation in shallower areas of a lake, and lower the overall aesthetics of a lake. CLP has an additional mode of impact whereby senescence of the plant in late June to early July can cause a release of phosphorus from plant tissue and also decrease oxygen in the water column during plant breakdown, potentially initiating or accelerating sediment phosphorus release.

Both CLP and EWM have been treated with herbicides beginning in 2003 (Table 1). Treatment of CLP occurs in the spring using Endothall while granular 2,4 D is used in both the spring and fall to manage the growth of EWM. All macrophyte treatment maps are shown in Appendix B. 2008 CLP treatment areas are mostly clustered within Lower East Bay with two areas immediately to the west of Lower East Bay. Only two small areas were treated for EWM in the Lower East Bay (0.5 acres total) and one larger area (3.8 acres) at the mouth of Muskellunge Creek, was treated in the East Bay.

Table 1. Herbicide treatment details for CLP and EWM in Little St. Germain Lake.

Treatment Date	Species	Chemical	Treated Area (Acres)
05/14/03	CLP	Liquid Endothall	42.7
07/01/03	EWM	Granular 2,4-D	3.0
08/04/03	EWM	Granular 2,4-D	9.0
05/11/04	CLP	Liquid Endothall	44.0
07/01/04	EWM	Granular 2,4-D	13.0
08/24/04	EWM	Granular 2,4-D	33.0
05/09/05	CLP	Liquid Endothall	50.0
07/13/05	EWM	Granular 2,4-D	8.5
05/12/06	EWM	Granular 2,4-D	6.2
05/13/06	CLP	Liquid Endothall	21.3
05/10/07	EWM	Granular 2,4-D	21.5
05/11/07	CLP	Liquid Endothall	41.6
05/27/07	CLP	Liquid Endothall	4.7
06/06/08	CLP	Liquid Endothall	54.4
06/07/08	EWM	Granular 2,4-D	24.0

CLP has been treated during years 2006 through 2008. In each successive year, larger areas of the lake needed treatment to manage CLP, indicating colonization of the lakebed may be increasing.

2.3 Nutrient Related Water Quality

2.3.1 Historical Trends

Historical, nutrient related water quality data for Little St. Germain Lake are available for the Lower East and East Bays for a 7 and 16 year period, respectively. Total phosphorus, Secchi disc depth, and chlorophyll *a* growing season averages are shown in Figure 3 and were calculated using data from May through August for years with at least four data points that were representative of the entire season. The growing season averages typically included at least one measurement per month.

Annual average total phosphorus appears to have increased during the period of record, especially in the Lower East Bay. However, a drop in total phosphorus in 2004 (the final year for both data sets) was seen in East Bay and it was not substantially different from data collected during the early 1990s. Chlorophyll *a* reached a low during 2000 but more than tripled by 2003 in both bays and remained higher than all growing season means during the 1990s. Secchi disc depth was lower when comparing recent data from 2001 through 2006 to data collected in the late 1990s.

Historical averages for all data collected from West, East, South, and Lower East Bays were calculated to compare water quality spatially across the lake (Figure 4). Both East Bay and Lower East Bay were highest in total phosphorus and chlorophyll *a* for surface samples and had the lowest average Secchi disc depth of the four bays. This may appear counterintuitive given that internal loading can be observed for both the West and South Bays during the summer months. However, both of these bays are highly stratified and bottom phosphorus does not regularly reach the lake surface.

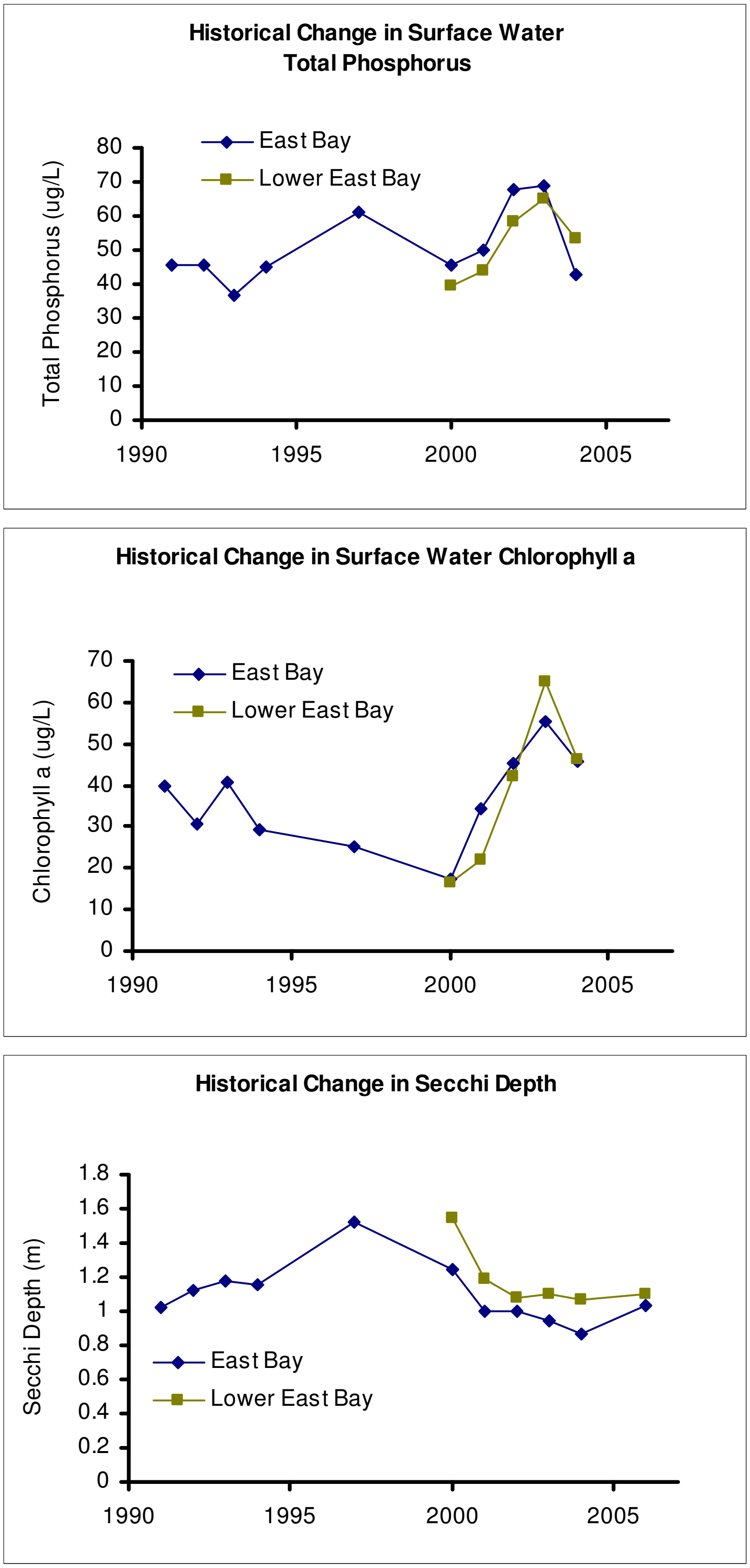


Figure 3. Historical changes in water quality at East Bay and Lower East Bay sites.

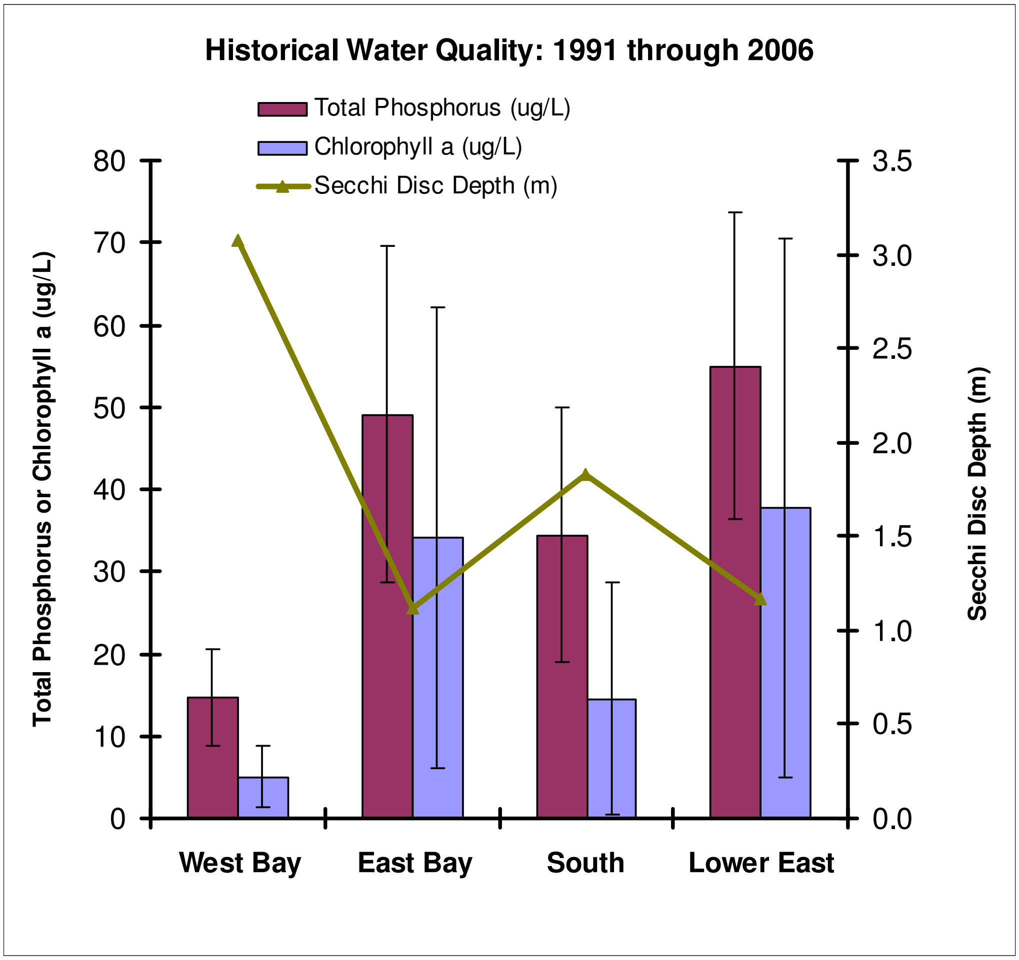
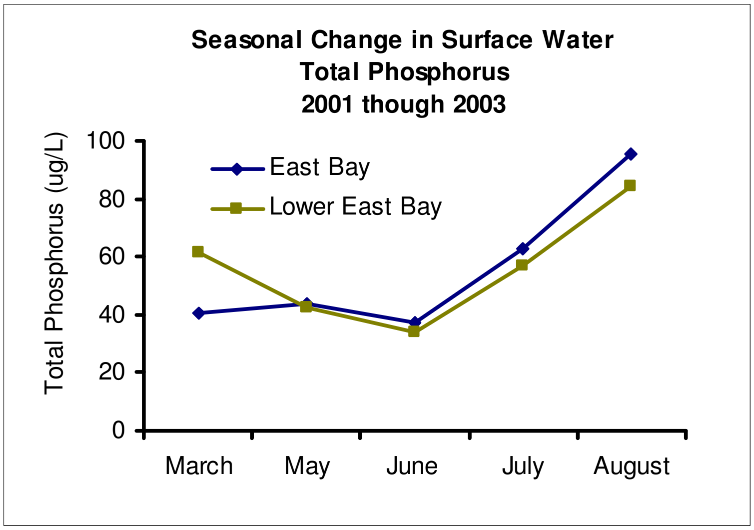
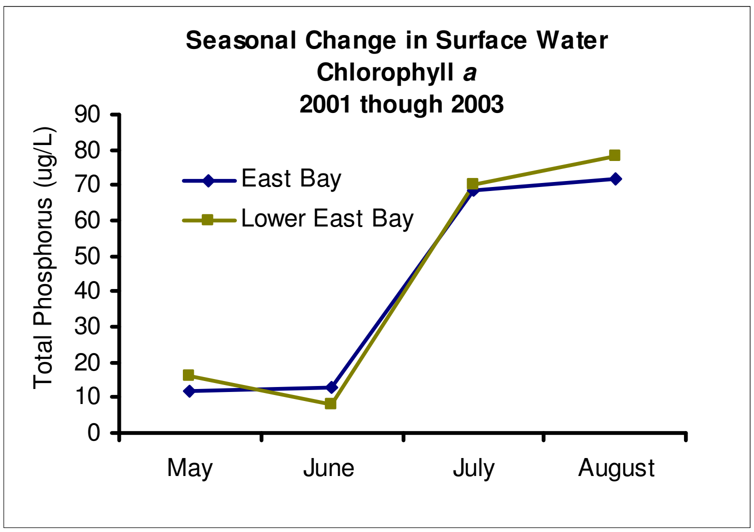


Figure 4. Average historical water quality at all monitoring locations in Little St. Germain Lake.

2.3.2 Seasonal Trends

To detect changes within a growing season, monthly averages for data from 2001 through 2003 were calculated and are shown in Figure 5. As the summer progresses, both total phosphorus and chlorophyll *a* increase, causing a corresponding decrease in water clarity. This trend is typical in lakes that are influenced by internal loading.





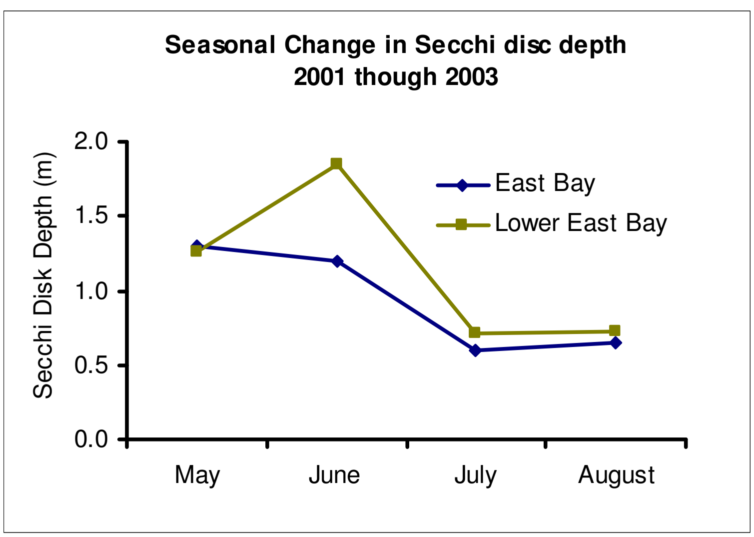


Figure 5. Seasonal changes in water quality in East Bay and Lower East Bay.

Evidence of internal phosphorus loading was seen during sampling in 2007 (Figure 6). Phosphorus concentrations reached over 0.4 mg/L near the bottom of East Bay but then decreased to less than 0.1 mg/L by the next sampling event, likely due to a mixing event. Concurrent with the decrease in bottom phosphorus was an increase in surface phosphorus in East Bay. Total phosphorus increased from approximately 0.05 mg/L in July to 0.087 mg/L by late August. Lower East Bay showed a similar trend although bottom water phosphorus data were not available. From these data it appears that phosphorus from the bottom waters was transported to the surface, decreasing water quality.

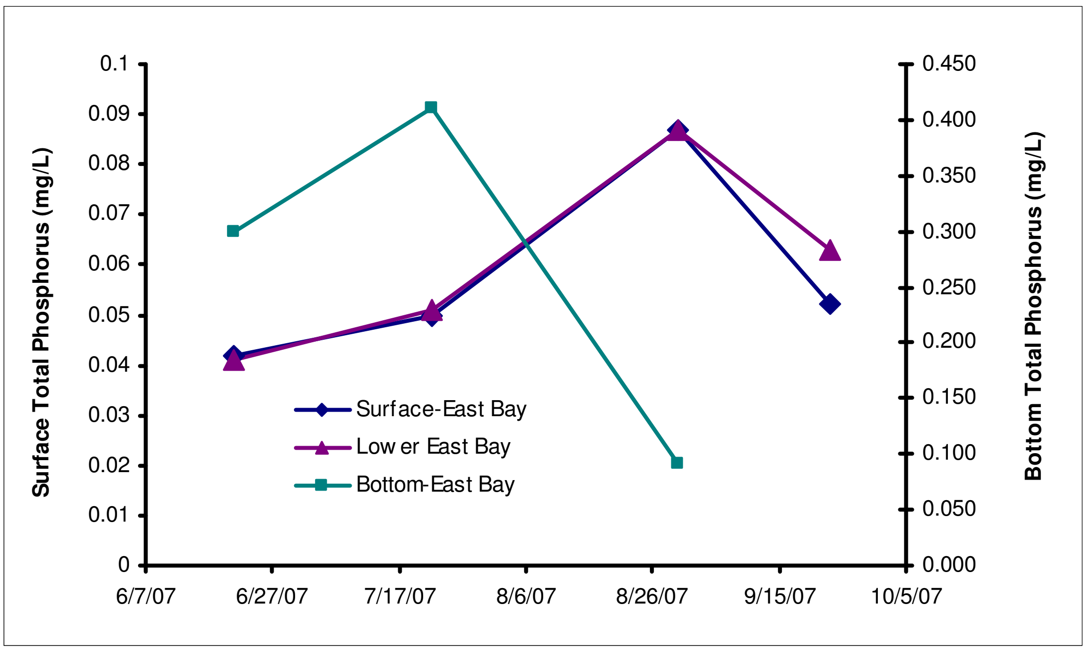


Figure 6. 2007 water column total phosphorus data for East Bay and Lower East Bay.

Data recently collected from Lower East Bay indicate that internal phosphorus loading appears to occur in areas of the lake that are generally thought of as mixed (Figure 7). Even though loading rates are likely lower in these areas when compared to the South and West Bays (based on the collected data), the water volume of these bays is smaller (less dilution) and the phosphorus is readily available for algal growth during the summer months. This is due to breakdown of stratification in these areas whereas loading in deeper areas of the lake may not impact the surface water significantly until turnover occurs in late fall.

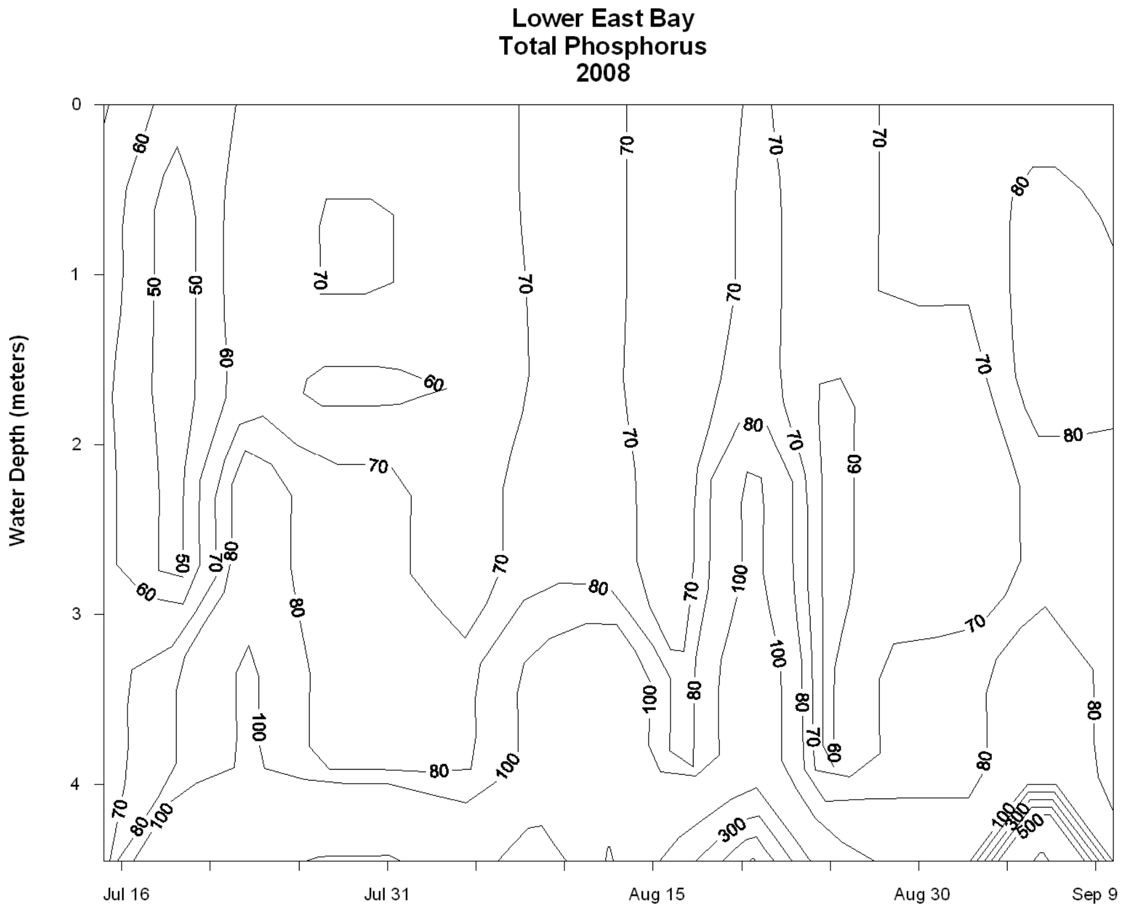


Figure 7. 2008 Lower East Bay water column total phosphorus isopleths data.

2.3.3 Overall Trends in Water Quality

Based on historical data, there appears to be a decline in water quality over the last two decades in Little St. Germain Lake. Overall the lake is eutrophic and water quality declines as the growing season moves into late summer. Water quality is poorest in East Bay and Lower East Bay but South Bay, strongly influenced by East Bay and Lower East Bay water quality, is also in the eutrophic range. Data indicate that internal phosphorus generation and mixing lead to higher surface water concentrations of total phosphorus in East Bay and Lower East Bay.

3.0 Management Options

Using all available data for Little St. Germain Lake, two options were investigated to reduce phosphorus levels in the lake. These options included external or internal phosphorus source loading reductions.

3.1 Inflow Alum Treatment

The feasibility of constructing and operating an alum treatment facility designed to remove phosphorus from Muskellunge Creek was evaluated (Barr, March 2007). The feasibility analysis was performed assuming that the facility would be constructed north east of the intersection of Birchwood Drive and Muskellunge Creek Road. The criteria for evaluation included capital and operation costs, physical constraints of the site, the capacity of the site to accommodate required treatment facility structures, and the expected in-lake phosphorus levels (East Bay) with a range of potential treatment facility designs and operating conditions. The findings of this study were as follows:

- Proper operation, performance, and cost effectiveness of the treatment facility will be constrained by the limited size of the site that is available for the construction of the facility and the large percentage of total flow volume in Muskellunge Creek that will need to be treated.
- A total of twelve alternative plant operating conditions were evaluated. The conditions evaluated include treatment of 50% (flows <6.6 cfs), 75% (flows <11.1 cfs), and 100% (flows <21 cfs) of Muskellunge Creek flows, alum doses of 3 and 6 mg/L as aluminum, and the use of baffle or mechanical mixing of alum and water.
- The cost of capital, engineering and design, and treatment system optimization is expected to range from $0.7 to $1.1 million if a mechanical mixer is used. Greater treatment performance is expected with the mechanical mixing system.
- Annual operation and maintenance costs were expected to range from $130,000 to $600,000, depending on the volume of stream flows treated and whether alum doses of 3 or 6 mg/L are used. (Note, this cost would be higher now due to higher prices for aluminum)

- Because the available treatment site is constrained by its size, treatment of stream flows less than 6.6 cfs was recommended.

- There are several physical and chemical constraints that may affect system performance or will require some operational adjustments. The use of a 6 mg/L dose may be constrained by the low alkalinity of Muskellunge Creek (expected to range from 35 to 60 mg/L as $CaCO_3$) and the potential to suppress the pH of water in the creek below 6.0. Hence, the lack of alkalinity in Muskellunge Creek may restrict the treatment system performance (because a lower dose will need to be used) or an alternative, higher cost coagulant (e.g., polyaluminum chloride) will need to be considered.

- Using a calibrated water quality model for the East Bay of Little St. Germain Lake and 2001 monitoring data collected by the USGS, it is estimated that average treatment season (mid-April through September) phosphorus levels would decline from 0.051 mg/L to somewhere within the range of 0.038 to 0.041 mg/L with the treatment of stream flows less than 6.6 cfs (Figure 8).

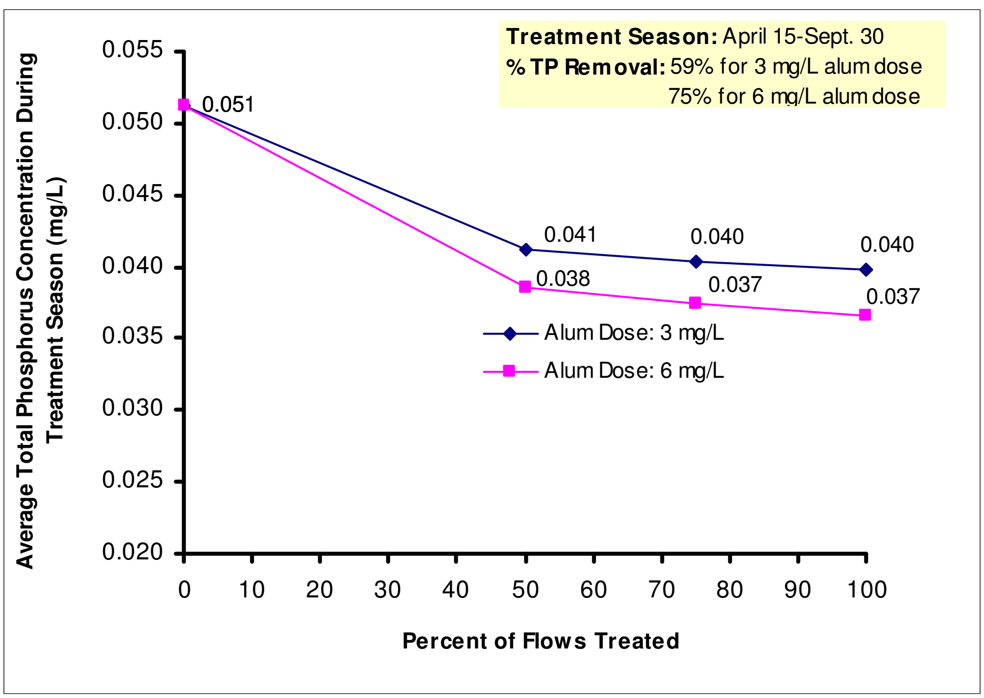


Figure 8. Expected phosphorus reductions from Muskellunge Creek using alum.

- The use of the calibrated lake model and the sediment studies conducted by the USGS and Barr indicate that phosphorus release from the sediments (internal loading) of the East Bay of Little St. Germain has a significant effect on phosphorus levels in the East Bay. If internal phosphorus loading were reduced by 90%, the average phosphorus level in the East Bay (mid-April through September) would have been 0.036 mg/L in 2001 (Figure 9).

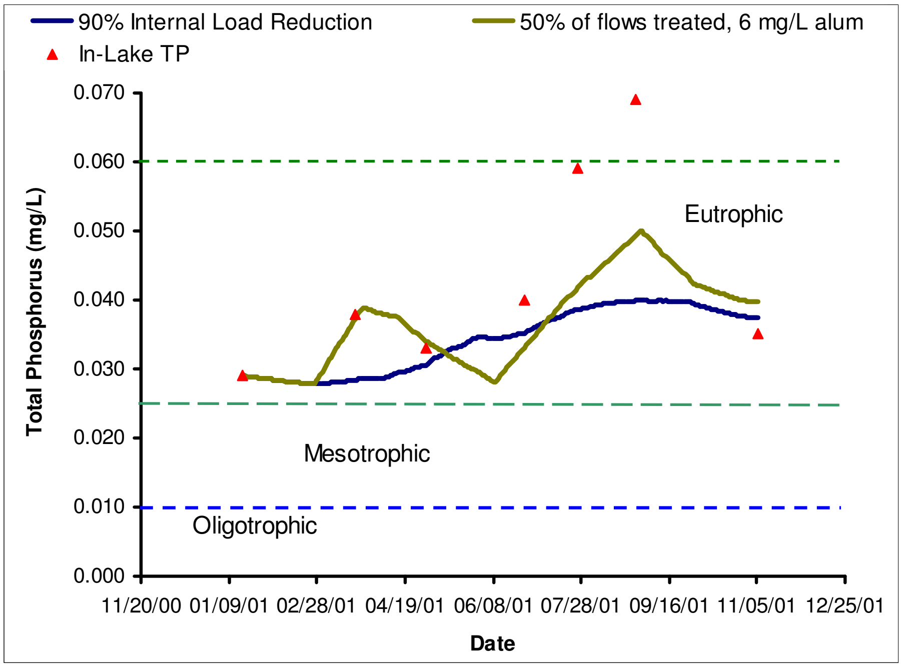


Figure 9. A comparison of total phosphorus in East Bay after inflow treatment and whole lake treatment using alum.

After assessing the in-lake and inflow data, the expected cost for a treatment facility, and the expected benefit to in-lake phosphorus levels, it was decided to further investigate the effect of internal phosphorus loading and the feasibility of in-lake alum treatment to improve water quality in Little St. Germain Lake.

3.2 In-Lake Alum Treatment

To better quantify internal phosphorus loading, and expected costs and potential benefits of treating the lake sediments with alum in Little St. Germain Lake, sediment cores were collected at 26 locations in the lake in June 2007. Sediment was analyzed to determine the spatial distribution of phosphorus (mobile, aluminum-bound, and organic) for different regions of the lake and corresponding potential phosphorus release rates and appropriate alum doses were estimated,. The distribution of phosphorus (mobile fraction) is shown in Figure 10. Overall, the concentration of sediment phosphorus in the lake was high, even when compared to lakes in urban areas, and there is a high potential for internal phosphorus loading to affect water quality in the lake. The sediment data indicate that the highest phosphorus was in West Bay, followed by South Bay, Lower East Bay, East Bay, and then No Fish Bay.

Although there is a potential for internal phosphorus loading to affect phosphorus levels in the water column of each bay, factors such and dissolved oxygen levels, bathymetry, the volume of each bay, stratification, and the rate of transport of phosphorus from the lake bottom, determine whether high sediment phosphorus actually results in high phosphorus in the surface water (and hence high algal growth).

Water quality models were developed to evaluate the expected change in phosphorus in East Bay, Lower East Bay, and South Bay with alum treatment. A model was not developed for West Bay because water monitoring data collected in 2007 indicate that the potential for phosphorus transport to the surface of this bay to be minimal, and phosphorus levels are relatively low in the summer.

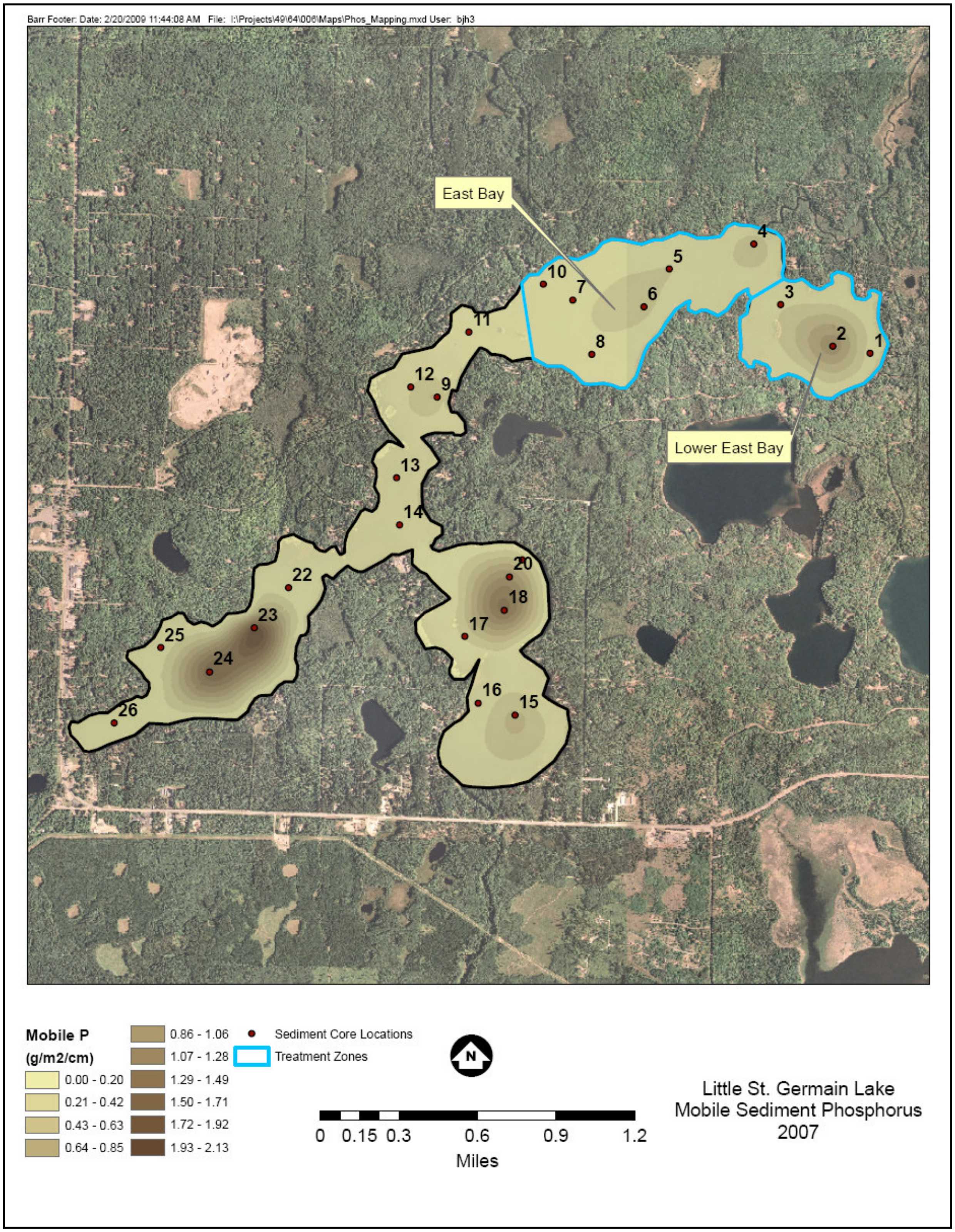


Figure 10. Sediment mobile phosphorus in Little St. Germain Lake.

The results of the modeling work, provided in Table 2, indicate that there will be a significant water clarity benefit to East Bay and Lower East Bay as well as South Bay (water from East Bay affects South Bay phosphorus levels) if East Bay and Lower East Bay are treated with alum at the doses prescribed (see Section 5). The additional benefit of treating South Bay, in addition to East Bay and Lower East Bay, should be weighed against the additional cost of treating South Bay. The primary benefit of treating South Bay with alum will be a reduction in spring to early summer algal blooms and reduced potential and magnitude for late summer blooms.

This study recommended treatment of East Bay and Lower East Bay because it was expected that the water quality improvement in South Bay would be adequate with only the treatment of the upper two bays.

Table 2. Expected Improvement in Total Phosphorus, Chlorophyll a, and Secchi Disc Depth (June through August) with Alum Treatment.

Alum Treated Area	East Bay/Lower East Bay			South Bay[2]		
	Total Phosphorus (ug/L)	Chlorophyll *a* (ug/L)	Secchi disc depth (ft/m)	Total Phosphorus (ug/L)	Chlorophyll *a* (ug/L)	Secchi disc depth (ft)
No Treatment	62	38	2.8/0.9	46	19	4.3/1.3
East/Lower East Bay Only: **Proposed Option**[1]	33	15	4.2/1.3	28	10	6.1/1.9
South Bay Only	62	38	2.8/0.9	35	13	5.2/1.6
South Bay and East Lower East	33	15	4.2/1.3	19	6	7.6/2.3

(1)Average of modeling results for 2001, 2002, and 2007. Average for June through August period.

(2)Average of modeling results for the year 2002.

4.0 In-Lake Alum Treatment: Dosing, Benefits and Potential for Adverse Effects

Alum has been used to reduce phosphorus in lakes for approximately four decades (Landner 1970, Kennedy et al. 1987, Welch and Cooke 1998, Reitzel et al. 2003, Huser et al. 2009). Numerous studies have been conducted on both the benefits and potential drawbacks to using alum for phosphorus reduction in surface water.

4.1 Alum Dosing

Historically, dosing of alum was based on lake water alkalinity in order to avoid toxicity effects from aluminum (Kennedy and Cooke 1982). The alkalinity-based method was used as a “rule of thumb” because aluminum acts as a weak acid and reduces lake water pH once the available alkalinity is consumed. The alkalinity-based method, however, lacks any relationship to phosphorus control, the issue of concern. In most cases this dosing method restricted the amount of alum that was applied, leading to an under-dosing of the lake. This was especially true for shallow and/or soft water lakes where there is less buffering capacity.

More recently, sediment phosphorus release rates along with expected treatment longevity had been used in alum dose calculations (Kennedy et al. 1987). This is an improvement because it focuses on the phosphorus released from sediments that may eventually become available for algal uptake. Use of internal phorphorus release rates alone, however, may overestimate, or more likely, underestimate the amount of aluminum needed to inactive phorsphorus available for release from the sediment. Another drawback to this method is that the entire mobile phosphorus pool may not be released during the experimental time frame. That is, the internal P release measured will yield a rate of phosphorus released from the sediment, but not necessarily the total mass of P available for release over the long term.

Another reason under-dosing may have occurred with the internal loading method is due to the fact that sediments previously treated with alum have shown that the phorphorus sorption capacity of aluminum is significantly lower than the previously assumed stoichiometric ratio of 1 (Kennedy and Cooke 1982; Kennedy 1987). In Washington and Wisconsin alum-treated lake sediments, the aluminum to phosphorus ratios (Al:Al-P) ranged from 5:1 to 11:1 (by weight), respectively (Rydin and Welch 1999; Rydin et al. 2000). The ratio ranged from 4.4 to 12 in alum-treated lakes in

Minnesota (Huser 2009). Basically this means that lakes treated with alum under the assumption of a 1 to 1 binding ratio for aluminum and phosphorus were under-dosed by up to a factor of 12.

An advanced method developed by Rydin and Welch (1999) focuses on the conversion of sediment mobile phosphorus to aluminum bound phosphorus by adding alum directly to sediments in laboratory experiments. For complete removal of mobile phosphorus, an aluminum to aluminum bound phosphorus ratio of 100:1 was recommended with a treatment depth of four centimeters. Although both were mentioned in the study, this method did not account for the variability of mobile phosphorus by sediment depth or organic bound phosphorus release.

Another important factor for dosing alum to control internal phosphorus loading is the sediment depth distribution of mobile phosphorus. This depth has varied in different studies; Rydin and Welch (1998) found excess mobile P in the top 4 cm of sediment, James and Barko (2003) reported elevated levels in the top 30 cm, and Pilgrim et al. (2007) found a range of 4 to 7 cm. According to Pilgrim et al., the mobile phosphorus method could be improved by relating mobile phosphorus in the sediment with expected internal release rate of P after treatment. Thus a net reduction of P released to the water column from conversion of mobile phosphorus to aluminum bound phosphorus could be calculated. A method to incorporate expected breakdown of organic phosphorus in the dosing method was also included by Pilgrim et al. (2007).

The method used in this study to calculate alum doses for Little St. Germain Lake is a combination of the Rydin and Welch and Pilgrim et al. methods. Dosing is based on the reduction of mobile and organic phosphorus in the sediment and the final dose is then calculated based on the desired internal phosphorus loading rate. The alum dosing results for Little St. Germain Lake are explained in more detail in Section 5.

4.2 Case Studies

As noted above, the best method to determine the amount of aluminum needed to control internal phosphorus loading in lakes is based on mobile phosphorus levels in sediment. However, this method has only recently been developed and therefore, nearly all lakes with long term data sets were dosed based on alkalinity (buffering capacity) or internal phosphorus loading rates.

Deep lakes have shown better results when compared to shallow lakes in cases where aluminum has been added to control internal phosphorus loading. Because shallow lakes have low buffering capacity (primarily because of lower water volume) and internal phosphorus loading rates can be difficult to estimate, these lakes generally receive much lower alum doses compared to deep lakes.

Longevity can also be limited in shallow systems because the same mass of internal load generally has a greater impact in a shallow lake. Thus, dosing for shallow lakes is particularly important because even a small residual internal phosphorus loading rate after treatment can limit effectiveness.

4.2.1 Deep (Dimictic) Lakes

In a study conducted by Welch and Cooke (1998), water quality data collected from 12 alum-treated, dimictic lakes was analyzed before and after alum application to determine the longevity of treatment. Total phosphorus (surface), chlorophyll *a*, and internal loading were reduced and control lasted from 4 to 20 years. Internal loading reduction averaged 80% of pre-treatment levels during these years in lakes where monitoring was conducted.

Little published data exist for lakes where alum was dosed based on sediment mobile phosphorus. However, Huser et al. (2009) showed that in Lake Calhoun (Minneapolis, MN), a sharp reduction in surface total phosphorus occurred during the first four years after treatment (2002-2005). Secchi depth increased from approximately 10 to 20 feet and internal loading was reduced by over 90%. Two of the lakes in this study were not dosed using sediment phosphorus content (Cedar Lake and Lake of the Isles). Cedar Lake has shown positive treatment effectiveness through the 10 years of data that the study summarized. Although this lake has two relatively deep bays (up to 40 feet), it is polymictic during some years.

Other alum treated lakes described in Cooke et al. (2005) and Huser (1999) include Mirror and Shadow Lakes (WI, 13 year treatment effectiveness), Lake Morey (VT, at least 12 year treatment effectiveness), West Twin Lake (OH, 18 year treatment effectiveness), and Medical Lake (WA, ~20 year treatment effectiveness).

4.2.2 Shallow (Polymictic) Lakes

There are no peer-reviewed water quality studies for shallow lakes in which alum was dosed and applied based on mobile phosphorus. All cases listed below were dosed on alkalinity or, at best, using an estimate of internal phosphorus loading rates.

Surface total phosphorus and chlorophyll *a* were reduced by an average of 40% from 5 to 11 years after treatment in six out of nine alum treated, polymictic lakes (Welch and Cooke 1998). Three of the polymictic lakes in this study showed little to no improvement after treatment. It was hypothesized that macrophytes caused either uneven floc distribution or in two cases, where effectiveness was not as long as expected, large amounts of phosphorus were transported through plant tissue via senescence and decay. However, a number of the other lakes in the study with positive results also had substantial macrophyte coverage. It is more likely that these lakes, because they were shallow and most had low alkalinity, were under-dosed with respect to phosphorus in the sediment. Because of the low buffering capacity in shallow lakes (due to low water volume), most shallow lakes have been severely under-dosed using older, non-sediment based alum dosing methods.

Huser et al. (2009) showed that treatment effectiveness lasted approximately five years in Lake of the Isles (Minneapolis, MN), a shallow lake. Again, however, alum dosing was not based on sediment mobile phosphorus and sediment data showed that a large pool of mobile phosphorus remained in the sediment after alum treatment (Huser 2009).

4.3 Potential Toxicity and pH Effects

The toxicology of aluminum has been studied extensively. Most of the work on aluminum toxicity has centered on the effect of acid rain, low pH, and the subsequent increase in aluminum toxicity due to acidification. A broad summary of the toxicity literature by Pilgrim and Brezonik (2005) suggested that the potential for aluminum toxicity to invertebrates, zooplakton, and fish is negligible if pH is maintained above 6.0 but not significantly higher than 8.5. This is largely due to the fact that within this pH range aluminum is bound to hydroxide as $Al(OH)_3$ and the reactivity of aluminum is reduced.

Generally, studies have shown short-term, initial impacts from alum treatment on phytoplankton, zooplankton and benthic species. These effects are mainly due to physical properties of the aluminum floc once it has entered the water. However, due to the increase in water quality following treatment; abundance and diversity generally increase.

Lakes Harriet and Calhoun (Minneapolis, MN) were treated with alum in 2001 and provide an example of in-lake aluminum concentrations after treatment. Lake Harriet, treated in early May 2001, had elevated total aluminum concentrations in the surface water shortly after treatment up to 0.298 $mg \cdot L^{-1}$ (Figure 11). By June, surface concentration was back near the pre-treatment condition. Lake Calhoun was treated in the autumn of 2001, but no aluminum sampling was conducted immediately after treatment. By the early spring of 2002, however, aluminum concentrations were low and comparable to pre-treatment levels, even in the water just above the alum treated sediment. By June of 2003, both lakes had total aluminum concentrations slightly below pre-treatment levels throughout the water column. It was hypothesized that organic matter, which can act as a carrier of aluminum, was lower (i.e., less algae) and hence less aluminum was held in the water column.

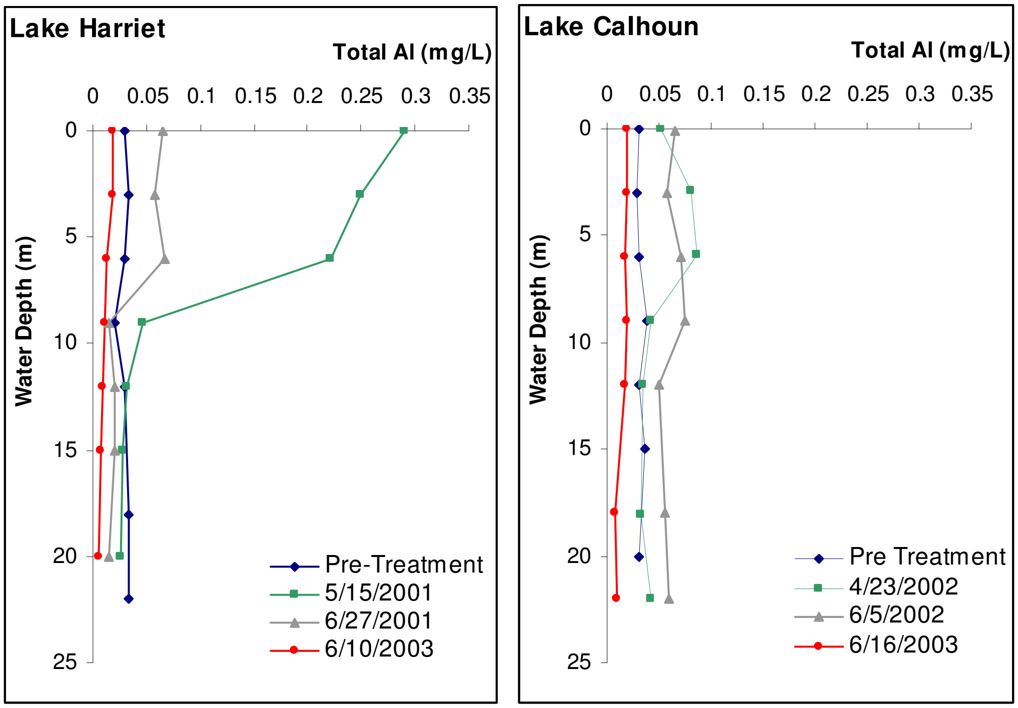


Figure 11. Water column concentrations of aluminum in two lakes before and after alum treatment.

4.3.1 Fish

When aluminum is highly soluble (at low pH) it can interfere with fish respiratory function. Therefore, as long as lake pH remains above 6 (aluminum solubility is minimum near pH 6.3) during an alum treatment, fish mortality is not expected to occur. Monitoring of two alum treatments in Minneapolis in 2001, conducted directly behind the application barge, showed the operator was able to maintain pH levels well above 6 and no fish mortality was observed directly after treatment or in the following days (Mike Perneil, Mpls. Park and Rec. Board, personal communication). Zero mortality of trout between pH 6 and 10 was seen in an experiment using alum-treated sludge from a wastewater facility (Ramamoorthy 1988). This was expected as aluminum is in the particulate phase within this pH range and in the presence of solids.

Bioaccumulation of aluminum in rainbow trout after alum treatment was examined in Medical Lake, Washington (Buergel and Soltero 1983). 12.2 mg Al/L (~120 g Al/m^2) was added during treatment and no physiological stress, gill hyperplasia, or retardation of growth was detected. Aluminum in gill tissues in a nearby hatchery (not affected by alum treatment) was higher than that detected in Medical Lake.

In some cases exposure to continuously elevated levels of aluminum hydroxide can produce long-term, chronic effects in fish. Kane and Rabeni (1987) showed that smallmouth bass had significantly reduced activity after a continuous exposure to $Al(OH)_3$ over 30 days. However, during an alum treatment, the $Al(OH)_3$ floc settles out of the water column (usually within hours) and continued exposure would not be expected.

Fisheries data from the past 20 years were obtained from the MN DNR for four of the lakes in the Minneapolis Chain of Lakes (the Chain): Lake Calhoun, Lake Harriet, Lake of the Isles, and Cedar Lake (Table 1). Data were also obtained for Lake Nokomis; a nearby, non-alum treated lake within the same system. All four of the lakes in the Chain have been treated with alum at different times in the past:

- Lake of the Isles (1996)
- Cedar Lake (1996-97)
- Lake Harriet (2001)
- Lake Calhoun (2001)

Lake Nokomis has not been treated with alum, but did have a common carp removal as part of its management. Approximately 5,000 lbs of carp were removed from the lake in January 2001. Harvesting of macrophytes (mainly Eurasian watermilfoil and curlyleaf pondweed) occurs in all five lakes.

Data for the predatory fish species, northern pike, muskie, and tiger (hybrid) muskie, were compared for each lake. Data is in pounds of fish per set (i.e. gill net) and each survey event employed anywhere from 2 to 9 gill nets. This surveying method is more useful for a qualitative analysis when looking at large predatory fish because the number of large fish caught is typically small and does not generally provide statistically meaningful data. Northern pike and muskie/tiger muskie were observed in all lakes before and after alum treatments. Based on available data there was no evidence of population changes due to alum treatment in any of the lakes (see Appendix A).

It should be noted the Chain is nicknamed such because the lakes are connected via shallow channels/waterways that allow fish to migrate from one lake to another. Fish that were stocked in one lake were captured in another lake on several occasions. Also, field crews observed dead muskies or tiger muskies on various lakes on several occasions during the fish surveys. When examined closely, most of these dead fish showed evidence of severe injury from recreational fishing.

4.3.2 Benthic Invertebrates

Bottom dwelling, benthic organisms have not been affected greatly by previous lake alum treatments except that diversity increases in some cases (Conner et al. 1989; Narf 1989). The increase in diversity was attributed to more oxygen in the deep water due to lower settling of algal matter and increased quality of habitat. In Liberty Lake, WA, alum treatment appeared to negate toxic effects of a previous rotenone treatment, as crayfish, absent previous to alum treatment, were found 5 days after treatment (Funk et al. 1982).

It can be concluded from the peer-reviewed literature regarding whole-lake alum treatments that the potential for adverse effects with alum treatment only occurs when there is a significant accumulation of alum floc (i.e., a physical disturbance of habitat). The literature also indicates that alum is not toxic to benthic invertebrates when pH is above 6.0, while notable toxicity is apparent near pH 5.0 (Gensemer and Playle, 1999; Pilgrim and Brezonik, 2005). Lamb and Bailey, 1981, studied the effect of alum floc on benthic invertebrates in a laboratory setting to determine if proposed whole lake alum treatments would have adverse effects on benthic invertebrates. They reported significant mortality of a chironomid (aquatic insect) in chronic laboratory tests only with very high alum doses of 80 to 480 milligrams per liter. The authors suggested that observed mortality was potentially due to aluminum toxicity but they also noted that there was significant alum floc accumulation and the chironomids were using the floc as habitat by burrowing within the floc. It was noted that heavy aluminum floc was likely causing stress and mortality for tests with higher alum doses. This study also demonstrated that there were minimal chronic effects with an alum dose of 10 milligrams per liter (as Al) and that there were no acute effects even at doses as high as 960 milligrams per liter.

As part of a district-wide evaluation of wetland health, the Ramsey Washington Metro Watershed District (Ramsey County, Minnesota) performed benthic invertebrate and water quality monitoring of a wetland (T-31) that is downstream of an in-line alum treatment facility. The alum treatment facility has been operating from spring though the fall of each year since 1998. During that time the concentration of aluminum entering the wetland has averaged from 1 to 6 milligrams per liter of aluminum. Aluminum was enriched in the wetland sediment, indicating that aluminum that entered the wetland also deposited as alum floc in the wetland. Benthic invertebrate monitoring results and IBI (index of biotic integrity) analysis for the T-31 wetland revealed that the wetland had similar quality to the high quality reference (unimpacted) wetlands in the study. The conclusion of this study was that the deposition of aluminum floc in this wetland had not adversely affected the biota of the wetland. Based upon the alum dosing suggested for Little St. Germain Lake (4.7 mg/L of aluminum), levels are comparable for the T-31 wetland and alum application to Little St. Germain Lake.

A study was conducted on a wetland/settling pond and a lake downstream of an in-line treatment system (continuous alum dosing) in Eagan, Minnesota (Twin Cities Metropolitan Area). This study found no effects on the benthic invertebrate community downstream of the alum treatment system with the accumulation of 10 cm of alum floc (Pilgrim and Brezonik, 2005). However, the benthic invertebrate community was nearly eliminated with over 1 foot (approximately 35 cm) of alum floc accumulation. The alum floc had an aluminum content of 200 milligrams per gram of dry sediment, roughly 10 times the background concentration. The authors hypothesized that the loss of the invertebrate community was largely due to the physical disruption of suitable habitat and the creation of severe anaerobic conditions below the floc layer. The potential for significant alum floc accumulation to disrupt benthic invertebrate communities also has been documented for an experimental treatment system on the Cuyahoga River (Barbiero et al., 1988).

Finally, a study on the Lake Morey (VT) alum treatment did show an initial decrease in the condition factor for yellow perch and benthic invertebrate habitat but the decline was temporary and there was a long term benefit to biotic populations (Smeltzer et al. 1999).

Overall it appears that under certain conditions and treatment circumstances (e.g., pH below 6.0 and very heavy alum floc accumulation-approximately one foot or greater), negative impacts to the benthic community can occur with alum treatment. However, if pH is carefully controlled and floc is not excessively deep (floc accumulation will not be more than 1-2 cm with this treatment), adverse effects are not likely to occur. Over the long-term, benthic species seem to benefit as abundance and diversity are generally greater after treatment.

4.3.3 Potential Combined Effects Between Endothall, 2,4-D, and Alum

There were no data or published reports on the combined effects of using the herbicides Endothall and 2,4-D and alum. However, because changes in the local environment can stress living species in general, it is recommended that alum treatment occur at least two weeks after any late summer or early fall herbicide treatment.

One potential indirect effect of combining herbicide and alum treatments in the same area could be dissolved oxygen depletion. As macrophytes die back due to herbicide application, decomposition of the plant material decreases oxygen in the water column. Alum (it forms aluminum hydroxide once applied to the lake water) causes settling of particulate matter from the water column to the sediment as it flocculates and settles. If algal biomass is high in the water column at the time of treatment, alum treatment will cause the algal matter to settle out of the water column. Degradation of the settled algal matter, along with decomposing macrophyte matter, may cause areas of dissolved oxygen depletion, potentially stressing fish and benthic invertebrates. Although this process has not been documented (at least in peer-reviewed literature), it is a potential risk if both herbicide application and alum treatment are conducted

4.4 Effects on Macrophytes

Submerged macrophytes in surface waters are dependent upon light and nutrients for growth. An increase in water clarity will increase the amount of sediment surface available to colonization by macrophytes. It is generally assumed that plant rooting depth in lakes is equal to the average summer Secchi disc depth. This also assumes that some other factor is not limiting the depth or area of macrophyte growth (e.g. fetch, fish, sediment substrate, etc.). Based on these assumptions and the morphology of each basin, the potential increase in lake surface area that could be colonized due to increased water clarity is shown in Table 3.

Table 3. Expected changes in lake surface area for macrophyte rooting depth after alum treatment.

	Average Secchi Depth (feet)	Current Macrophyte Rooting Area (acres)	Proposed Secchi Depth (feet)	Proposed Macrophyte Rooting Area (acres)	Potential Increase in Rooting Area (acres)
East Bay	7.6	65.3	9	80.5	15.2
Lower East Bay	7.6	70.3	9	91.2	20.9

The expected increase in water clarity may increase the potential rooting area for macrophytes by 15.2 acres in East Bay and 20.9 acres in Lower East Bay. However, as stated above, other factors may limit colonization including fetch and sediment type and composition.

Previous work has shown that the health of a macrophyte community improves with increasing water clarity. Diversity is generally higher in systems with higher water clarity and a greater number of high value species are usually present. Madsen et al. (2004) showed that as the light extinction coefficient increased (clarity decreased) in two northern Minnesota lakes, aquatic macrophyte diversity declined. Rybicki and Landwehr (2007) saw an increase in native species (with respect to invasive species) and plant community diversity increased when water quality increased between 1985 and 2001 in the Potomac Estuary. Macrophyte diversity and density increased after water clarity increased due to fish removal in a large, shallow wetland (Schrage and Downing 2004).

4.5 Aeration and Alum

Aeration of lake water is designed to increase the limit the impacts from oxygen by adding air or compressed oxygen to the water column. The main purpose of the aeration systems installed in Little St. Germain is to prevent low oxygen conditions from negatively impacting the fishery. The systems are run from approximately December until ice-out.

There are a number of case studies where alum and aeration were used to improve water quality. Medical Lake (WA state) showed improved water quality after alum application and aeration for at least 20 years (Huser 1999). Powderhorn Lake, a small stormwater pond with a deep hole, was treated with alum in 2003 and has an aeration system. No evidence of interference between the aerator and the aluminum floc has been seen in the lake and the water quality has improved. Both of these aeration systems were designed to increase dissolved oxygen to improve the fishery in each lake.

Because aeration systems are designed to increase oxygen in the water column, there should be minimal effect on the alum treatment in Little St. Germain Lake. Once applied, alum forms aluminum hydroxide, sinks down to the sediment surface, and mixes with the surficial sediment layers. There may be some floc re-distribution when the aeration systems are installed but the aluminum-phosphorus bond will not be affected.

5.0 Alum Treatment Plan

5.1 Problem Definition

Water quality in East Bay and Lower East Bay is poor, especially when compared with other areas of the lake. Water quality conditions worsen towards the middle of the summer and continue to decline until the fall. Phosphorus loading from the sediment is contributing to the poor water quality in these bays and thus, a reduction of internal loading is expected to increase water quality, especially in the mid to late summer months.

5.2 Goals and Objectives

The goal with this alum treatment is to reduce mid and late summer algal blooms and to have a more stable, consistent, and improved lake clarity from spring through fall. This goal can be achieved by reducing internal phosphorus loading by 90% in both East Bay and Lower East Bay.

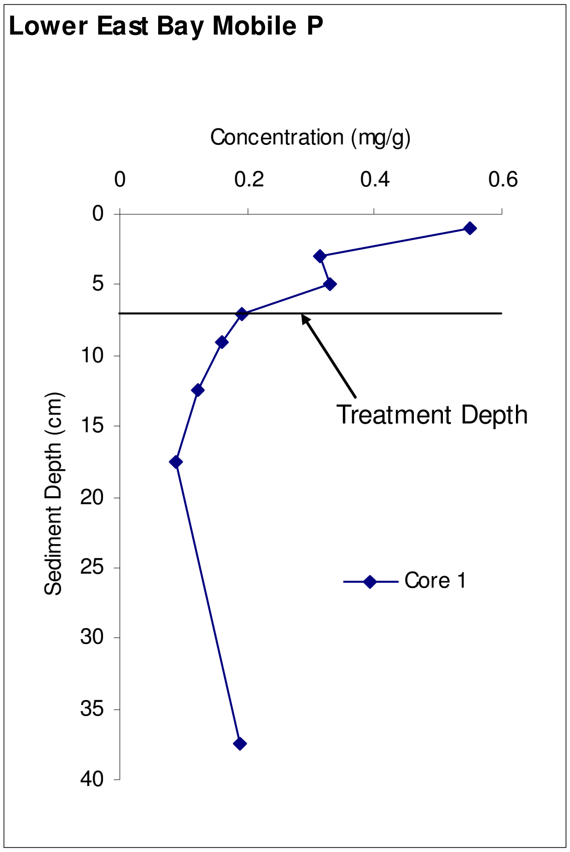
5.3 Treatment Plan

The treatment plan involves developing an alum dose to control internal phosphorus loading, timing of the application, and sequencing. A risk assessment for the treatment is also presented below.

5.3.1 Dosing and cost

The total alum dose is based on sediment content and depth distribution of mobile phosphorus in Little St. Germain Lake. The maximum amount that can be applied during treatment is limited by the alkalinity (or buffering capacity) of the lake water. Because aluminum is slightly acidic, dosing must be calculated so the lake water pH does not drop below pH 6.

For Little St. Germain Lake, alum dosing was based on using an aluminum to mobile phosphorus ratio of 75:1, an average sediment treatment depth of 6 cm, and a 90% reduction of internal phosphorus load. Treatment depth was determined by analyzing the depth distribution of excess phosphorus in the sediment of the lake (see example in Figure 12).



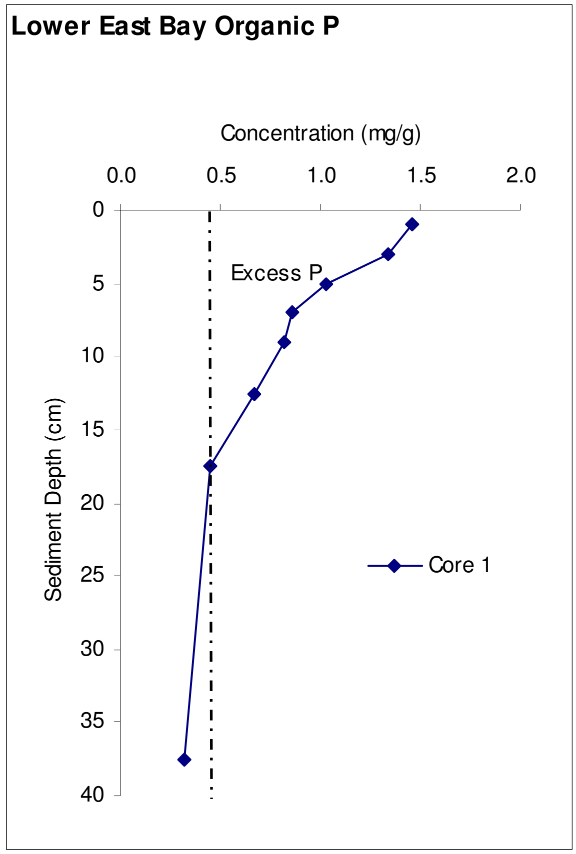


Figure 12. Examples of alum dose calculation parameters for mobile and organic phosphorus.

In addition, alum dosing incorporated organic-phosphorus degradation in the sediment. As organic phosphorus is broken down through microbial activity, it contributes directly to the mobile phosphorus fraction and, over time, labile organic matter will contribute to internal phosphorus loading. The amount of alum needed to bind this future source of mobile phosphorus was calculated by determining the amount of excess labile organic phosphorus and estimating the conversion to mobile phosphorus using annual degradation rates of 1.5 to 2% over a 10 year period.

Because the use of this alum dosing method generally results in higher alum doses over traditional methods, multiple phases for treatment are usually recommended for shallow lakes or lakes with low buffering capacity. This is done both to save cost by using the natural buffering capacity of the lake water and to improve phosphorus contact with aluminum through multiple smaller doses instead of one large dose. For Little St. Germain Lake, alum will need to be applied in annual phases (once per year) for two to three successive years.

Applying alum in phases provides flexibility and an adaptive management approach. After each phase of treatment, sediment and water monitoring are conducted to re-evaluate the need for further treatments. If monitoring results indicate that sediment treatment and water quality goals are being met before the entire alum dose is applied, the final phase of treatment could be reduced or postponed.

Based on the mobile phosphorus content of the sediments collected from East Bay and Lower East Bay in 2007, the total amount of alum that will be added for each bay is 740 gal/acre(East Bay) and 1620 gal/acre (Lower East Bay). The average alum dose across the entire area (East Bay and Lower East Bay) is 1048 gallons per acre (Table 4).

Table 4. Total alum doses required to convert mobile phosphorus to aluminum bound phosphorus in the East and Lower East Bays (based upon a total of three phases).

Treatment Zone	Alum Dose* (gal/acre)
East	740.0
Lower East	1620
Total	1048

*Based on total lake area

Due to the relatively low alkalinity (48 mg/L) in Little St. Germain surface waters, the amount of alum applied per treatment must be adjusted to prevent pH depression below 6. Based on water chemistry data and the water volume in each bay, eight treatments (without any buffer) would be needed to add enough aluminum to the sediment and not suppress pH below 6.0. Because this is not feasible, several options were considered so that more aluminum could be added without exceeding pH guidelines. The options included the use of a buffered solution of sodium aluminate, use of polyaluminum chloride, or the addition of lime in conjunction with alum treatment. Treatment costs for sodium aluminate and polyaluminum chloride were at least 50% greater than any other option and were not considered further. The least expensive and most feasible option involves adding straight alum to East Bay and alum with lime to buffer the treatment in Lower East Bay.

Lime has been added to lakes over the past few decades to control internal phosphorus loading but the effects have been short lived (Cooke et al. 2005), mainly due to the fact that calcium bound phosphorus formation is most effective at a pH greater than 9. Because the pH in Little St. Germain Lake is not generally sustained at or above 9, especially above the sediment surface where the lime would settle to, no additional impact to internal phosphorus loading is considered with its application.

Alum and lime would be applied in up to three phases in the lake over a three year period. The estimated cost per treatment phase for the application of alum will be $202,000 (Table 5). It should be noted that this cost estimate is based on historically high aluminum prices due to substantial cost increases during 2008. Contract rates for alum have increased approximately 45% since 2007 when the original feasibility study for the alum treatment was conducted.

Table 5. Recommended alum treatment option for East Bay and Lower East Bay. Cost includes lime.

Treatment Zone	Alum Applied (gallons)
East	57250
Lower East	66690
Total	123940
Treatment cost per phase	**$ 202,000**

Additional cost for treatment monitoring and contractor bidding and supervision will be $7,750 for each additional phase of treatment beyond the first application. For example, the total cost of the phase two treatment would be approximately $209,750 including the application itself, treatment monitoring, contract bidding, and contractor supervision costs.

5.3.2 Timing

Alum treatment should occur during the fall to avoid fish spawning concerns, low dissolved oxygen conditions, and impacts to recreational users. Treatment should also occur at least two weeks after any planned herbicide treatment in the fall. Dissolved oxygen monitoring should be conducted prior to alum treatment to determine if oxygen depletion after herbicide treatment is a concern.

Additional water column samples should also be collected and analyzed immediately before treatment. Surface and bottom water samples should be collected from each bay and analyzed for pH, ions and alkalinity. The results will be used to modify the alum dose, if necessary, to maintain a pH of 6 or greater.

Due to potential horizontal drift of the aluminum floc in the lake water, requirements will be placed on the contractor stipulating a maximum wind velocity threshold for treatment. In addition, treatment will not be allowed if substantial precipitation is expected to occur immediately prior to, during, or shortly after the application.

5.3.3 Application Sequencing

An approximate breakdown of the alum treatment schedule is below.

Day 1:

- Treatment should begin from the far eastern end of Lower East Bay and move to the Lower East Bay/East Bay border. Only a portion of the required alum dose for this phase should be applied. The remainder will be applied after lime application.

Day1/Day 2

- An application of lime should then be conducted at the Lower East Bay.

Day 3/4

- Alum treatment should then continue from the eastern end of the East Bay in a southward direction.

Day 4 or 5

- The Lower East Bay should be treated with the second portion of the alum dose in the same manner that the first dose was applied.

Sequencing the alum treatment in this manner will allow greater fish movement during the alum application and will also allow the lime addition to buffer the alum treatment in Lower East Bay.

5.4 Risk Assessment

Section four of this report discussed potential for fish and benthic invertebrate toxicity along with possible changes in macrophyte growth patterns. This section covers risks associated directly with water quality improvement and alum treatment.

5.4.1 No treatment option

If no treatment of the East and Lower East Bays is chosen, average summer total phosphorus, chlorophyll *a*, and Secchi disc depth are expected to remain within the range seen since 2000 (Table 6). Annual increases in late summer total phosphorus levels will continue to degrade water quality in the lake. Phosphorus in the water of East Bay and Lower East Bay will continue to negatively impact South Bay water quality as well.

5.4.2 Lower than expected performance

Because alum treatment of Little St. Germain was designed based on sediment mobile phosphorus, it is not expected that internal loading will substantially contribute to excess phosphorus in the water column of the treated bays. However, due to low alkalinity in the water column, the total alum dose required to convert mobile phosphorus to aluminum bound phosphorus in the sediment will need to be split and applied in separate phases. It is possible that lower than expected results will occur until the final alum treatment is complete and all mobile phosphorus is converted to aluminum bound phosphorus.

Treatment effectiveness will appear to be lower in years with high precipitation and elevated external phosphorus loading relative to average or dry years. However, internal phosphorus loading will remain low, even in wet years.

5.4.3 Expected performance and longevity

After all phases of alum treatment are complete for East Bay and Lower East Bay, average summer water clarity will increase by approximately 1.4 feet and total phosphorus and chlorophyll *a* will decrease by 29 µg/L and 23 µg/L, respectively. Improvements in water quality are also expected in South Bay (Table 6).

Table 6. Expected increases in water quality for East Bay, Lower East Bay, and South Bay after alum treatment.

	East Bay/Lower East Bay			South Bay[(2)]		
Alum Treated Area	**Total Phosphorus (ug/L)**	**Chloropyll *a* (ug/L)**	**Secchi disc depth (ft/m)**	**Total Phosphorus (ug/L)**	**Chloropyll *a* (ug/L)**	**Secchi disc depth (ft/m)**
No Treatment	62	38	2.8/0.9	46	19	4.3/1.3
East/Lower East Bay Only: **Proposed Option**[(1)]	33	15	4.2/1.3	28	10	6.1/1.9

(1)Average of modeling results for 2001, 2002, and 2007. Average for June through August period.
(2)Average of modeling results for the year 2002.

Treatment longevity is a function of multiple factors, including:

- The amount of mobile sediment phosphorus remaining after treatment
- The deposition rate of new sediment in each bay
- The concentration of different phosphorus fractions in newly deposited sediment

- The physical and chemical properties (e.g. dissolved oxygen) of the lake water after treatment

- The guidelines used to determine treatment effectiveness

As new sediment accumulates in each bay, internal phosphorus loading will eventually return due to new phosphorus entering the system. Because there are no data on sediment deposition rates or the fraction of phosphorus in newly deposited sediment, a precise longevity estimate is difficult to calculate. In addition, sedimentation rates can be expected to drop after treatment, due to lower algal production and settling to the sediment. Based on previous treatments with alum, and the fact that this treatment dose was calculated based on mobile sediment phosphorus, longevity is expected to be at least 10 years from the initial application.

5.5 Treatment Effectiveness Monitoring Program

Sufficient pre-treatment data exist so that post treatment monitoring can determine the effectiveness of alum treatment in Little St. Germain Lake. Both water column and sediment data should be collected after alum application to monitor the effectiveness of treatment and to determine if alum was applied by the contractor as required. Monitoring of pH will also be conducted during the treatment.

5.5.1 Monitoring of pH

During the treatment application, pH will be monitored to make sure that it does not drop below 6. Monitoring will need to be conducted for each phase of the treatment.

5.5.2 Sediment Cores

Sediment cores should be collected from the same locations determined during the 2007 sediment monitoring program. Approximately 10 sediment cores will be collected and analyzed for mobile and aluminum bound phosphorus fractions, as well as aluminum, to determine if the coverage of the alum application was completed properly and how much mobile phosphorus in the sediment was converted to aluminum bound phosphorus. Aluminum and phosphorus distribution maps will be created so this can be seen graphically as well.

5.5.3 Water Quality Monitoring

Water quality monitoring in the lake should continue to be conducted as it was before treatment to assess the treatments impact on surface water quality and longevity of the treatment effects. Additional monitoring in the Lower East Bay, similar to that conducted in 2008, should be conducted after treatment to assess the treatment effectiveness to reduce internal phosphorus loading to the water column.

6.0 Summary

Water quality in Little St. Germain Lake is in the eutrophic range and appears to have worsened over the last two decades. The East and Lower East Bays are especially problematic with higher phosphorus and chlorophyll *a* and lower Secchi disc depth than either South Bay or West Bay. Seasonal trends in water quality show that degradation occurs during the summer when phosphorus contributions from inflows are lower but internal phosphorus loading is elevated. The degraded water quality has negative impacts on aesthetics leading to lower enjoyment of the lake by residents and others who use the lake for these purposes.

Two options were studied to improve lake water quality in Little St. Germain Lake: inflow alum treatment and whole lake alum treatment. Modeling of the two different options showed that applying alum to the lake would improve water quality to a greater degree than treating the inflows with alum, and for a lower cost.

Based on sediment analysis and lake modeling, an alum dose was calculated to improve water quality in the lake. Applying the required alum and reducing internal phosphorus loading from the sediment will have the following benefits:

- An average decrease in phosphorus and chlorophyll *a* in the surface water of the East/Lower East Bays by 29 µg/l and 23 µg/L, respectively. The reduction in algal growth will lead to an average increase in Secchi disc depth of approximately 1.4 feet.

- An average decrease in phosphorus and chlorophyll *a* in the surface water of South Bay 18 µg/L and 9 µg/L and an average increase in Secchi disc depth of 1.8 feet.

Due to the low buffering capacity of the lake water, it is recommended that the alum treatment be split into three phases and that lime be used to buffer the treatment in Lower East Bay. Applying alum in phases reduces the need for more costly buffered aluminum compounds to prevent pH depression during treatment and allows for an adaptive management approach. The need for the third and final treatment phase will be based upon mobile phosphorus reduction (goal is 90 % reduction) in the lake sediment, and lake water clarity improvement after the second phase of treatment. It is recommended that water clarity be evaluated on an ongoing basis to determine the need for a potential third phase of alum treatment.

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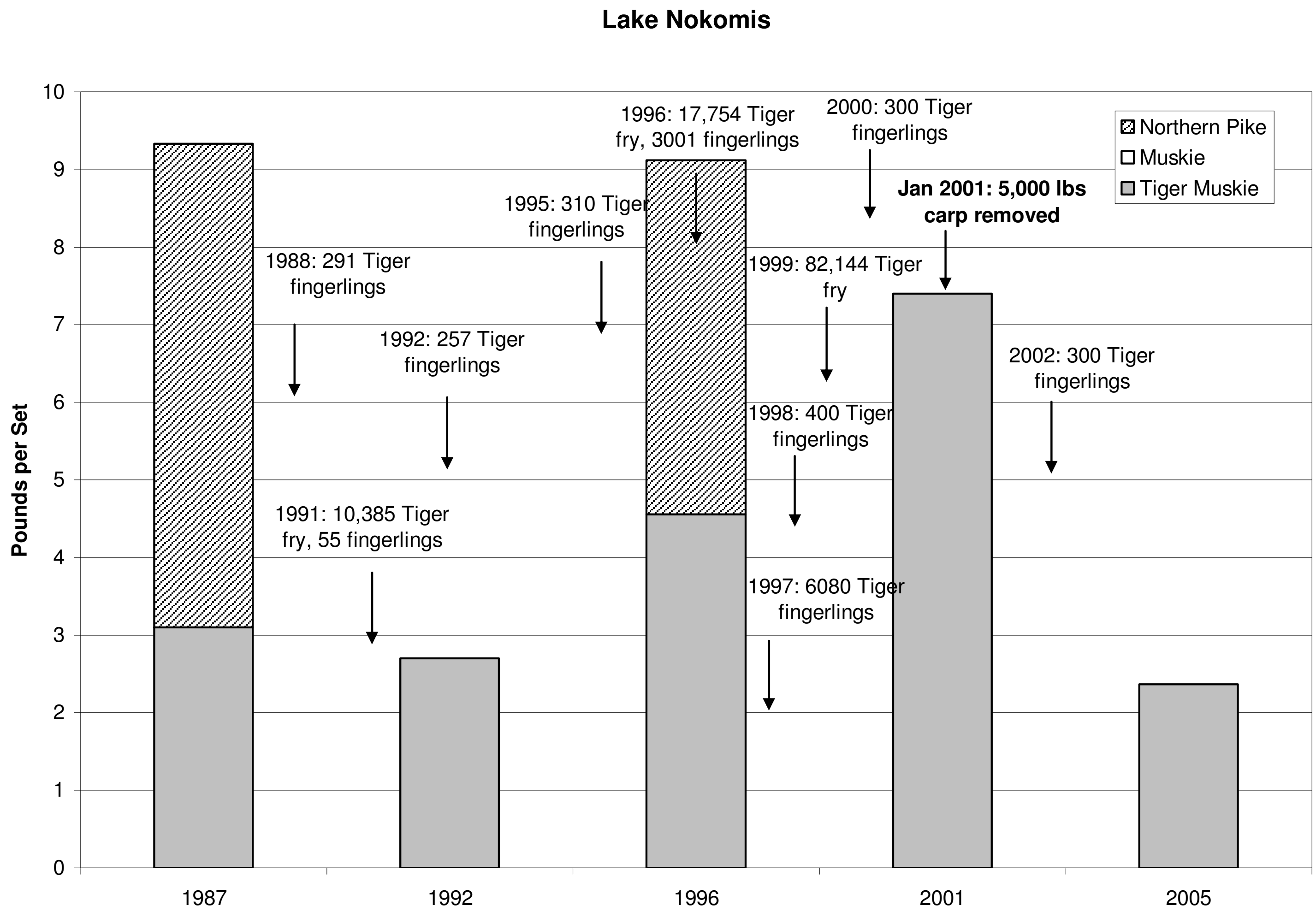
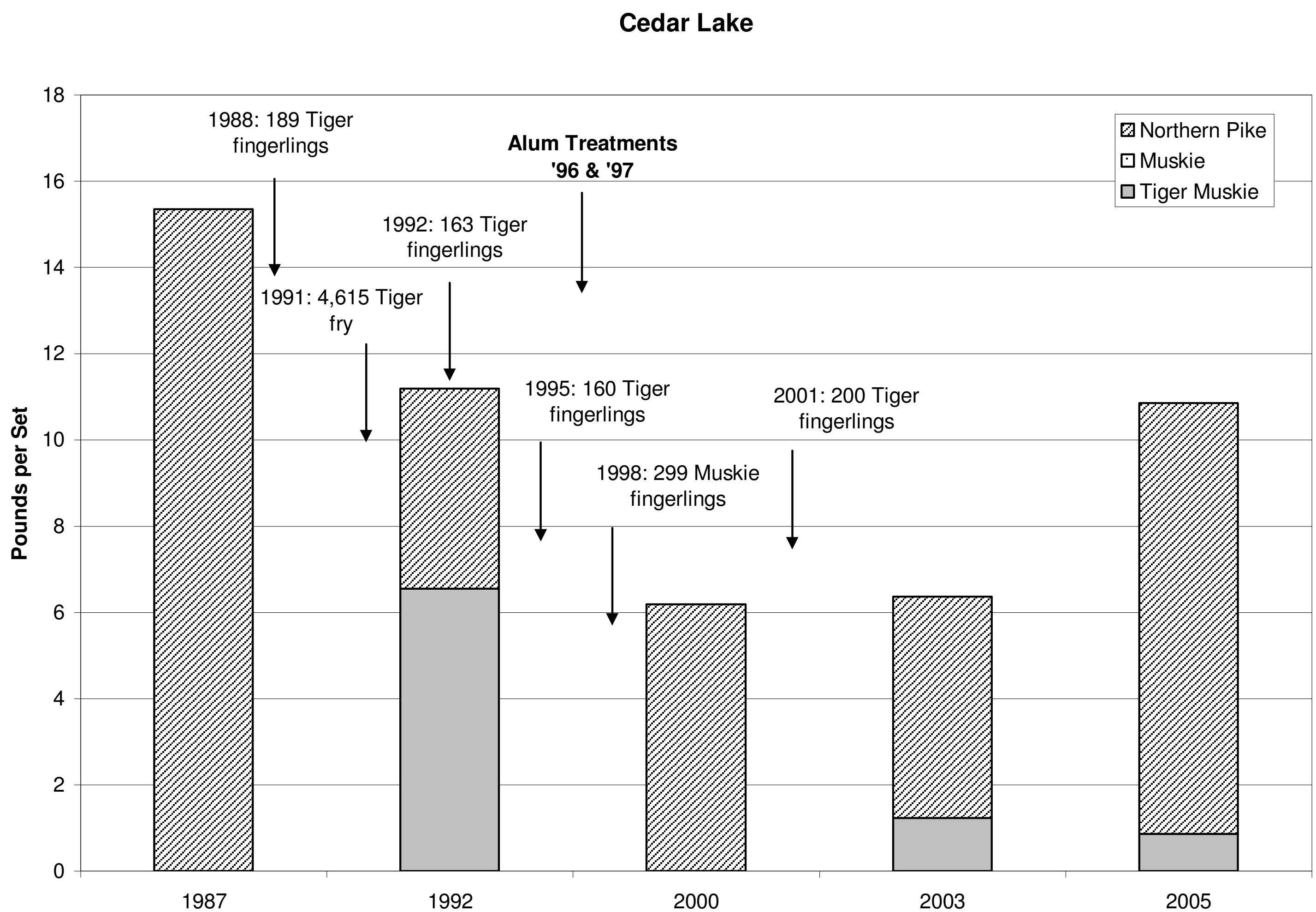
Appendices

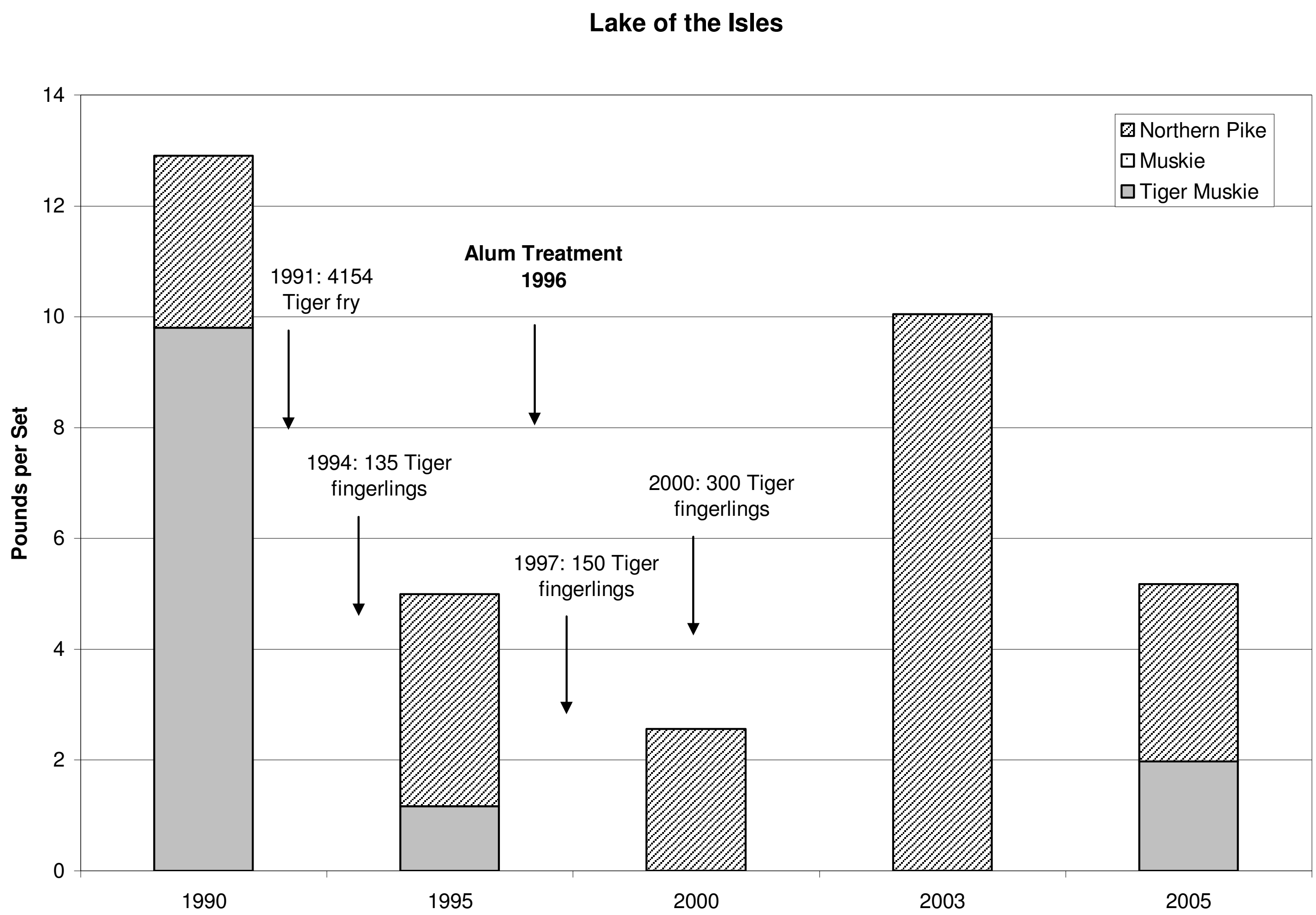
Appendix A-Fisheries Data

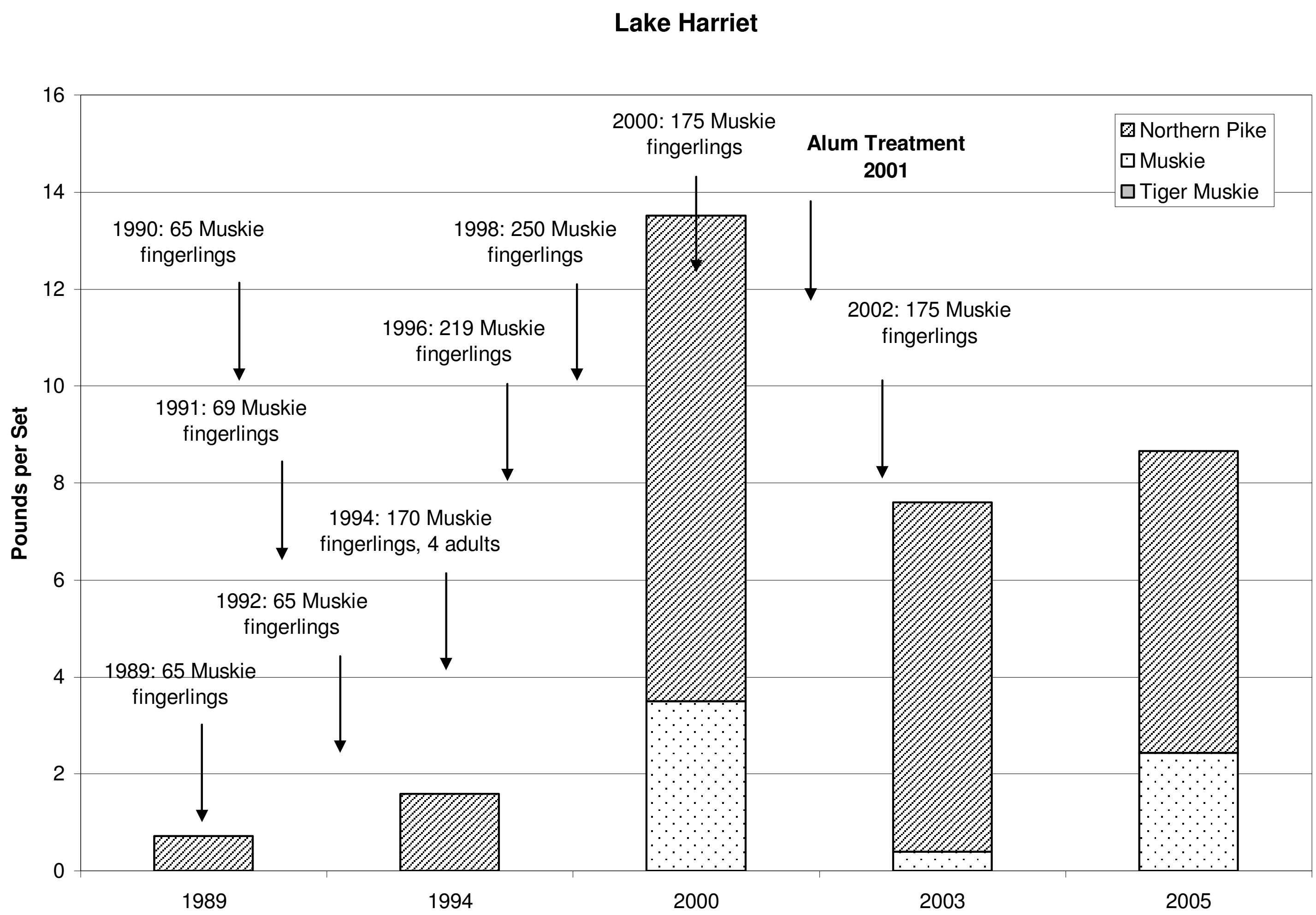
Minneapolis Chain of Lakes Data



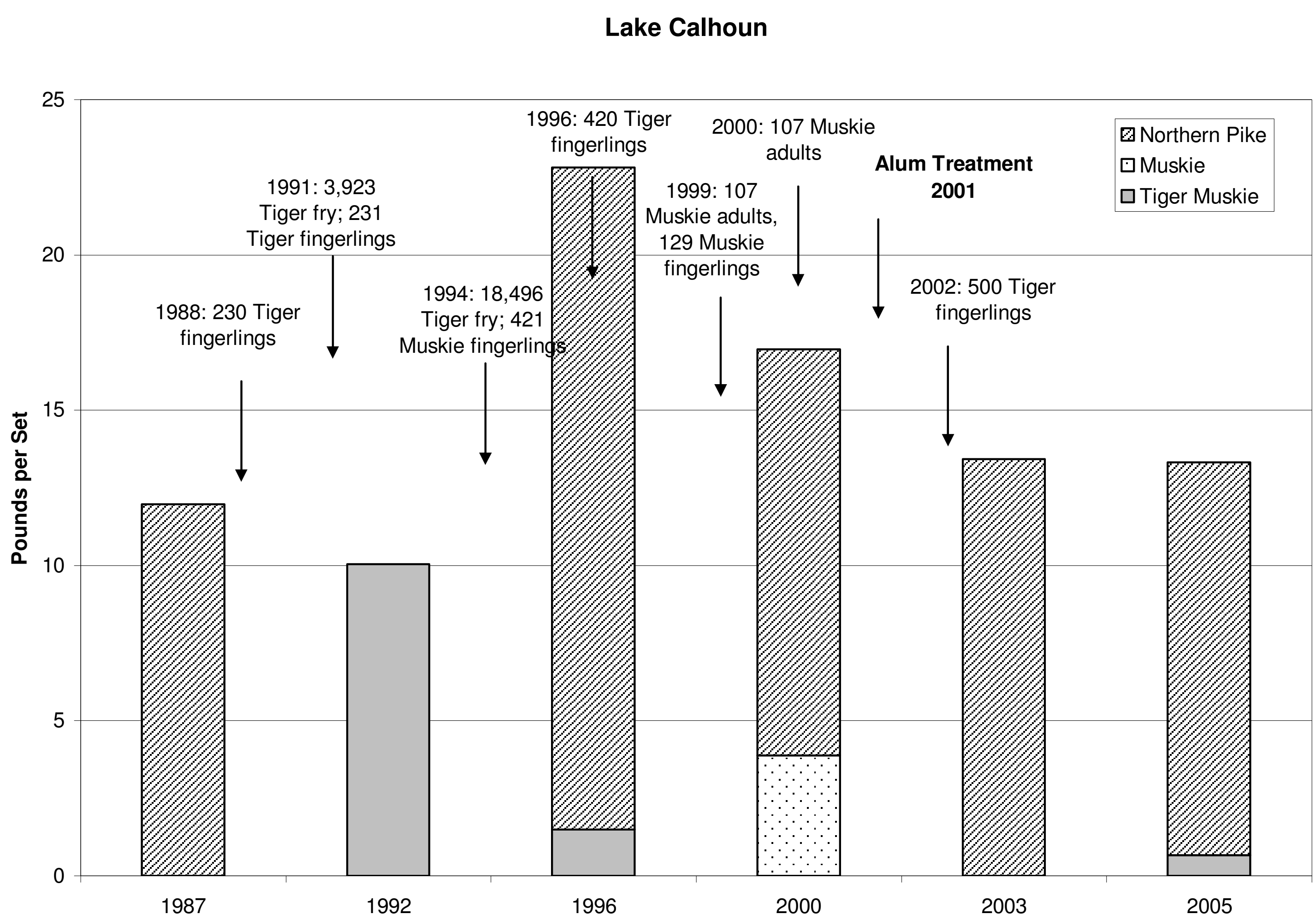


Cedar Lake
Pounds per Set
18
16
14
12
10
8
6
4
2
0
1987
1992
2000
2003
2005
Northern Pike
Muskie
Tiger Muskie
1988: 189 Tiger fingerlings
1991: 4,615 Tiger fry
1992: 163 Tiger fingerlings
Alum Treatments '96 & '97
1995: 160 Tiger fingerlings
1998: 299 Muskie fingerlings
2001: 200 Tiger fingerlings


Lake of the Isles
Pounds per Set
14
12
10
8
6
4
2
0
1990
1995
2000
2003
2005
Northern Pike
Muskie
Tiger Muskie
1991: 4154 Tiger fry
Alum Treatment 1996
1994: 135 Tiger fingerlings
1997: 150 Tiger fingerlings
2000: 300 Tiger fingerlings



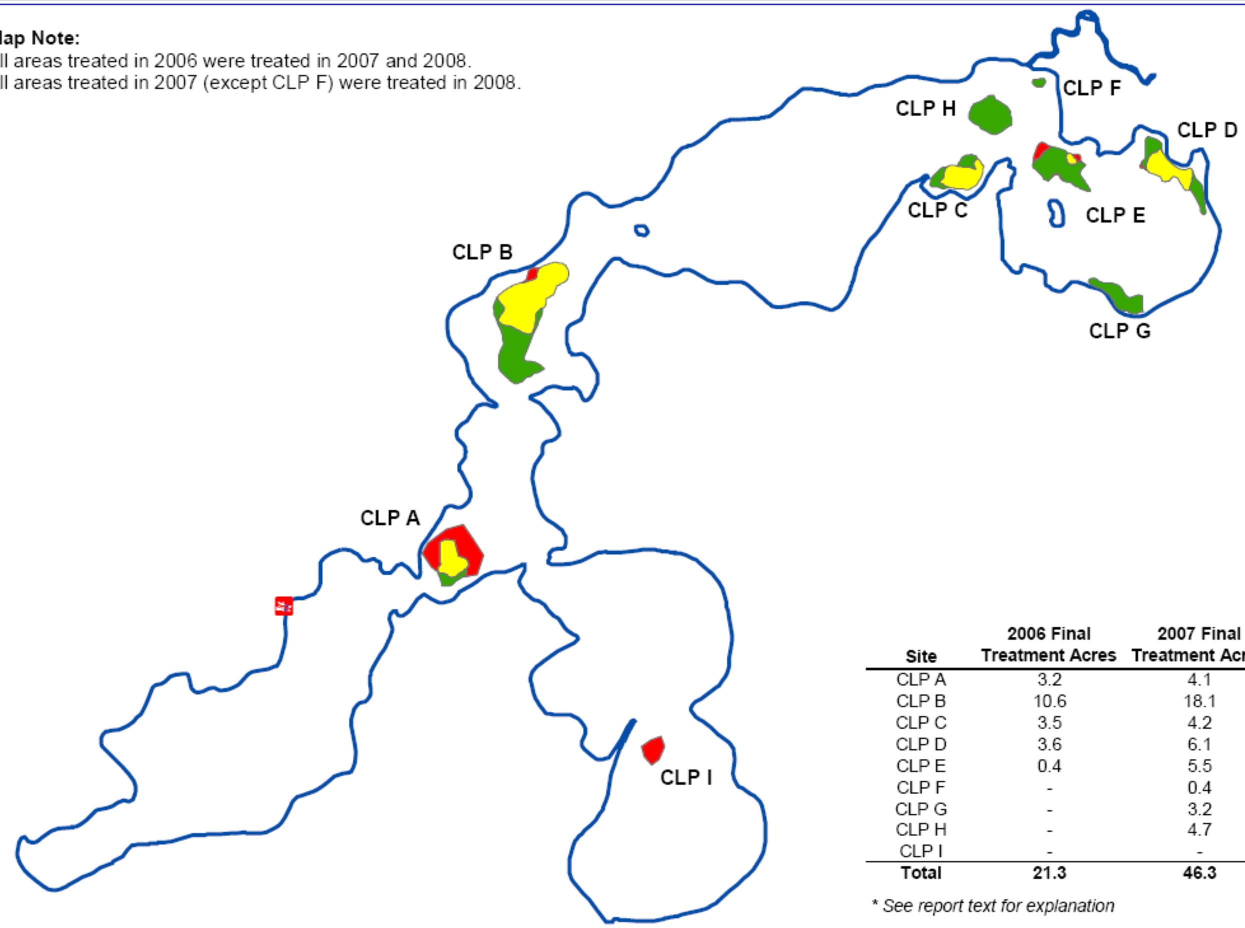
Lake Harriet
Pounds per Set
16
14
12
10
8
6
4
2
0
1989
1994
2000
2003
2005
Northern Pike
Muskie
Tiger Muskie
1989: 65 Muskie fingerlings
1990: 65 Muskie fingerlings
1991: 69 Muskie fingerlings
1992: 65 Muskie fingerlings
1994: 170 Muskie fingerlings, 4 adults
1996: 219 Muskie fingerlings
1998: 250 Muskie fingerlings
2000: 175 Muskie fingerlings
Alum Treatment 2001
2002: 175 Muskie fingerlings



Lake Calhoun
Pounds per Set
25
20
15
10
5
0
1987
1992
1996
2000
2003
2005
Northern Pike
Muskie
Tiger Muskie
1988: 230 Tiger fingerlings
1991: 3,923 Tiger fry; 231 Tiger fingerlings
1994: 18,496 Tiger fry; 421 Muskie fingerlings
1996: 420 Tiger fingerlings
1999: 107 Muskie adults, 129 Muskie fingerlings
2000: 107 Muskie adults
Alum Treatment 2001
2002: 500 Tiger fingerlings

Appendix B-Macrophyte Maps

Map Note:
All areas treated in 2006 were treated in 2007 and 2008.
All areas treated in 2007 (except CLP F) were treated in 2008.



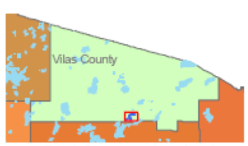
Site	2006 Final Treatment Acres	2007 Final Treatment Acres	2008 Final Treatment Acres
CLP A	3.2	4.1	9.9
CLP B	10.6	18.1	18.6
CLP C	3.5	4.2	5.1
CLP D	3.6	6.1	6.1
CLP E	0.4	5.5	6.3
CLP F	-	0.4	Not Treated*
CLP G	-	3.2	3.2
CLP H	-	4.7	4.7
CLP I	-	-	1.8
Total	**21.3**	**46.3**	**55.7**

** See report text for explanation*

2,000
Feet

Onterra LLC
Lake Management Planning
135 South Broadway Suite C
De Pere, WI 54115
920.338.8860
www.onterra-eco.com

Sources:
Roads & Hydro: WDNR
Aquatic Plants: Onterra, 2006,2008
Bathymetry: WDNR
Map date: October 28, 2008



Extent of large map shown in red.

Legend

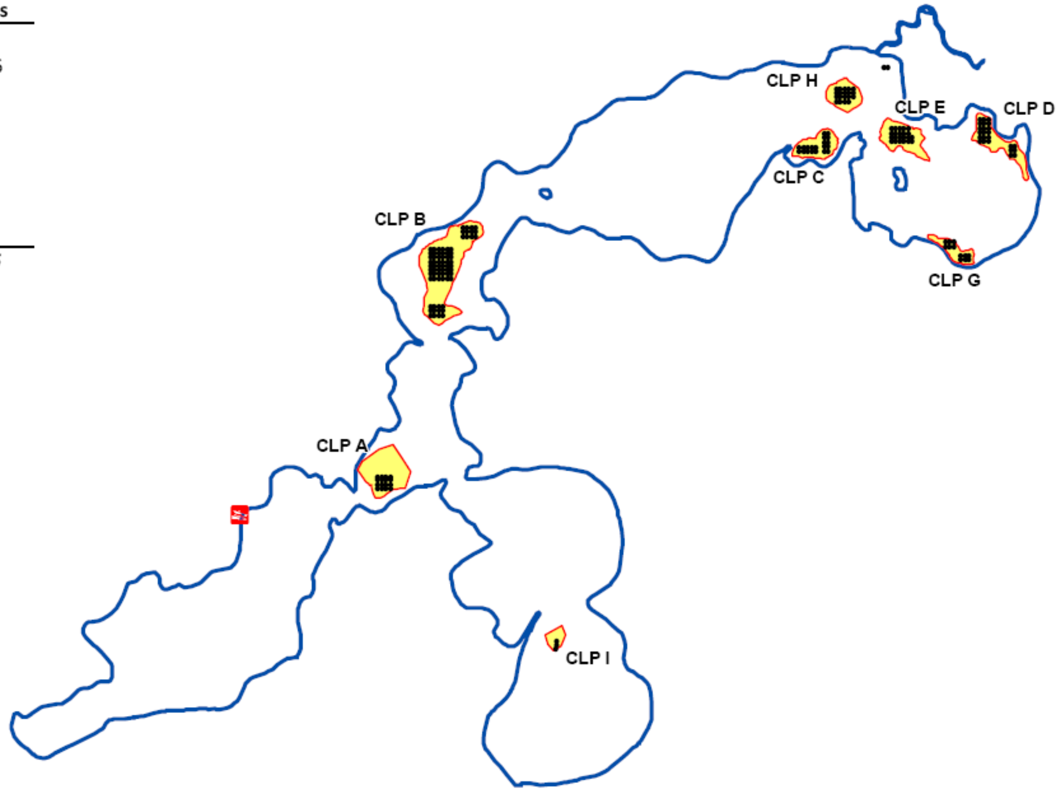
- 2006 Treatment
- Additional Areas included in 2007 Treatment
- Additional Areas included in 2008 Treatment

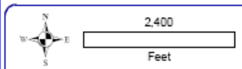
Map 5

Little Saint Germain Lake
Vilas County, Wisconsin

2006-2008 CLP Treatment Areas

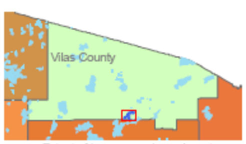
Treatment Areas	
Site	**Acres**
CLP A	9.9
CLP B	18.6
CLP C	5.1
CLP D	6.1
CLP E	6.3
CLP G	3.1
CLP H	4.7
CLP I	1.7
Total	***55.5***





Onterra LLC
Lake Management Planning
135 South Broadway Suite C
De Pere, WI 54115
920.338.8860
www.onterra-eco.com

Sources:
Roads & Hydro: WDNR
Aquatic Plants: Onterra, 2008
Bathymetry: WDNR
Map date: May 8, 2008



Extent of large map shown in red.

Legend

2008 Final Treatment Area

• Point-intercept sub-sample location

Map 6
Little Saint Germain Lake
Vilas County, Wisconsin
2008 Final
CLP Treatment Areas
Following Pretreatment Surveys

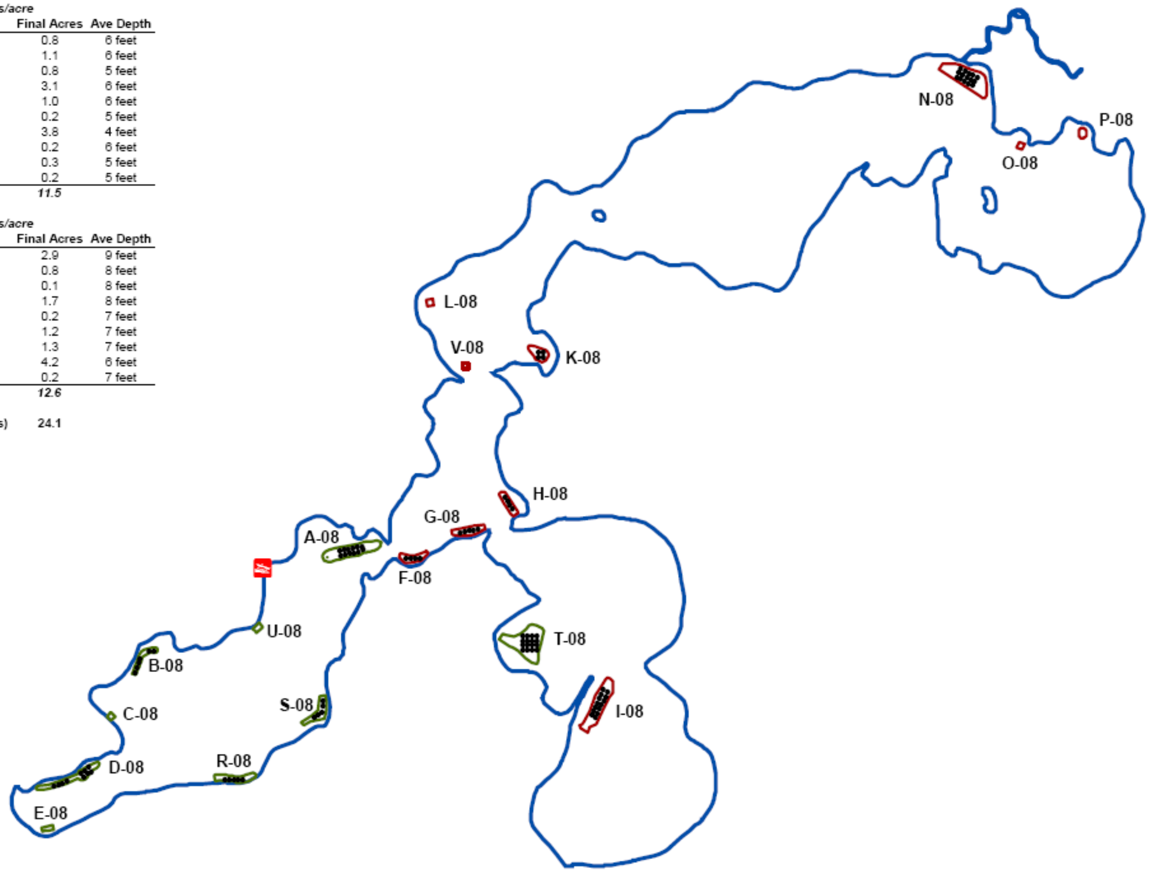
Treatment Area - 100 lbs/acre

Site	Final Acres	Ave Depth
F-08	0.8	6 feet
G-08	1.1	6 feet
H-08	0.8	5 feet
I-08	3.1	6 feet
K-08	1.0	6 feet
L-08	0.2	5 feet
N-08	3.8	4 feet
O-08	0.2	6 feet
P-08	0.3	5 feet
V-08	0.2	5 feet
Subtotal	*11.5*	

Treatment Area - 150 lbs/acre

Site	Final Acres	Ave Depth
A-08	2.9	9 feet
B-08	0.8	8 feet
C-08	0.1	8 feet
D-08	1.7	8 feet
E-08	0.2	7 feet
R-08	1.2	7 feet
S-08	1.3	7 feet
T-08	4.2	6 feet
U-08	0.2	7 feet
Subtotal	*12.6*	

Grand Total (Final Acres) 24.1

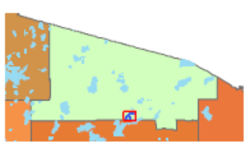


2,100

Feet

Onterra LLC
Lake Management Planning
135 South Broadway Suite C
De Pere, WI 54115
920.338.8860
www.onterra-eco.com

Sources:
Roads & Hydro: WDNR
Aquatic Plants: LSGPRD, 2007
Bathymetry: WDNR, Digitized by Onterra
Map date: June 4, 2008



Extent of large map shown in red.

Legend

2008 EWM Final Treatment Areas

100 lbs/acre

150 lbs/acre

Point-intercept sub-sample location

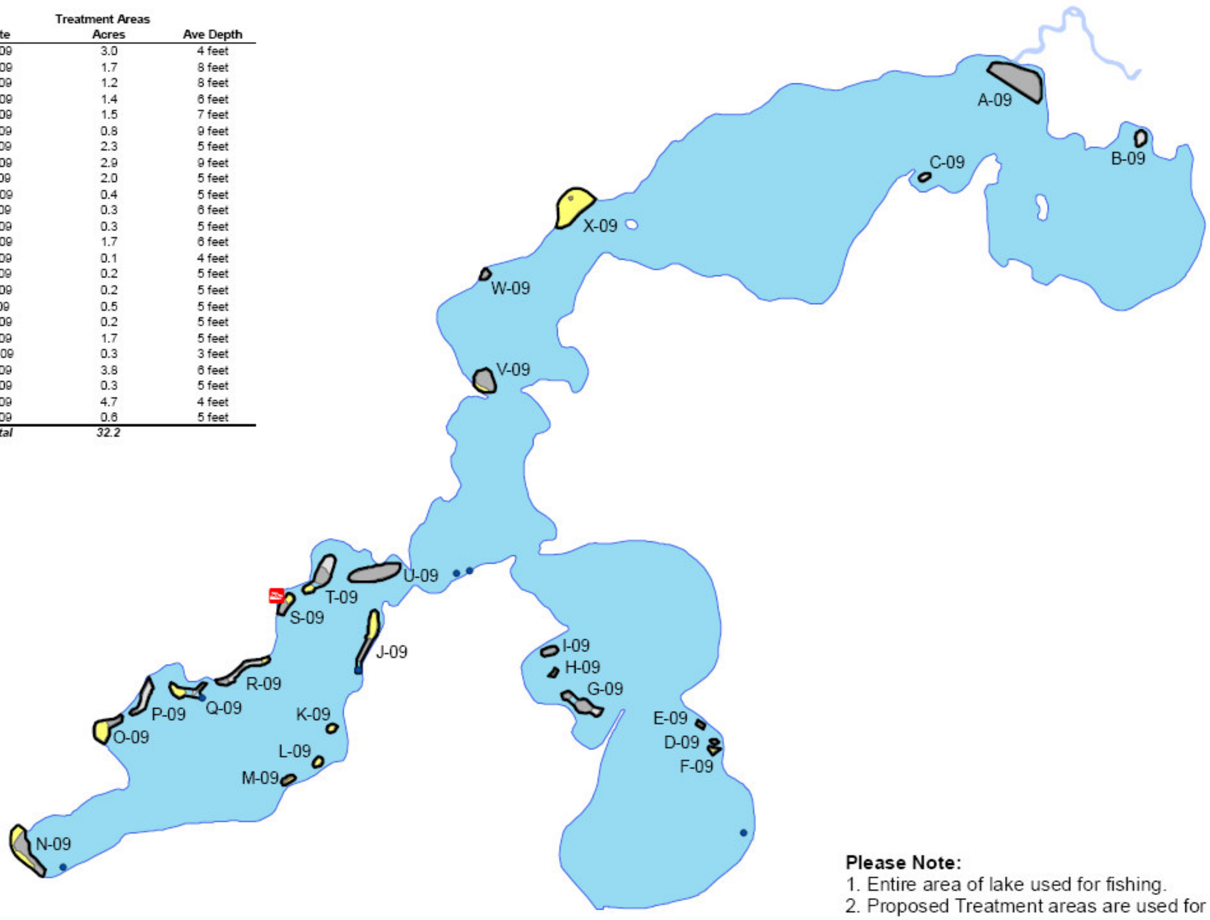
Map 7

Little Saint Germain Lake

Vilas County, Wisconsin

2008 Final
EWM Treatment Areas

	Treatment Areas	
Site	Acres	Ave Depth
N-09	3.0	4 feet
O-09	1.7	8 feet
P-09	1.2	8 feet
Q-09	1.4	6 feet
R-09	1.5	7 feet
S-09	0.8	9 feet
T-09	2.3	5 feet
U-09	2.9	9 feet
J-09	2.0	5 feet
M-09	0.4	5 feet
L-09	0.3	6 feet
K-09	0.3	5 feet
G-09	1.7	6 feet
E-09	0.1	4 feet
F-09	0.2	5 feet
D-09	0.2	5 feet
I-09	0.5	5 feet
H-09	0.2	5 feet
V-09	1.7	5 feet
W-09	0.3	3 feet
X-09	3.8	6 feet
C-09	0.3	5 feet
A-09	4.7	4 feet
B-09	0.6	5 feet
Total	*32.2*	



Please Note:
1. Entire area of lake used for fishing.
2. Proposed Treatment areas are used for all boating activities.

1,900
Feet

Onterra LLC
Lake Management Planning
135 South Broadway Suite C
De Pere, WI 54115
920.338.8860
www.onterra-eco.com

Sources:
Roads & Hydro: WDNR
Treatment Areas: Onterra, Sept. 2008
Aquatic Plants: Onterra, Sept. 2008
Map date: June 4, 2008

Extent of large map shown in red.

Legend

Summer 2008 EWM Densities
- Clump
- Highly Scattered
- Scattered
- Dominant
- Highly Dominant

2009 Proposed EWM Treatment Areas v.1 (summer 2008)

Public Boat Landing

Map 9
Little Saint Germain Lake
Vilas County, Wisconsin
2009 Proposed EWM Treatment Areas