FEASIBILITY STUDY FOR THE CONTROL OF INTERNAL PHOSPHORUS LOADING IN ST. ALBANS BAY, LAKE CHAMPLAIN

Phase 1: Evaluation of Alternatives

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June 2007

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1.0 EXECUTIVE SUMMARY

St. Albans Bay of Lake Champlain covers 3,903 acres and has an estimated volume of 74,221 acre-feet. Stevens Brook, Jewett Brook, and the Mill River drain 87% of the 33,359 acre watershed. These tributaries have some of the largest phosphorus loads in the Lake Champlain drainage basin (Smeltzer, 2003). The phosphorus in the tributaries originates from point sources (21%), such as direct farmyard flow and wastewater from the St. Albans municipal wastewater treatment plant, and nonpoint sources (79%), such as agriculture, development, forests/wetlands, and city stormwater not regulated under NPDES.

St. Albans Bay experiences late summer algal blooms and excessive aquatic plant growth due to high phosphorus loads to the bay. These conditions have led to a considerable decline in recreational use of the bay since the 1960's and a depression of property values relative to other areas on Lake Champlain. Efforts to reduce loads began in the 1980s when phosphorus levels in the discharge from the wastewater treatment plant were greatly reduced. Some best management practices have been implemented on agriculture lands in the watershed in recent years. However, despite these efforts, the phosphorus levels in St. Albans Bay remain well above the 17 μg/L Vermont State Water Quality criterion and the 8 mt P/yr TMDL for phosphorus loading (VT DEC and NY DEC 2002a). High phosphorus loads from the past are thought to have accumulated in the sediment and internal phosphorus loading may be slowing or inhibiting the water quality improvement that was anticipated with watershed management efforts.

The goals of attaining water quality standards for designated uses and eliminating algal blooms may be facilitated through the control of internal phosphorus loading. In-lake phosphorus management options include circulation, dredging, and internal phosphorus inactivation. However, in-lake treatments alone will not solve the algal bloom problems of the enriched bay on more than a temporary basis. It is believed that further reductions in watershed nutrient loading must precede any in-lake treatment. An interim nutrient input reduction accomplished by dosing tributaries with aluminum during periods of elevated flow and nutrient concentrations could provide relief until watershed management actions can be fully implemented.

Upon consideration of the technical feasibility, cost, and impacts of each applicable management technique, it appears that no one technique is adequate by itself and that the extent of needed application is very large if the problems of the entire bay are to be addressed. A phased program with a focus on one portion of the bay is suggested as a logical approach at this time. The planned summer 2007 test of water column mixing equipment near the park in the inner bay will supply useful information on the efficacy of that approach. Phosphorus loading from the watershed should be substantially reduced prior to any inlake treatment. However, a program that blends inactivation of phosphorus in incoming water from Black Creek as an interim measure while watershed sources are being addressed along with with treatment of surficial sediments of the inner bay using phosphorus inactivation is recommended as the approach most likely to have acceptable and reliable results. If watershed loads are reduced substantially prior to the inner bay sediment phosphorus inactivation project, the inner bay treatment could occur without treating the inflow from Black Creek.

The cost of the suggested program could be between $1 million and $4.3 million based on currently available information. The most critical variable for determining the cost is the quantity of aluminum required for inactivating surficial sediment phosphorus. A Phase II program to better define the management program and prepare for implementation is suggested at a cost of $41,500. The cost could be reduced if the state, locals or the University of Vermont were to complete some of the identified tasks.

2.0 INTRODUCTION

St. Albans Bay is located in the northeast arm of Lake Champlain in Vermont near the city of St. Albans (Figure 1). St. Albans Bay covers 3,903 acres with an estimated volume of 74,221 acre-feet. The remainder of the 33,359 acre-watershed is used for agriculture (56%), covered by forest (24%), and developed (14%) (Gaddis, 2006). The bay is oriented along a SW-NE axis, exposing it to the prevailing southerly winds in the late summer and early fall.

St. Albans Bay is one of the most nutrient rich parts of Lake Champlain. The bay has long suffered from cyanobacteria blooms caused by high phosphorus enrichment, excess aquatic plant growth, and fecal bacteria contamination (Smeltzer, 2003). These water quality impairments led to reduction in recreation use as early as the 1960s and a depression of property values of land adjacent to the bay relative to properties elsewhere on Lake Champlain.

Both point and nonpoint sources contributed to the historic phosphorus enrichment in St. Albans Bay. In 1987, the upgrade of the St. Albans wastewater treatment plant reduced phosphorus export from that source by 90% to achieve an effluent concentration of 0.50 mg/L. Initiation of the U.S. Department of Agriculture Rural Clean Water Program in 1980 attempted to reduce nonpoint source loading by implementing best management practices on 60% of farms in the watershed. However, phosphorus levels in St. Albans Bay have remained higher than the 17 μg/L Vermont state water quality criterion and the 8 mt P/yr TMDL phosphorus loading target (VT DEC and NY DEC 2002a). Nonpoint source phosphorus loading from the St. Albans Bay watershed continues to rate the highest in the Lake Champlain Basin (Smeltzer, 2003).

In response to the lack of improvement in water quality despite efforts to reduce external phosphorus loading, Vermont Agency of Natural Resources contracted ENSR Corporation to conduct a feasibility study of the techniques to manage internal phosphorus loading to St. Albans Bay. This report outlines a plan of action to reduce internal phosphorus loading as a part of an effort to rehabilitate St. Albans Bay by attaining phosphorus concentration standards and reducing algal blooms to the lowest possible level.

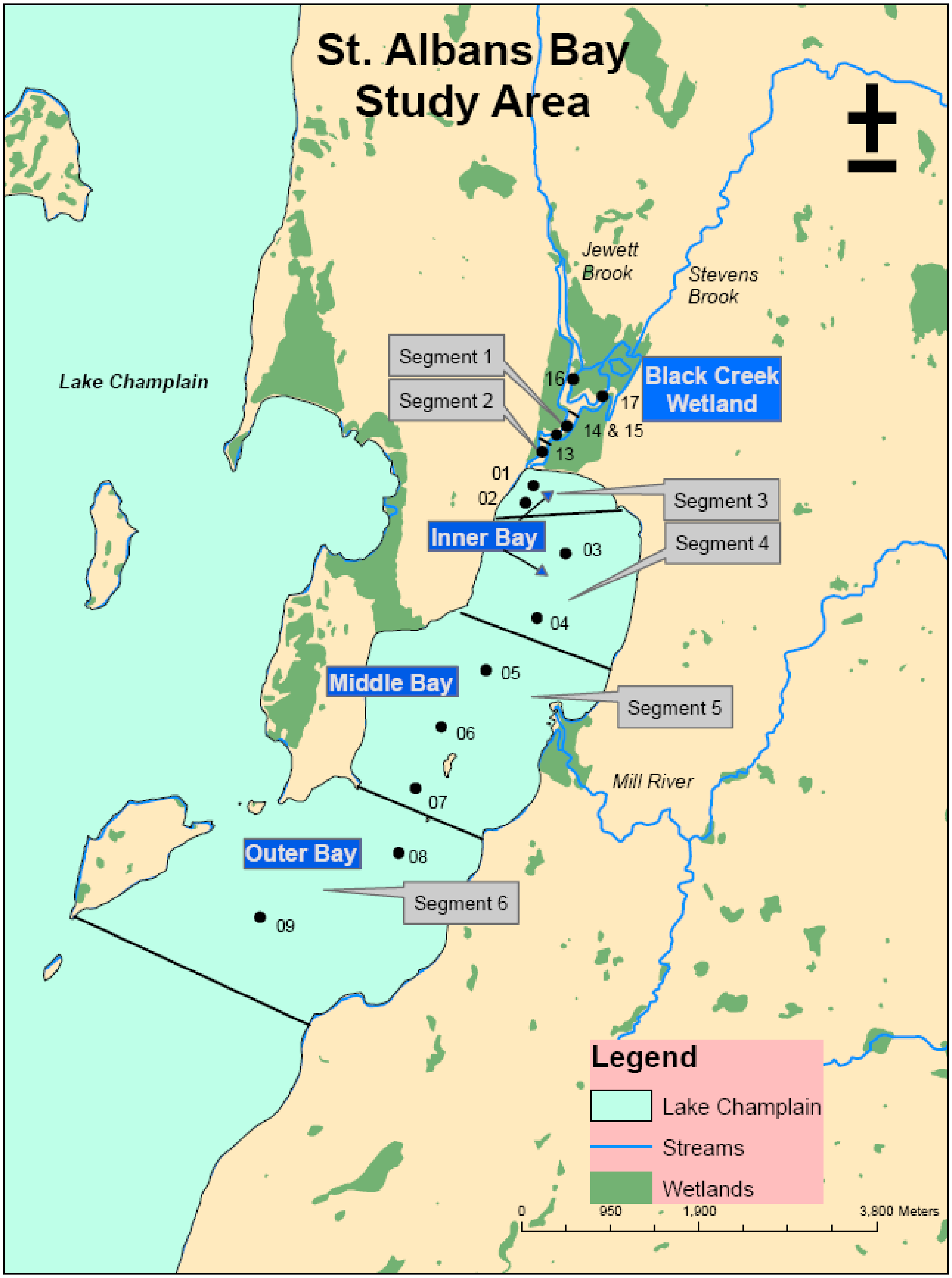

Figure 1. St. Albans Bay study regions. The Black Creek Wetland region refers to the area defined by Druschel (2005) as Stevens Brook Wetland. The Inner Bay incorporates both Segments 3 and 4 (Smeltzer et al., 1994). The Middle Bay refers to Segment 5 and the Outer Bay refers to Segment 6.

3.0 HISTORICAL DATA REVIEW AND DIAGNOSTIC SUMMARY

St. Albans Bay receives most of its water from three major tributaries draining the 33,359-acre watershed with three major tributaries: Stevens Brook, Jewett Brook, and Mill River. Stevens and Jewett Brooks combine to form Black Creek and its adjacent wetland located at the north end of the bay (Smeltzer et al. 1994). Wind-driven exchange and flow from the watershed mixes and flushes water from the bay into the body of Lake Champlain. Water quality in the bay is perceived to be poor, mainly as a consequence of summer algal blooms. During times of strong southerly winds it is likely that surface waters and associated floating algal cells are blown to the north end of the bay, exacerbating the perception of the poor quality of water in the bay.

Due to excessive nutrient inputs from its watershed, St. Albans Bay is a eutrophic section of Lake Champlain. High levels of phosphorus, which fuel primary production, contribute to late summer algal blooms and excessive vegetative growth observed in the bay. Cyanobacteria genera identified in late summer algal blooms include *Microcystis*, *Aphanizomenon, Anabaena, Coelosphaerium, and Gloeotrichia,* with toxins detected during summer monitoring. The nutrient rich lake also fuels excessive aquatic plant growth. Submerged vegetative beds are dominated by the invasive species Eurasian watermilfoil (*Myriophyllum spicatum*) and the native but often nuisance species common waterweed (*Elodea canadensis*).

Common warmwater fish species that inhabit St Albans Bay include white perch, yellow perch, bluegill, pumpkinseed, largemouth bass, smallmouth bass, emerald shiner, spottail shiner, and banded killifish. State listed and rare species present in the bay and adjacent wetland include the spiny softshell turtle, white-water crowfoot, western chorus frog, eastern ribbonsnake, the common tern, and possibly mussel species (S. Parren 2007 and B. Pientka 2007, Personal Communication).

The Black Creek wetland associated with Jewett and Stevens Brooks is also very productive, but may not suffer from the same negative perception as the bay. In addition to submergent plants, it hosts emergent plant species including cattail (*Typha angustifolia*) and gaint burreed (*Sparganium eurycarpum*), as well as large populations of floating duckweed (*Lemna* sp) in the summer months. Waterfowl and other migratory birds utilize this wetland. The wetland also provides a spring spawning area for northern pike (B. Pientka 2007, Personal Communication). While there may be water quality and related biological issues associated with overfertilization, this wetland is viewed as a valuable habitat for many species.

Phosphorus loading from the watershed into the tributaries is estimated to be as high as 10.56 metric tons of phosphorus per year (mt P/yr) from a variety of point and nonpoint sources (Gaddis 2006). Loading from point sources, the St. Albans Waste Water Treatment Plant and direct barnyard runoff, account for 2.49 mt P/yr (21%) to the bay from the landscape. Nonpoint watershed sources account for most of the phosphorus loading 8.07 mt P/yr (79%) to the bay from the landscape. Most of the nonpoint loading occurs during storm events (42%) and snowmelt (27%) (Gaddis 2006). Nonpoint source contributions include agriculture (59%), non-city development (11%), forest and wetlands (5%), and city stormwater (4%). The percent contribution of specific point and nonpoint phosphorus sources presented by Harpe and Gaddis (2006) are shown in Figure 2.

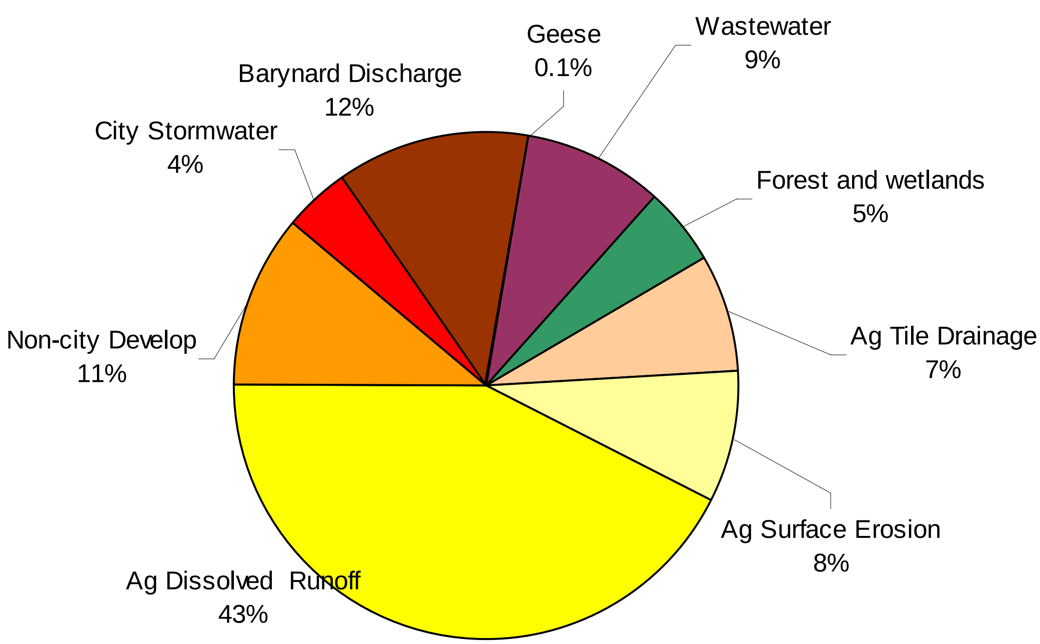

Figure 2. Relative contribution of phosphorus loading sources to St. Albans Bay tributaries (Harpe and Gaddis, 2006).

The long standing eutrophication of St. Albans Bay has resulted in intensive monitoring and study of the nutrient budget of the bay. The 2002 TMDL for the Vermont portion of Lake Champlain set a target load that would result in an in-lake phosphorus concentration of 17 μg/L for St. Albans Bay (VT DEC and NY DEC 2002a). The long-term monitoring records from Lake Champlain Long-Term Water Quality and Biological Monitoring Program (1992-2005), and the Vermont Lay Monitoring Program (1979-2005) provided a long-term water quality record to evaluate progress since the implementation of phosphorus reduction strategies in the 1980s (Picotte 2002, VT DEC and NY DEC 2002b). Water column phosphorus levels did not decrease toward target levels as expected (Figure 3).

In order to further understand why phosphorus levels were not decreasing as anticipated, HydroQual and Youngstown State University modeled internal phosphorus loading to the bay in the 1990s. Nonpoint sources in the watershed were identified as the primary sources of phosphorus loading, but the unbalanced budget suggested that 82% of total phosphorus was retained in the sediments (Cornwell and Owens 1999a, Cornwell and Owens 1999b). A mass balance suggested that phosphorus released from the sediments from historic loading prolonged the anticipated water column phosphorus reductions and continued to cause algal blooms (Martin et al. 1994, Smeltzer 1991).

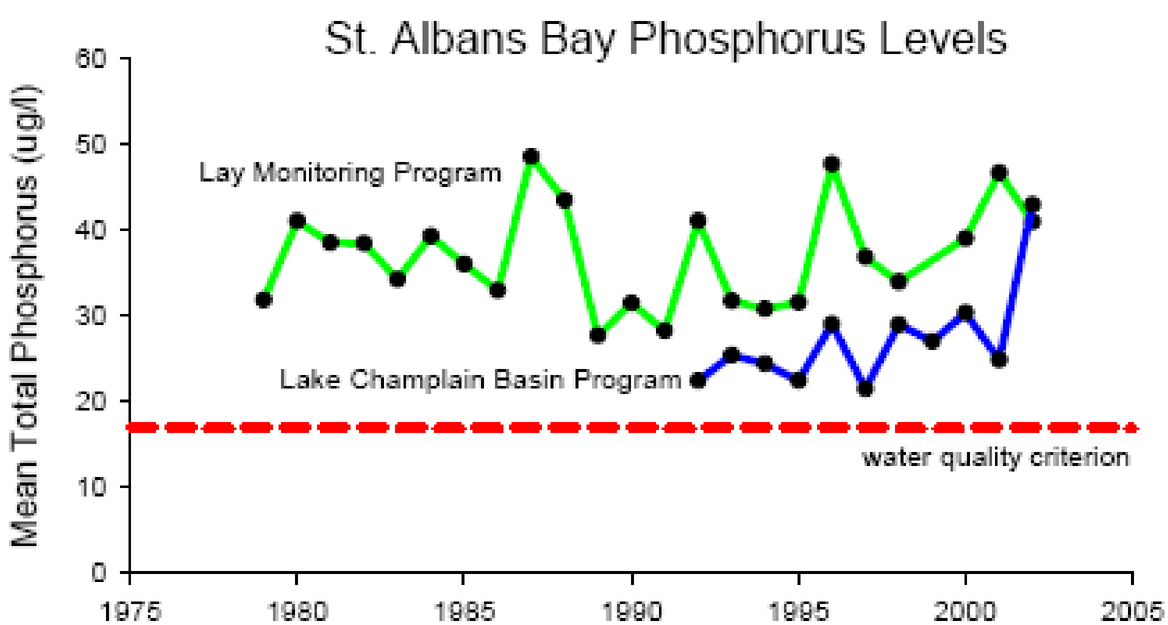

Figure 3. Long-term trends in total phosphorus concentration in St. Albans Bay compared to the Vermont water quality criterion of 17 ug/L. Water quality data are from Vermont Lay Monitoring Program and Lake Champlain Basin Program (Smeltzer 2003).

Studies also suggested that temperature, pH, and the release of nutrients from submerged and wetland vegetation may be contributing to internal phosphorus loading in St. Albans Bay (Smeltzer et al. 1994). This report demonstrated using both monitoring data and modeling that most of the internal loading of phosphorus to St. Albans Bay comes from the inner bay and the Black Creek wetland area as shown in Figure 1. Given the lack of thermal stratification and the consistently oxygenated water column of St. Albans Bay observed in monitoring data, anoxia in the overlying water is not believed to be a large contributing factor to phosphorus release from the sediments. Still, release of phosphorus under oxygenated conditions can be significant, if anoxic zones exist in the sediment and binding of phosphorus upon release is not rapid. This can occur when there is a large quantity of phosphorus in the sediment and not enough natural binder (e.g., iron, calcium, aluminum).

The most current data on phosphorus in the sediment were presented by Druschel et al. (2005), who estimated that approximately 500 tons of reactive phosphorus was in the top 4 cm of sediments over 1574 ha (3903 ac) of St. Albans Bay and the Black Creek wetland. Approximately 1200 tons were in the top 10 cm of sediments in that same area. Average "available" phosphorus, defined as reactive phosphorus in an extraction process using ascorbic acid, was 372 ug P/g of sediment. A separate, different pair of extractions (ammonium chloride and sodium hydroxide) for available phosphorus yielded average values of 14 ug/g and 363 ug/g, which added together, equals 377 ug/g, which is very similar to the first extraction procedure. Total phosphorus levels were actually much higher, averaging 3000 ug/g, but much of this phosphorus is permanently bound in the sediment. The values for available phosphorus were highly variable over space (Table 1), but are generally considered quite high. There is a high potential for release from sediments, even with oxygenated overlying water.

Table 1. Tabulated average, minimum, and maximum values for each region of St. Albans Bay; IB=Inner Bay, MB=Middle Bay, OB=Outer Bay, SBW=Stevens Brook Wetland. (Table 7 from Druschel et al. 2005)

				Totals Aqua Regia Extraction			Sequential Extraction ——> NH4Cl	NaOH	HCl			Ascorbic Acid		
		Porosity	Organic Content	Fe	Mn	P	P	P	Fe	Mn	P	Fe	Mn	P
average	IB	55%	4.2	32883	794	1394	13	280	5066	83	400	3435	207	272
	MB	60%	4.1	39689	845	1474	10	263	5421	126	337	3620	288	263
	OB	65%	4.3	52864	2394	2058	18	567	8586	302	221	8188	1238	717
	SBW	52%	7.8	31343	797	2293	29	950	3044	46	213	2684	132	907
maximum	IB	75%	35	63438	2445	3195	177	627	37361	440	2413	7303	565	579
	MB	77%	7.6	81945	3017	2868	23	751	11945	641	617	9171	1049	719
	OB	76%	6.6	68526	7654	3881	32	835	10783	920	291	12873	3605	1713
	SBW	75%	26	52515	2438	4432	69	2292	8461	122	416	5232	211	1880
minimum	IB	15%	0.2	5497	48	493	2.8	18	534	15	134	372	20	23
	MB	21%	0.7	7487	16	525	0.2	36	481	9	126	207	18	9
	OB	21%	1.0	35096	628	1101	4	247	5442	118	143	4520	444	344
	SBW	24%	2.5	10681	130	667	8	229	118	7	2	881	47	274

Given the large reservoir of phosphorus in the bay sediments and the apparent mobilization of this phosphorus to the water column, remedial action to control internal phosphorus loading appears necessary to expedite improvements in water quality associated with reductions in point and nonpoint loading. However, continued excessive loading from the watershed may offset gains made by internal load reduction, possibly within just a year or two, and certainly over an extended time period. Therefore, it is unlikely that control of internal loading alone will result in long-term reduction in nutrient concentrations in St. Albans Bay. Further reduction of watershed loading and a lowering of the internal loading of phosphorus both appear necessary to meet water quality goals for St. Albans Bay. The watershed reductions should precede any treatment in the bay to maximize the lifespan of the treatment.

4.0 MANAGEMENT GOALS AND OBJECTIVES

The primary goal of this project is to restore St. Albans Bay so that its condition is consistent with the use statements in the Vermont Water Quality Standards (February 9, 2006):

"In all waters, total phosphorus loadings shall be limited so that they will not contribute to the acceleration of eutrophication or the stimulation of the growth of aquatic biota in a manner that prevents the full support of uses."

Algal blooms represent one of the major impairments and thus the reduction of algal blooms ties into this goal. Reducing the internal loading of phosphorus is necessary to attain this goal. However, the watershed is the ultimate source of nutrients to the bay, and strategies aimed at reducing the internal phosphorus load must be implemented in combination with watershed load reductions in order to improve water quality in St. Albans Bay. The total phosphorus concentration goal for St. Albans Bay is 0.017 mg/l which is significantly lower than concentrations currently observed in the bay.

Summarized in outline form, the goals and objectives are as follows:

Overall Goal: Shift the bay from its current eutrophic state to a mesotrophic condition.
  Objective: Attain the designated uses of the water body: habitat, aesthetics, and recreation opportunities for swimming, boating, and fishing.

Immediate Goal: Minimize the frequency and severity of algae blooms.
  Objective: Reduce internal loading in order to minimize algal growth.

Strategies for achieving these goals and objectives will be addressed in the subsequent sections.

5.0 MANAGEMENT OPTIONS

Minimize the Frequency and Severity of Algae Blooms

Excessive algal growth can become a serious nuisance in aquatic habitats. Two growth forms are most troublesome in lakes:

- Free-floating microscopic cells, colonies or filaments, called phytoplankton, that discolor the water and sometimes form green scum on the surface of the waterbody. These algae come from a variety of algal groups, including blue-greens, greens, diatoms, goldens, euglenoids and dinoflagellates, although the blue-greens tend to be the most troublesome group as a consequence of high densities, taste and odor issues, and possible toxins. Numerous toxic cyanobacteria genera have been identified during late summer algal blooms in St. Albans Bay.
- Mats of filamentous algae associated with sediments and weed beds, but often floating to the surface after a critical density is attained. These are most often green algae of the orders *Cladophorales* or *Zygnematales*, or blue-green algae (more properly cyanobacteria) of the order *Oscillatoriales*. These are objectionable to some bathers and can have ecological consequences as well. This is not currently a problem in St. Albans Bay. However, should water clarity improve, there is the potential for attached benthic algae and other macrophytes (vascular plants) to become more abundant, possibly to the point of impairing uses.

Algae reproduce mainly through cell division, although resting cysts are an important mechanism for surviving unfavorable periods. When growth conditions are ideal (warm, lighted, nutrient-rich), algae multiply rapidly and reach very high densities (blooms) in a matter of weeks. Many algae contribute to taste and odor problems at high densities and the decay of algal blooms can lead to oxygen depression.

The factors that control the abundance of algae form the basis for attempts to manage and limit them. Light and nutrients are the primary needs for algae growth. Where algal densities, non-algal turbidity, or shading by rooted plants do not create a light limitation, the quantity of algae in a lake is usually directly related to the concentration of the essential plant nutrient in least supply. In many cases this element is phosphorus. Even when phosphorus is not currently the limiting nutrient, it is usually more appropriate to create a phosphorus limitation than to control other nutrients. Algae management techniques (Table 2) such as dyes, artificial circulation and selective plantings seek to establish light limitation, while methods such as aeration, dilution and

TABLE 2. MANAGEMENT OPTIONS FOR THE CONTROL OF ALGAE

OPTION	MODE OF ACTION	ADVANTAGES	DISADVANTAGES	APPLICATION TO ST. ALBANS BAY
Physical Controls				
1) Hypolimnetic aeration or oxygenation	♦ Addition of air or oxygen at varying depth provides oxic conditions ♦ May maintain or break stratification ♦ Can also withdraw water, oxygenate, then replace	♦ Oxic conditions promote binding/sedimentation of phosphorus ♦ Counteraction of anoxia improves habitat for fish/invertebrates ♦ Build-up of dissolved iron, manganese, ammonia and phosphorus reduced	♦ May disrupt thermal layers important to fish community ♦ May promote supersaturation with gases harmful to fish	No. Widespread anoxia has not been observed in St. Albans Bay.
2) Circulation and destratification	♦ Use of water or air to keep water in motion ♦ Intended to prevent or break stratification ♦ Generally driven by mechanical or pneumatic force	♦ Reduces surface build-up of algal scums ♦ Promotes uniform appearance ♦ Counteraction of anoxia improves habitat for fish/invertebrates ♦ Can eliminate localized problems without obvious impact on whole lake	♦ May spread localized impacts ♦ May increase oxygen demand at greater depths ♦ May promote downstream impacts	The Bay does not stratify but there may be some benefit to mixing algal cells down in the water column. See Management Options for discussion.
3) Dilution and flushing	♦ Addition of water of better quality can dilute nutrients ♦ Addition of water of similar or poorer quality flushes system to minimize algal build-up ♦ May have continuous or periodic additions	♦ Dilution reduces nutrient concentrations without altering load ♦ Flushing minimizes detention; response to pollutants may be reduced	♦ Diverts water from other uses ♦ Flushing may wash desirable zooplankton from lake ♦ Use of poorer quality water increases loads ♦ Possible downstream impacts	No. The option is not practical given the size of St. Albans Bay and the lack of hydraulic control.

OPTION	MODE OF ACTION	ADVANTAGES	DISADVANTAGES	APPLICATION TO ST. ALBANS BAY
Physical Controls				
4) Drawdown	♦ Lowering of water over autumn period allows oxidation, desiccation and compaction of sediments ♦ Duration of exposure and degree of dewatering of exposed areas are important ♦ Algae are affected mainly by reduction in available nutrients.	♦ May reduce available nutrients or nutrient ratios, affecting algal biomass and composition ♦ Opportunity for shoreline clean-up/structure repair ♦ Flood control utility ♦ May provide rooted plant control as well	♦ Possible impacts on contiguous emergent wetlands ♦ Possible effects on overwintering reptiles or amphibians ♦ Possible impairment of well production ♦ Reduction in potential water supply and fire fighting capacity ♦ Alteration of downstream flows ♦ Possible overwinter water level variation ♦ May result in greater nutrient availability if flushing inadequate	No. Drawing down Lake Champlain basin sufficiently to uncover St. Albans Bay sediments is not technically feasible.
5) Dredging	♦ Sediment is physically removed by wet or dry excavation, with deposition in a containment area for dewatering ♦ Dredging can be applied on a limited basis, but is most often a major restructuring of a severely impacted system ♦ Nutrient reserves are removed and algal growth can be limited by nutrient availability	♦ Can control algae if internal recycling is main nutrient source ♦ Increases water depth ♦ Can reduce pollutant reserves ♦ Can reduce sediment oxygen demand ♦ Can improve spawning habitat for many fish species ♦ Allows complete renovation of aquatic ecosystem	♦ Temporarily removes benthic invertebrates ♦ May create turbidity ♦ May eliminate fish community (complete dry dredging only) ♦ Possible impacts from containment area discharge ♦ Possible impacts from dredged material disposal ♦ Interference with recreation or other uses during dredging	Option under consideration, however costs are high. See Management Options for discussion.

OPTION	MODE OF ACTION	ADVANTAGES	DISADVANTAGES	APPLICATION TO ST. ALBANS BAY
Physical Controls				
5.a) "Dry" excavation	♦ Lake drained or lowered to maximum extent practical ♦ Target material dried to maximum extent possible ♦ Conventional excavation equipment used to remove sediments	♦ Tends to facilitate a very thorough effort ♦ May allow drying of sediments prior to removal ♦ Allows use of less specialized equipment	♦ Eliminates most aquatic biota unless a portion left undrained ♦ Eliminates lake use during dredging	No. Drawing down Lake Champlain basin sufficiently to uncover St. Albans Bay sediments is not technically feasible.
5.b) "Wet" excavation	♦ Lake level may be lowered, but sediments not substantially exposed ♦ Draglines, bucket dredges, or long-reach backhoes used to remove sediment	♦ Requires least preparation time or effort, tends to be least cost dredging approach ♦ May allow use of easily acquired equipment ♦ May preserve aquatic biota	♦ Usually creates extreme turbidity ♦ Tends to result in sediment deposition in surrounding area ♦ Normally requires intermediate containment area to dry sediments prior to hauling ♦ May cause severe disruption of ecological function ♦ Usually eliminates most lake uses during dredging	No. Bay is too large to access with shore based equipment.
5.c) Hydraulic removal	♦ Lake level not reduced ♦ Suction or cutterhead dredges create slurry which is hydraulically pumped to containment area ♦ Slurry is dewatered; sediment retained, water discharged	♦ Creates minimal turbidity and impact on biota ♦ Can allow some lake uses during dredging ♦ Allows removal with limited access or shoreline disturbance	♦ Often leaves some sediment behind ♦ Cannot handle coarse or debris-laden materials ♦ Requires sophisticated and more expensive containment area ♦ Requires overflow discharge from containment area	Under consideration. Hydraulic dredging would be the most feasible of methods should dredging be chosen as a management option.

OPTION	MODE OF ACTION	ADVANTAGES	DISADVANTAGES	APPLICATION TO ST. ALBANS BAY
Physical Controls				
6) Light-limiting dyes and surface covers	♦ Creates light limitation	♦ Creates light limit on algal growth without high turbidity or great depth ♦ May achieve some control of rooted plants as well	♦ May cause thermal stratification in shallow ponds ♦ May facilitate anoxia at sediment interface with water	No. These options would be impractical given the size and flushing rate of St. Albans Bay.
6.a) Dyes	♦ Water-soluble dye is mixed with lake water, thereby limiting light penetration and inhibiting algal growth ♦ Dyes remain in solution until washed out of system.	♦ Produces appealing color ♦ Creates illusion of greater depth	♦ May not control surface bloom-forming species ♦ May not control growth of shallow water algal mats	No. See above.
6.b) Surface covers	♦ Opaque sheet material applied to water surface	♦ Minimizes atmospheric and wildlife pollutant inputs	♦ Minimizes atmospheric gas exchange ♦ Limits recreational use	No. See above.
7) Mechanical removal	♦ Filtering of pumped water for water supply purposes ♦ Collection of floating scums or mats with booms, nets, or other devices ♦ Continuous or multiple applications per year usually needed	♦ Algae and associated nutrients can be removed from system ♦ Surface collection can apply on an "as needed" basis ♦ May remove floating debris ♦ Collected algae dry to minimal volume	♦ Filtration requires high backwash and sludge handling capability for use with high algal densities ♦ Labor intensive unless a mechanized system applied, in which case it is capital intensive ♦ Many algal forms not amenable to collection by net or boom ♦ Possible impacts on non-targeted aquatic life	No. St. Albans Bay is too large for this to be practical. This option would not significantly decrease phosphorus levels in the Bay so would require ongoing operation.

OPTION	MODE OF ACTION	ADVANTAGES	DISADVANTAGES	APPLICATION TO ST. ALBANS BAY
Physical Controls				
8) Selective withdrawal	♦ Discharge of bottom water which may contain (or be susceptible to) low oxygen and higher nutrient levels ♦ Intake of water from low algae layer to maximize supply quality ♦ May be pumped or utilize passive head differential	♦ Removes targeted water from lake efficiently ♦ Complements other techniques such as drawdown or aeration ♦ May prevent anoxia and phosphorus build up in bottom water ♦ May remove initial phase of algal blooms which start in deep water ♦ May create coldwater conditions downstream	♦ Possible downstream impacts of poor water quality ♦ May eliminate colder thermal layer important to certain fish ♦ May promote mixing of some remaining poor quality bottom water with surface waters ♦ May cause unintended drawdown if inflows do not match withdrawal	No. Not applicable to St. Albans Bay due to a lack of hydraulic control and a lack of documented anoxic conditions in St. Albans Bay.

OPTION	MODE OF ACTION	ADVANTAGES	DISADVANTAGES	APPLICATION TO ST. ALBANS BAY
Chemical controls				
9) Algaecides	♦ Liquid or pelletized algaecides applied to target area ♦ Algae killed by direct toxicity or metabolic interference ♦ Typically requires application at least once/yr, often more frequently	♦ Rapid elimination of algae from water column , normally with increased water clarity ♦ May result in net movement of nutrients to bottom of lake	♦ Possible toxicity to non-target areas or species of plants/animals ♦ Restrictions on water use for varying time after treatment ♦ Increased oxygen demand and possible toxicity from decaying algae ♦ Possible recycling of nutrients, allowing other growths	No. This is considered a short-term measure that will not address internal loading of phosphorus and may increase internal cycling. Chemical application may also harm non-target plant and animals in the benthic zone as well as in the Black Creek wetland.
9.a) Forms of copper	♦ Contact algaecide ♦ Cellular toxicant, suggested disruption of photosynthesis, nitrogen metabolism, and membrane transport ♦ Applied as wide variety of liquid or granular formulations, often in conjunction with chelators, polymers, surfactants or herbicides	♦ Effective and rapid control of many algae species ♦ Approved for use in most water supplies	♦ Toxic to aquatic fauna as a function of concentration, formulation, temperature, pH, and ambient water chemistry ♦ Ineffective at colder temperatures ♦ Copper ion persistent; accumulates in sediments or moves downstream ♦ Certain green and blue-green nuisance species are resistant to copper ♦ Lysing of cells releases cellular contents (including nutrients and toxins) into water column	No. Copper sulfide treatments were unsuccessful in the past (1968-1970s).

OPTION	MODE OF ACTION	ADVANTAGES	DISADVANTAGES	APPLICATION TO ST. ALBANS BAY
Chemical controls				
9.b) Forms of endothall (7-oxabicyclo [2.2.1] heptane-2,3-dicarboxylic acid)	♦ Contact algaecide ♦ Membrane-active chemical which inhibits protein synthesis ♦ Causes structural deterioration ♦ Applied as liquid or granules, usually as hydrothol formulation for algae control	♦ Moderate control of thick algal mats, used where copper is ineffective ♦ Limited toxicity to fish at recommended dosages ♦ Rapid action	♦ Non-selective in treated area ♦ Toxic to aquatic fauna (varying degrees by formulation) ♦ Time delays on use for water supply, agriculture and recreation ♦ Safety hazards for applicators	No. See above.
9.c) Forms of diquat (6,7-dihydropyrido [1,2-2',1'-c] pyrazinediium dibromide)	♦ Contact algaecide ♦ Absorbed directly by cells ♦ Strong oxidant; disrupts most cellular functions ♦ Applied as a liquid, sometimes in conjunction with copper	♦ Moderate control of thick algal mats, used where copper alone is ineffective ♦ Limited toxicity to fish at recommended dosages ♦ Rapid action	♦ Non-selective in treated area ♦ Toxic to zooplankton at recommended dosage ♦ Inactivated by suspended particles; ineffective in muddy waters ♦ Time delays on use for water supply, agriculture and recreation	No. See above.

OPTION	MODE OF ACTION	ADVANTAGES	DISADVANTAGES	APPLICATION TO ST. ALBANS BAY
Chemical controls				
10) Phosphorus inactivation	♦ Typically salts of aluminum, iron or calcium are added to the lake, as liquid or powder ♦ Phosphorus in the treated water column is complexed and settled to the bottom of the lake ♦ Phosphorus in upper sediment layer is complexed, reducing release from sediment ♦ Permanence of binding varies by binder in relation to redox potential and pH ♦ Potential for use on inlet streams as well	♦ Can provide rapid, major decrease in phosphorus concentration in water column ♦ Can minimize release of phosphorus from sediment ♦ May remove other nutrients and contaminants as well as phosphorus ♦ Flexible with regard to depth of application and speed of improvement	♦ Possible toxicity to fish and invertebrates, especially by aluminum at low pH ♦ Possible release of phosphorus under anoxia or extreme pH ♦ May cause fluctuations in water chemistry, especially pH, during treatment ♦ Possible resuspension of floc in shallow areas with extreme turbulence ♦ Adds to bottom sediment, but typically an insignificant amount	Option under consideration. See Management Options for discussion.
11) Sediment oxidation	♦ Addition of oxidants, binders and pH adjustors oxidizes sediment ♦ Binding of phosphorus is enhanced ♦ Denitrification is stimulated	♦ Can reduce phosphorus supply to algae ♦ Can alter N:P ratios in water column ♦ May decrease sediment oxygen demand	♦ Possible impacts on benthic biota ♦ Longevity of effects not well known ♦ Possible source of nitrogen for blue-green algae	No. Anoxic conditions have not been observed in the water column of St. Albans Bay. The oxic state of surficial sediments is variable.

OPTION	MODE OF ACTION	ADVANTAGES	DISADVANTAGES	APPLICATION TO ST. ALBANS BAY
Chemical controls				
12) Settling agents	♦ Closely aligned with phosphorus inactivation, but can be used to reduce algae directly too ♦ Lime, alum or polymers applied, usually as a liquid or slurry ♦ Creates a floc with algae and other suspended particles ♦ Floc settles to bottom of lake ♦ Re-application typically necessary at least once/yr	♦ Removes algae and increases water clarity without lysing most cells ♦ Reduces nutrient recycling if floc sufficient ♦ Removes non-algal particles as well as algae ♦ May reduce dissolved phosphorus levels at the same time	♦ Possible impacts on aquatic fauna ♦ Possible fluctuations in water chemistry during treatment ♦ Resuspension of floc possible in shallow, well-mixed waters ♦ Promotes increased sediment accumulation	Alum is being considered to inactivate phosphorus. It may serve a dual role as a settling agent.
13) Selective nutrient addition	♦ Ratio of nutrients changed by additions of selected nutrients ♦ Addition of non-limiting nutrients can change composition of algal community ♦ Processes such as settling and grazing can then reduce algal biomass (productivity can actually increase, but standing crop can decline)	♦ Can reduce algal levels where control of limiting nutrient not feasible ♦ Can promote non-nuisance forms of algae ♦ Can improve productivity of system without increased standing crop of algae	♦ May result in greater algal abundance through uncertain biological response ♦ May require frequent application to maintain desired ratios ♦ Possible downstream effects	No. This option is contrary to ongoing watershed management initiatives and would require vast increases in nitrogen loading with uncertain effects on the algal community. It would not change internal phosphorus loading to the bay.

OPTION	MODE OF ACTION	ADVANTAGES	DISADVANTAGES	APPLICATION TO ST. ALBANS BAY
Chemical controls				
14) Management for nutrient input reduction	♦ Includes wide range of watershed and lake edge activities intended to eliminate nutrient sources or reduce delivery to lake ♦ Can involve adding doses alum and alum into tributaries ♦ Essential component of algal control strategy where internal recycling is not the dominant nutrient source, and desired even where internal recycling is important	♦ Acts against the original source of algal nutrition ♦ Decreased effective loading of nutrients to lake ♦ Generally most cost effective over long term ♦ Facilitates ecosystem management approach which considers more than just algal control	♦ May involve considerable lag time before improvement observed?? ♦ May not be sufficient to achieve goals without some form of in-lake management ♦ Reduction of overall system fertility may impact fisheries ♦ May cause shift in nutrient ratios which favor less desirable species ♦ May cost more in the short term, as source management is generally more involved than one or a few treatments of symptoms of eutrophication	Option is essential to ultimate success of management actions to control internal phosphorus loading but will not be discussed in detail as a part of this investigation. See Management Options for further discussion.
Biological Controls				
15) Enhanced grazing	♦ Manipulation of biological components of system to achieve grazing control over algae ♦ Typically involves alteration of fish community to promote growth of large herbivorous zooplankton, or stocking with phytophagous fish	♦ May increase water clarity by changes in algal biomass or cell size distribution without reduction of nutrient levels ♦ Can convert unwanted biomass into desirable form (fish) ♦ Harnesses natural processes to produce desired conditions	♦ May involve introduction of exotic species ♦ Effects may not be controllable or lasting ♦ May foster shifts in algal composition to even less desirable forms	No. Modification of aquatic ecosystem in St. Albans Bay would be impractical given connection to the rest of Lake Champlain.

OPTION	MODE OF ACTION	ADVANTAGES	DISADVANTAGES	APPLICATION TO ST. ALBANS BAY
Chemical controls				
15.a) Herbivorous fish	♦ Stocking of fish which eat algae	♦ Converts algae directly into potentially harvestable fish ♦ Grazing pressure can be adjusted through stocking rate	♦ Typically requires introduction of non-native species ♦ Difficult to control over long term ♦ Smaller algal forms may be benefited and bloom	No. See above.
15.b) Herbivorous zooplankton	♦ Reduction in planktivorous fish to promote grazing pressure by zooplankton ♦ May involve stocking piscivores or removing planktivores ♦ May also involve stocking zooplankton or establishing refugia	♦ Converts algae indirectly into harvestable fish ♦ Zooplankton community response to increasing algae can be rapid ♦ May be accomplished without introduction of non-native species ♦ Generally compatible with most fishery management goals	♦ Highly variable response expected; temporal and spatial variability may be problematic ♦ Requires careful monitoring and management action on 1-5 yr basis ♦ May involve non-native species introduction(s) ♦ Larger or toxic algal forms may be benefited and bloom	No. See above.
16) Bottom-feeding fish removal	♦ Removes fish which browse among bottom deposits, releasing nutrients to the water column by physical agitation and excretion	♦ Reduces turbidity and nutrient additions from this source ♦ May restructure fish community in more desirable manner	♦ Targeted fish species are difficult to eradicate or control ♦ Reduction in fish populations valued by some lake users (human and non-human)	No. Modification of aquatic ecosystem in St. Albans Bay would be impractical given connection to the rest of Lake Champlain.

OPTION	MODE OF ACTION	ADVANTAGES	DISADVANTAGES	APPLICATION TO ST. ALBANS BAY
Chemical controls				
17) Fungal/bacterial/viral pathogens	♦ Addition of inoculum to initiate attack on algal cells	♦ May create lakewide "epidemic" and reduction of algal biomass ♦ May provide sustained control for several years ♦ Can be highly specific to algal group or genera	♦ Largely experimental approach at this time ♦ Considerable uncertainty of results ♦ May promote resistant forms with high nuisance potential ♦ May cause high oxygen demand or release of toxins by lysed algal cells ♦ Effects on non-target organisms uncertain	No. Modification of aquatic ecosystem in St. Albans Bay would be impractical given connection to the rest of Lake Champlain.
18) Competition and allelopathy	♦ Plants may tie up sufficient nutrients to limit algal growth ♦ Plants may create a light limitation on algal growth ♦ Chemical inhibition of algae may occur through substances released by other organisms	♦ Harnesses power of natural biological interactions ♦ May provide responsive and prolonged control	♦ Some algal forms appear resistant ♦ Use of plants may lead to problems with vascular plants ♦ Use of plant material may cause depression of oxygen levels	No. Modification of aquatic ecosystem in St. Albans Bay would be impractical given connection to the rest of Lake Champlain.
18.a) Plantings for nutrient control	♦ Plant growths of sufficient density may limit algal access to nutrients ♦ Plants can exude allelopathic substances which inhibit algal growth	♦ Productivity and associated habitat value can remain high without algal blooms ♦ Portable plant "pods" , floating islands, or other structures can be managed to limit interference with recreation and provide habitat ♦ Wetland cells in or adjacent to the lake can minimize nutrient inputs	♦ Vascular plants may achieve nuisance densities ♦ There will be a water depth limitation on rooted plants but not algae ♦ Vascular plant senescence may release nutrients and cause algal blooms ♦ The switch from algae to vascular plant domination of a lake may cause unexpected or undesirable changes in lake ecology, especially energy flow	No. Modification of aquatic ecosystem in St. Albans Bay would be impractical given connection to the rest of Lake Champlain.

OPTION	MODE OF ACTION	ADVANTAGES	DISADVANTAGES	APPLICATION TO ST. ALBANS BAY
Chemical controls				
18.b) Plantings for light control	♦ Plant species with floating leaves can shade out many algal growths at elevated densities	♦ Vascular plants can be more easily harvested than most algae ♦ Many floating species provide valuable waterfowl food	♦ At the necessary density, the floating plants will be a recreational nuisance ♦ Low surface mixing and atmospheric contact will promote anoxia near the sediment	No. Much of St. Albans Bay is too deep and too high energy an environment for floating leaved plants.
18.c) Addition of barley straw	♦ Input of barely straw can set off a series of chemical reactions which limit algal growth ♦ Release of allelopathic chemicals can kill algae ♦ Release of humic substances can bind phosphorus	♦ Materials and application are relatively inexpensive ♦ Decline in algal abundance is more gradual than with algaecides, limiting oxygen demand and the release of cell contents	♦ Success appears linked to uncertain and potentially uncontrollable water chemistry factors ♦ Depression of oxygen levels may result ♦ Water chemistry may be altered in other ways unsuitable for non-target organisms ♦ Some forms of algae may be resistant and could benefit from the treatment	No. St Albans Bay is too large to treat with this method.

flushing, drawdown, dredging, phosphorus inactivation, and selective withdrawal are used to reduce nutrient availability.

Table 2 provides an outline of virtually all of the algal management techniques available for use at this time. Many do not address internal loading as a means to control algae, which is the focus of this project, but the table does convey a clear impression of the range of options that could be applied. These techniques take advantage of algal ecology and supplement or counteract the forces involved in algal losses or growth, respectively.

Natural algal losses occur through settling, consumption by grazers, and cellular death. Accelerated loss processes are the focus of techniques such as settling agents, biomanipulation (either grazing enhancement or addition of bacteria or viruses which kill algal cells), algaecide applications, and mechanical removal. Unfortunately, algae are remarkably adaptable as a community, and none of the techniques derived from loss processes are effective on the complete range of algae that commonly occur. Many blue-greens are buoyant and resist settling. Nuisance forms of green algae (*Chlorococcales* and *Cladophorales*) and certain blue-green algae (especially *Aphanizomenon*) are highly resistant to copper, the most common algaecide, and are also largely grazer-resistant. Only very dense algal mats can be feasibly harvested, and then with some difficulty.

Selective nutrient addition may provide an ecologically complex solution in some cases. By altering the ratio of nutrients, types of algae may be favored which are more amenable to other control techniques, most notably increased grazing pressure and settling rates. Productivity may not be reduced, but more efficient processing of primary production may lead to lower standing crop (biomass). Although sound in theory (Kilham 1971, Tilman 1982), this approach has rarely been applied in practical lake management efforts. It is important to recognize, however, that a productive lake need not suffer algal blooms (high biomass) if the algae and their consumers can be manipulated to increase the rate at which the energy represented by the algae can be passed to other trophic levels.

Filamentous algal mats have a distinctive ecology and are difficult to control. Mats typically form at the sediment-water interface or in association with rooted plant beds, taking nutrition from decay processes in that zone and surviving at low light levels through high densities of photosynthetic pigments. As mat density increases, photosynthetic gases are often trapped, and the mat may float upward and expand. Grazing control of mats is negligible, settling is not a major force, and harvesting is not practical in most cases. Algaecides are often ineffective once a dense mat has formed, as contact between algae and algaecide is limited. Prevention of mat formation through sediment removal or treatment (phosphorus inactivation or early algaecide application) is preferable to dealing with extensive, well-formed mats.

Use of algaecides typically releases taste and odor agents, toxins, other organic compounds, and nutrients into the water column, where they may remain a problem. Techniques that prevent the formation of high algal biomass are preferable to those that counteract the effects of a phytoplankton bloom or extensive mats. Where algaecides are to be used, maximum effectiveness is achieved by tracking algal composition and abundance and by timing treatments to coincide with the exponential growth phase of target algae.

Given the above caveats, the specific circumstances in St. Albans Bay, and a focus on internal load as a key factor in continued algal blooms, many of the listed techniques warrant no further consideration. Details of the options most applicable to St. Albans Bay are provided in narrative form below.

5.1 Circulation and Destratification

Circulation affects mixing and the uniformity of lake conditions. Thermal stratification and features of lake morphometry such as coves create stagnant zones that may be subject to loss of oxygen, accumulation of sediment, or algal blooms. Artificial circulation minimizes stagnation and can eliminate thermal stratification or prevent its formation. Movement of water by pneumatic or mechanical means is normally used to create the desired circulation pattern in shallow (<20 ft) lakes. Surface aerators, bottom diffusers, and water pumps have all been used to mix small ponds and shallow lakes. The effect is largely cosmetic in many instances; algae are simply mixed more evenly in the available volume of water. Some blue-greens, however, can be disrupted by this movement, potentially leading to a shift in algal composition.

Stratification is broken or prevented in deeper lakes through the injection of compressed air into lake water from a diffuser at the lake bottom. The rising column of bubbles, if sufficiently powered, will produce lake-wide mixing at a rate that eliminates temperature differences between top and bottom waters. The use of air as the mixing force also provides some oxygenation of the water, but the efficiency and magnitude of this transfer are generally low. In some instances, wind driven pumps have been used to move water. This mixing process will result in higher oxygen levels in the bottom waters, but is considered separately from hypolimnetic oxygenation, in which stratification is not broken.

Algal blooms are sometimes controlled by mixing and destratification, possibly through one or more of the following processes:

- In light-limited algal communities, mixing to the lake's bottom will increase a cell's time in darkness, leading to reduced net photosynthesis.
- Introduction of dissolved oxygen to the lake bottom may inhibit phosphorus release from sediments, curtailing this internal nutrient source.
- Rapid circulation and contact of water with the atmosphere, as well as the introduction of carbon dioxide-rich bottom water during the initial period of mixing, can increase the carbon dioxide content of water and lower pH, leading to a shift from blue-green algae to less noxious green algae.
- When zooplankton that consume algae are mixed throughout the water column, they are less vulnerable to visually feeding fish. If more zooplankton survive, their consumption of algal cells may also increase.

However, the lack of documented thermal stratification or anoxia in St. Albans Bay essentially limits the relevant processes addressed by this technique to the first one listed.

Mixing results have varied greatly from case to case. In most instances, problems with low dissolved oxygen have been solved. When destratification is properly used in a water supply reservoir, problems with iron and manganese can be eliminated. Where major temperature differences from top to bottom have been eliminated through the summer, algal blooms seem to be reduced. In other cases, phosphorus and turbidity have increased and transparency has decreased. Circulation should at least prevent the formation of distinct surface scums, although total algal biomass may not be reduced. Systems that bring deep water to the surface can be inexpensive, but unless enough water is moved to prevent anoxia near the sediment-water interface, the quality of water that is brought to the surface may cause deterioration of surface conditions. Systems that pump surface water to the bottom may improve the oxygen level near the bottom, but may also cause unfavorable circulation patterns and deterioration of surface conditions.

Failure to achieve the desired objective may be caused by lake chemistry or equipment. A lake that receives high nutrient loads from external sources is unlikely to respond acceptably. Underdesign of the mixing system is the major equipment-related cause of failure for this technique. Lorenzen and Fast (1977) suggested that an air flow of about

1.3 ft^3/min per acre of lake surface is required to maintain mixing and oxygen within the lake. Mixing may be induced with lower air flow through careful engineering design, but undersizing of mixing devices remains a common problem. There are many engineering details to be considered in the design of a circulation system, and knowledge of site conditions is essential; professionally designed systems are likely to be much more efficient than amateur efforts.

Because St. Albans Bay does not thermally stratify, destratification is not applicable. The only benefit to be derived from the application of this technology is therefore mixing of algal cells throughout the water column and potentially below the photic zone. This will not change the concentration of phosphorus in the bay nor will it change the amount of phosphorus entering the bay from the watershed or being released from the sediments. Natural mixing likely occurs frequently in the bay due to the shallow depth, lack of thermal stratification and orientation to strong prevailing southerly winds. Mixing for purposes of disrupting scum forming blooms is unlikely to be helpful on a bay-wide basis due to the large surface area of the bay and the mixing regime already present. An evaluation of mixing at selected nearshore locations, where surface scums concentrate and accumulate, is planned for summer of 2007 using Solarbee ™ technology. The ability of these devices to adequately disrupt algal blooms should be apparent after this deployment however, as stated above, they will not change water column P concentrations.

5.2 Dredging

The release of algae-stimulating nutrients from lake sediments can be controlled by removing layers of enriched materials. This may produce significantly lower in-lake nutrient concentrations and less algal production, assuming that there has been adequate diversion or treatment of incoming nutrient, organic and sediment loads from external sources. Even where incoming nutrient loads are high, dredging can reduce benthic mat formation and related problems with filamentous green and blue-green algae, as these forms rely on resting stages in the sediment to initiate summer populations and may initially depend on nutrient-rich substrates for nutrition. Although recolonization would be expected to be rapid, changes in algal composition can result.

Sediment removal to retard nutrient release can be effective. An example is provided by Lake Trummen in Sweden (Andersson 1988) where the upper 3.3 feet of sediments were extremely rich in nutrients. This layer was removed and the total phosphorus concentration in the lake dropped sharply and remained fairly stable for at least 18 years. Phytoplankton production was reduced as a result.

Algal abundance also decreased and water clarity increased in Hills Pond in Massachusetts after all soft sediment was removed and a storm water treatment wetland was installed in 1994 (Wagner 1996). Dredging of 6-acre Bulloughs Pond in Massachusetts in 1993 resulted in abatement of thick green algal mats for over a decade, despite continued high nutrient loading from urban runoff (Wagner, personal observation). These mats had previously begun as spring bottom growths and then floated to the surface in mid-summer.

While removing the entire nutrient-rich layer of sediment can control algae, dredging is most frequently done to deepen a lake, remove accumulations of toxic substances, or to remove and control macrophytes. Algal control benefits are largely ancillary in these cases. The expense of complete soft sediment removal and the more pressing need for watershed management in most cases are the primary reasons that dredging is not used more often for algal control.

Dredging would remove the phosphorus-laden sediments in St. Albans Bay and hence reduce internal phosphorus loading. However, the costs would be especially high due to shear quantity of sediment involved in such a large body of water. The work of Druschel et al. (2005) clearly indicates that available phosphorus levels are highest in the upper 2 cm of sediments, but the reduction by about 50% at a depth of 8 cm still results in a very high available phosphorus concentration (Figure 4). Dredging to a depth of much more than 8 cm is necessary, but we do not know exactly how deep dredging must go to reach low phosphorus sediment. Assuming just 0.1 m (about 4 inches), the quantity of sediment that would have to be removed from just the inner bay would be approximately 260,000 cubic meters (approximately 338,000 cubic yards) (Table 3). For all three areas of the bay and the Black Creek wetland, the total sediment volume would be approximately 1.4 million cubic meters (1.8 million cubic yards). Values climb proportionally as the dredging depth increases (Table 3).

Additionally, the higher phosphorus levels near the sediment surface may not be a result of higher concentration of phosphorus in deposited materials, but rather may result from upward migration of phosphorus from anoxic zones in the sediments to the more oxic zone near the sediment-water interface. Dissolved iron moving with the dissolved phosphorus will combine with that phosphorus in the presence of oxygen and form a precipitate, enriching the surficial layer with both elements. Dredging to a point where a layer with a lower phosphorus concentration is encountered may be only a temporary solution if there is substantial phosphorus even deeper in the sediment. The solution is normally to remove all “soft” sediment (organic material) that may be subject to the upward migration of iron and phosphorus under anoxic conditions. This could require dredging many more vertical feet of sediment in St. Albans Bay at considerable additional cost.

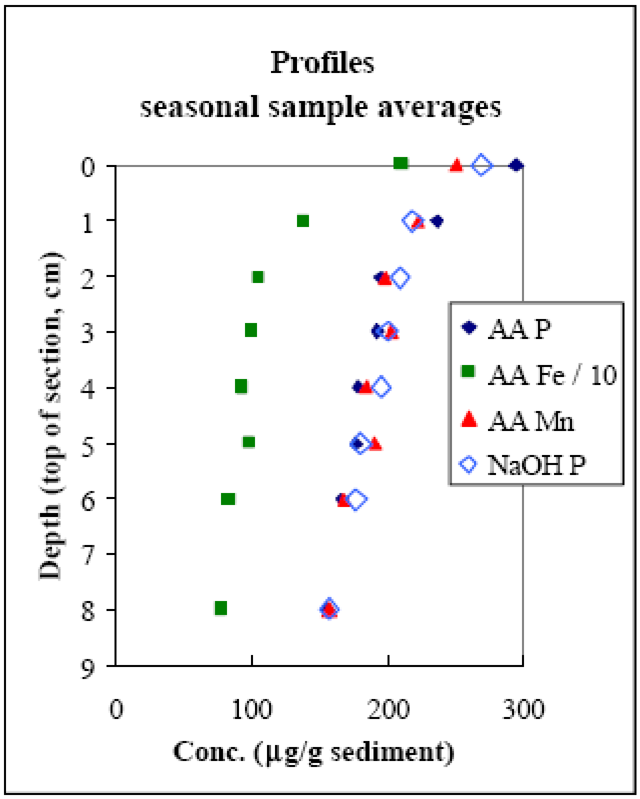

Figure 4. Profile of sediment averages for phosphorus, manganese and iron for seasonal site from extractions

Table 3. Dredging volume calculations.

Lake Section	IB	MB	OB	SBW
Target Area (ac)	643	1075	1743	44
Target Area (m2)	2592742	4334677	7028226	178000
	Dredging 4 inches of Sediment			
Target Depth of Sediment to be Dredged (ft)	0.3	0.3	0.3	0.3
Target Depth of Sediment to be Dredged (m)	0.1	0.1	0.1	0.1
Volume of Sediment to be Dredged (m3)	259274	433468	702823	17800
Volume of Sediment to be Dredged (cy)	338977	566719	918875	23272
	Dredging 1 ft of Sediment			
Target Depth of Sediment to be Dredged (ft)	1.0	1.0	1.0	1.0
Target Depth of Sediment to be Dredged (m)	0.3	0.3	0.3	0.3
Volume of Sediment to be Dredged (m3)	777823	1300403	2108468	53400
Volume of Sediment to be Dredged (cy)	1016931	1700157	2756626	69816
	Dredging 3.3 ft of Sediment			
Target Depth of Sediment to be Dredged (ft)	3.3	3.3	3.3	3.3
Target Depth of Sediment to be Dredged (m)	1.0	1.0	1.0	1.0
Volume of Sediment to be Dredged (m3)	2592742	4334677	7028226	178000
Volume of Sediment to be Dredged (cy)	3389770	5667189	9188754	232720

Without changes in the watershed load, or dredging to a depth where continued migration of phosphorus from deeper sediments is not an issue, the lifespan of a dredging program for algal control may be limited to a few years.

5.3 Phosphorus Inactivation

The release of phosphorus stored in lake sediments can be so extensive in some lakes and reservoirs that algal blooms persist even after incoming phosphorus has been significantly lowered. Phosphorus precipitation by chemical complexing removes phosphorus from the water column and can control algal abundance until the phosphorus supply is replenished. Phosphorus inactivation typically involves some amount of phosphorus precipitation, but aims to achieve long-term control of phosphorus release from lake sediments by adding as much phosphorus binder to the lake as possible within the limits dictated by environmental safety. It is essentially an "anti-fertilizer" addition. This technique is most effective after nutrient loading from the watershed is sufficiently reduced, as it acts only on existing phosphorus reserves, not new ones added post-treatment.

Aluminum has been widely used for phosphorus inactivation, mostly as aluminum sulfate and sometimes as sodium aluminate, as it binds phosphorus well under a wide range of conditions, including anoxia. However, concentrations of reactive aluminum (AL^{+3}) are strongly influenced by pH, and levels in excess of 50 µg/l may be toxic to aquatic fauna. A pH of between 6.0 and 8.0 virtually ensures that the 50 µg/l limit will not be reached, but aluminum sulfate addition can reduce the pH well below a pH of 6.0 in poorly buffered waters. In such cases sodium aluminate, which raises the pH, has been successfully used in combination with aluminum sulfate (Cooke et al. 1993b). It is also possible to add buffering agents to the lake prior to aluminum sulfate addition, such as lime and sodium hydroxide. Other chemicals that have been successfully employed to bind phosphorus include calcium hydroxide and ferric chloride; the former tends to raise the pH and the latter lowers the pH slightly. Ferric sulfate has also been applied, and lowers the pH substantially.

In practice, aluminum sulfate (often called alum) is added to the water and colloidal aggregates of aluminum hydroxide are formed. These aggregates rapidly grow into a visible, brownish to greenish white floc, a precipitate that settles to the sediments in a few hours to a few days, carrying sorbed phosphorus and bits of organic and inorganic particulate matter in the floc. After the floc settles to the sediment surface, the water will be very clear. If enough alum is added, a layer of several inches of aluminum hydroxide will cover the sediments, react with phosphorus in the surficial sediment, and significantly retard the release of phosphorus into the water column as an internal load. In lakes where sufficient reduction of external nutrient loading has occurred, this can create a phosphorus limitation on algal growth. Once P is inactivated by the aluminum it remains complexed (Cooke et al 2005) and is likely unavailable for algal uptake and growth even if the aluminum floc is temporarily resuspended by wave action. It is possible that in the immediate near shore wave zone, the floc will be sufficiently disrupted to expose untreated sediment and associated P to wave action. This would not occur over the majority of the treatment area but it could reduce the treatment effectiveness somewhat.

Good candidate lakes for this procedure are those that have had external nutrient loads reduced to an acceptable level and have been shown, during a diagnostic-feasibility study, to have a high internal phosphorus load (release from sediment). High alkalinity is also desirable to provide buffering capacity. Highly flushed impoundments are usually not good candidates because of an inability to limit phosphorus inputs.

Treatment of lakes with low doses of alum may effectively remove phosphorus from the water column, but may be inadequate to provide long-term control of phosphorus release from lake sediments.

Nutrient inactivation has received increasing attention over the last decade as long lasting results have been demonstrated in multiple projects, especially those employing aluminum compounds (Welch and Cooke 1999). Annabessacook Lake in Maine suffered algal blooms for 40 years prior to the 1978 treatment with aluminum sulfate and sodium aluminate (Cooke et al. 1993a). Low buffering capacity necessitated the use of sodium aluminate. A 65% decrease in internal phosphorus loading was achieved, blue-green algae blooms were eliminated, and conditions have remained much improved for nearly 30 years. Similarly impressive results have been obtained in two other Maine lakes using the two aluminum compounds together (Connor and Martin 1989a).

Kezar Lake was treated with aluminum sulfate and sodium aluminate in 1984 after a wastewater treatment facility discharge was diverted from the lake. Both algal blooms and oxygen demand were depressed for several years, but began to rise more quickly than expected (Connor and Martin 1989a, 1989b). Additional controls on external loads (wetland treatment of inflow) reversed this trend and conditions have remained markedly improved over pre-treatment conditions for over 20 years. No adverse impacts on fish or benthic fauna have been observed.

Aluminum sulfate and sodium aluminate were again employed with great success at Lake Morey, Vermont (Smeltzer 1990, Smeltzer et al. 1999). A pretreatment average spring total phosphorus concentration of 37 µg/l was reduced to 9 µg/l after treatment in late spring of 1987. Although epilimnetic phosphorus levels have varied since then, the pretreatment levels have not yet been approached. Hypolimnetic phosphorus concentrations have not exceeded 50 µg/l. Oxygen levels increased below the epilimnion, with as much as 10 vertical feet of suitable trout habitat reclaimed. Some adverse effects of the treatment on benthic invertebrates and yellow perch were found to be temporary phenomena following treatment; conditions have been excellent through 20 years of post-treatment monitoring.

Delevan Lake, Wisconsin (1800 acres), similar in size to St. Albans Bay (inner bay and wetlands 700 acres, whole bay 3500 acres) was treated with alum in 1991 (Robertson et al 2001). The treatment was very successful for the first few years despite being dosed with approximately 1/3 the alum that subsequent calculations suggested was optimal. However, a large influx of watershed phosphorus in 1993, negated the effect of the alum treatment and internal loading again became an important component of the nutrient budget of the lake.

Phosphorus inactivation has also been successful in some shallow wind-mixed lakes (Welch et al. 1988, Welch and Schrieve 1994), but has been unsuccessful in cases where the external loads have not been controlled prior to inactivation (Barko et al. 1990, Welch and Cooke 1999). Cooke et al (2005) state that effective internal P control should lead directly to water quality improvement in polymictic or frequently mixed lakes. St. Albans Bay functions like a polymictic lake.

Successful dose rates have ranged from 3 to 30 g Al/m^3 on a volumetric basis, or 15 to 50 g Al/m^2 on an areal basis, the latter being the more important value with regard to inactivation of sediment phosphorus and prevention of internal recycling from that source. Jar tests are often used to evaluate the appropriate dose, but a newer method (Rydin and Welch 1998, 1999) suggests that the dose should be determined from the measured available sediment phosphorus content. A ratio of aluminum sulfate to sodium aluminate of 2:1 is expected to cause no change in system pH where buffering is needed. Slight shifts in that ratio (about 1.6:1 to 2.2:1) can control pH at a slightly higher or lower level if so desired. Maintenance of the ambient pH is an appropriate goal, unless the pH is especially high as a consequence of excessive algal photosynthesis or especially low due to acid precipitation impacts.

Aluminum sulfate can be applied near the thermocline depth in deep lakes, providing a precautionary refuge for fish and zooplankton that could be affected by dissolved reactive aluminum. Application methods include modified harvesting equipment, outfitted pontoon boats, and barges specially designed for this purpose.

Success has also been achieved with calcium (Babin et al. 1989, Murphy et al. 1990) and iron (Walker et al. 1989) salts, but it has become clear that aluminum provides the greatest long-term binding potential for phosphorus inactivation (Harper et al. 1999). The use of calcium would seem to be appropriate in high pH lakes (pH >10 SU), and provides natural phosphorus inactivation in certain hardwater lakes. Iron seems to be most useful in conjunction with aeration systems; as noted previously, iron releases phosphorus under anoxic conditions. Aluminum salts can be used successfully in any of these cases, however, and alum tends to be the chemical of choice unless toxicity becomes a problem.

Longevity of alum treatments has generally been excellent where external inputs of phosphorus to the system have been controlled (Payne et al. 1991). As a general rule, inactivation with aluminum can be expected to last for at least three flushing cycles, with much longer effectiveness where external loading has been controlled. A review of 21 well-studied phosphorus inactivation treatments using aluminum (Welch and Cooke 1999) indicates that longevity of effects is typically 15 years or more for dimictic (summer stratified) lakes and about 10 years for shallow, polymictic (unstratified) lakes.

Despite major successes, addition of aluminum salts to lakes does have the potential for serious negative impacts, and care must therefore be exercised with regard to dosage and buffering capacity. The potential for toxicity problems is directly related to the alkalinity and pH of the lake water. In soft (low alkalinity) water, only very small doses of alum can be added before alkalinity is exhausted and the pH falls below 6.0. At pH 6.0 and below, $Al(OH)_2$ and dissolved elemental aluminum (Al^{+3}) become the dominant forms. Both can be toxic to aquatic species. Well-buffered, hard water lakes can handle much higher alum doses without fear of creating toxic forms of aluminum. Soft water lakes must be buffered, either with sodium aluminate or other compounds, to prevent the undesirable pH shift while allowing enough $Al(OH)_3$ to be formed to control phosphorus release.

Although pH depression is usually the major threat, elevated pH from over-buffering can also cause problems. Hamblin Pond in Massachusetts was treated with alum and sodium aluminate in 1995, after three years of pre-treatment study that demonstrated both the importance of internal phosphorus loading and limited buffering capacity (Wagner 2001). A number of problems arose during the treatment, resulting in an overdose of sodium aluminate throughout the lake. The pH rose from about 6.3 to over 9.0, and a fish kill resulted. Despite an increase in summer water transparency from about 4 ft to nearly 20 ft and a gain of 10 vertical feet of suitable coldwater fish habitat, the fish kill has fostered some sentiments against this technique among Massachusetts permitting agencies.

A similar situation occurred in Lake Pocotopaug, CT in 2000 (ENSR 2001), but the pH was never measured at greater than 8.0. Where sodium aluminate is used, it appears that the pH must be maintained below about 7.5 to prevent any toxicity, and the ratio of alum to aluminate should be tightly controlled. The ratio of 2 parts alum to 1 part aluminate did not harm fish in an assay (ENSR 2001), and re-treatment of Lake Pocotopaug in May/June 2001 resulted in no fishkill even with treatment at the surface at a dose of 10 mg Al/L.

Other potential adverse impacts relate to the spread of macrophytes and changes in water chemistry after addition of aluminum compounds. Although the sharp increase in water transparency is viewed as desirable in most cases, it may allow an existing rooted plant infestation to spread into new areas or deeper water. The addition of sulfates to the lake in an aluminum sulfate treatment may foster chemical reactions that disrupt the iron cycle and associated natural phosphorus binding capacity. Aluminum toxicity to humans has created substantial public controversy with respect to treatment of lakes with aluminum, but concerns have not been supported by the bulk of scientific investigations (Krishnan 1988, Harriger and Steelhammer 1989). A detailed knowledge of lake chemistry is necessary to understand and apply phosphorus inactivation as an algal control technique.

For St. Albans Bay, with very high available phosphorus levels in the surficial sediment, the amount of aluminum needed to effectively inactivate that phosphorus is quite high. Tests should be run to determine the level of inactivation at various potential doses, but assuming an aluminum level just equal to the phosphorus concentration yields an estimate of 45,580 kg (100,277 lb) of aluminum needed to inactivate phosphorus in the upper 4 cm of sediment in the inner bay alone (Table 4). Treating the entire bay and the Black Creek wetland would require 374,000 kg (822,000 lbs) of aluminum. In the absence of testing to determine the response curve of available sediment phosphorus to aluminum additions, we apply a general rule of an aluminum dose ten times the phosphorus quantity to be inactivated; this would raise the above dose estimates by an order of magnitude, at considerably greater cost.

Table 4. Estimation of minimum aluminum dose needed for St. Albans Bay.

Alum Dosing Calculations	**Putting in Al at 1 X P mass**			
Lake	**IB**	**MB**	**OB**	**SBW**
Mean Available Sediment P (mg/kg DW)	293.0	273.0	585.0	979.0
Target Depth of Sediment to be Treated (cm)	4.0	4.0	4.0	4.0
Volume of Sediment to be Treated per m2 (m3)	0.040	0.040	0.040	0.040
Specific Gravity of Sediment	1.50	1.50	1.50	1.50
Mass of Sediment to be Treated (kg/m2)	60.0	60.0	60.0	60.0
Mass of P to be Treated (g/m2)	17.6	16.4	35.1	58.7
Target Area (ac)	643	1075	1743	44
Target Area (m2)	2592742	4334677	7028226	177419
Stoich. Ratio (ratio of Al to P in treatment)	1	1	1	1
Aluminum Load				
Dose (kg/area)	45580	71002	246691	10422
Dose (lb/area)	100277	156204	542720	22928

Although St. Albans Bay continues to have a large external phosphorus load, nutrient inactivation has the potential to greatly reduce algal densities in St. Albans Bay at least in the short term because internal loading appears to contribute bio-available phosphorus. Treatment with alum and aluminate would be appropriate, with proper pre-treatment testing for dose verification and impact avoidance. However, treatment should be preceded by substantial cuts in the watershed phosphorus load to prolong the life of the treatment and reduce the potential for blooms in the future.

5.4 Management for Nutrient Input Reduction

Techniques which belong to this category are largely watershed management methods. Reduction of phosphorus loads from the watershed is an essential part of the ultimate success of this project. A blueprint for management of these nutrients to meet loading targets for the bay is presented in Gaddis (2006). Meeting the watershed loading goals by implementation of the recommendations outlined by Gaddis appears necessary to reduce the prevalence of algal blooms and insure the success of treating internal loading of phosphorus and extending the life of the treatment.

The boundary of watershed versus in-lake techniques blurs, however, when a part of the lake is used to create a detention, filtration or wetland treatment area to improve the quality of incoming water. The distinction is also less clear when tributary or storm water runoff is treated with phosphorus inactivators prior to discharge to a lake, where the floc then settles (Harper et al. 1999). However, it is important to recognize that at least some portion of the pollutant attenuation function ascribed to watersheds can be assumed by part of the lake, with proper planning and implementation. It is believed that dosing of one or more of the tributary streams with aluminum to inactivate phosphorus may provide a good interim step in the reduction of watershed loading of phosphorus. In addition to inactivating a large proportion of the incoming phosphorus from a portion of the watershed, the floc formed will settle in the bay and inhibit migration of phosphorus from the bay sediments to the water column. The initial dose would be lower than what would be required to inactivate P in bay sediments so not all internal recycling would be eliminated through tributary dosing but the potential to make a difference at considerably lower cost is evident. Tributary dosing may bridge the time gap until watershed P control is fully implemented allowing in lake treatment to proceed earlier, start the sediment inactivation in the bay and be a demonstration of what bay conditions would look like if spring and summer P loads from the watershed were reduced.

5.4.1 Dosing of tributary streams

In addition to watershed management techniques applied on land at some distance from the lake, treatment of phosphorus flowing through tributaries could be used to reduce nutrient inputs. The approach is simple enough, although there are many details to consider. One or more aluminum compounds are added to a tributary at a rate determined by expected phosphorus concentration and flow, both of which could be measured in near real time in a sophisticated system. Additions are usually in liquid form, but powdered aluminum compounds can also be used. Mixing enhances efficiency, so some form of mechanical or pneumatic mixing system is usually applied as well, although if additions are during turbulent flow conditions, further mixing may not be necessary. Additions can be made at any time, but are most commonly in response to increasing flows indicative of storm events that tend to account for most of the phosphorus loading in tributary systems where aluminum is applied.

When dosing a tributary, the dose is largely determined by the phosphorus concentration and flow, facilitating inactivation of as much incoming phosphorus as possible. As the completeness of reactions is favored by aluminum concentrations ten to twenty times the phosphorus level, there should be extra aluminum remaining that can inactivate receiving sediments downstream of the treatment zone. While the dose would be much lower for St. Albans Bay by this method than for a direct inactivation of sediment phosphorus, this approach limits incoming phosphorus availability while still providing some control of internal load. Over time, the load from existing sediments may be appreciably reduced, but this would not be expected within the first year or two of treatment.

Working from existing data, flows in the spring from the combined Jewett and Stevens Brooks averages about 46 cfs, which is very close to the assumed watershed yield for this time of year of about 2 cfs per square mile of watershed. Mill Creek has roughly the same area and flow as the combined Jewett and Stevens Brooks, called Black Creek at the entrance to St. Albans Bay. Remaining drainage contributes on the order of 13 cfs based on a yield of 2 cfs/sq.mi. of drainage area. These values can all be converted into metric units, cubic meters per day, for further calculation (Table 5). The average concentration of phosphorus in the tributaries during spring is between 0.1 and 0.2 mg/L, which can be multiplied by the daily flow to get a daily load of phosphorus that must be inactivated. Working from a rule of thumb of a 20:1 ratio for aluminum to phosphorus, the needed dose of aluminum can be calculated (Table 5); 224 to 448 kg Al/day for either the combined Jewett/Stevens Brooks system or Mill Creek. Remaining drainage area, without larger, defined tributaries, would be harder to treat and would most likely be ignored in this approach.

While the instream inactivation system could be run continuously, it would probably only be run for part of the spring, either on higher flow days in response to precipitation or for several weeks later in the spring, to ensure that the water reaching the bay shortly before the normally drier summer months will be of the lowest fertility possible. It would be reasonable to assume operation over about 45 days each spring. Not all phosphorus would be inactivated, but a large enough portion would be affected to significantly lower the tributary load to the bay during the critical spring and summer period while beginning the sediment inactivation process.

Dosing stations to accomplish instream inactivation usually consist of storage tanks and instrumentation to detect flow changes and initiate delivery of the aluminum compounds. Since the pH of incoming water may vary considerably over time during spring, buffering the aluminum addition is considered essential to avoid toxicity issues. Piping for delivery from the storage area to the stream may involve a simple gravity feed

Table 5. Aluminum dose necessary to inactivate incoming spring phosphorus loads.

Stream System	Stevens	Jewett	Mill	Other
Watershed Area (km2)	39	20	59	17
Watershed Area (mi2)	15.1	7.8	22.9	6.6
Assumed water yield (cfs/mi2)	2.0	2.0	2.0	2.0
Flow based on above (cfs)	30.2	15.5	45.7	13.2
Flow (m3/day)	73978	37938	111916	32247
	Inflow P at 0.1 mg/L			
P conc. (mg/L)	0.1	0.1	0.1	0.1
P load (kg/day)	7.4	3.8	11.2	3.2
Desired Al:P ratio	20	20	20	20
Necessary Al load/day (kg/day)	148	76	224	64
Necessary Al load/day (lb/day)	326	167	492	142
	Inflow P at 0.2 mg/L			
P conc. (mg/L)	0.2	0.2	0.2	0.2
P load (kg/day)	14.8	7.6	22.4	6.4
Desired Al:P ratio	20	20	20	20
Necessary Al load/day (kg/day)	296	152	448	129
Necessary Al load/day (lb/day)	651	334	985	284

or a pumping system. Mixing, if needed, can be mechanical or pneumatic, but will require a power source. The exact location of the dosing station is flexible; for St. Albans Bay, the primary question is whether Jewett and Stevens Brook should be treated separately upstream of the Black Creek wetland or downstream at the bridge just before this tributary enters the lake. As the wetland has considerable phosphorus in its sediments, inactivation in that area is desirable. However, potential impacts to wetland biota and cost savings by having a single dosing station at the downstream bridge suggest that the downstream location would be preferable as a test location if this approach was considered worthy of a trial.

Drawbacks to this approach include inadequate dosing to inactivate sediment phosphorus in the bay over the initial year of treatment, as the predicted dose extrapolated from Table 5 for a 45 day period for Jewett Brook, Stevens Brook and Mill Creek (20,160 kg) is less than half the needed dose to inactivate sediment phosphorus at only a 1:1 ratio (aluminum:phosphorus) in just the inner bay (45,580 kg, Table 4). If internal loading is truly the primary source of summer phosphorus fueling algal blooms, this approach will take several years to have an effect, if any. Also, while this approach will reduce the input of phosphorus during a key time of year for bay nutrient loading, it is less philosophically desirable than attacking the input problem closer to the sources, upstream in the watershed. However, use of such an approach as an interim measure while watershed controls are implemented appears justifiable, if those watershed inputs are directly responsible for summer algal blooms.

6.0 MANAGEMENT COSTS

The most applicable in-lake techniques for reducing internal loading of phosphorus to St. Albans Bay are circulation, dredging, internal phosphorus inactivation, and nutrient inactivation in the tributary streams. However, the ongoing watershed nutrient load to the bay appears high enough to negate the benefits of either dredging or inactivation with aluminum within several years. The circulation approach may work to some degree without addressing internal or external loading directly, but evidence for the ability of this approach to overcome intense loading in a shallow system is limited. The planned test of circulation equipment in summer of 2007 will provide useful data in this regard. The tributary inactivation system could counteract ongoing watershed loads to a large degree, and may eventually provide significant relief from internal loading, but is not a substitute for proper watershed controls and may not adequately control the internal load. No one solution appears adequate to the circumstances of St. Albans Bay and its watershed. Further consideration will need to reflect costs and environmental impacts. In this section we address the costs.

6.1 Circulation

Circulation may assist in reducing the late summer algal bloom problem by uniformly mixing the algal cells throughout the water column. This may reduce the prevalence of surface scums and potentially limit the growth of some nuisance algae by moving cells down in the water column below the photic or lighted zone for more time each day. Monitoring of the effects of Solarbees™ scheduled to be installed in summer of 2007 will help determine whether circulation assists in reducing algal blooms. This project is separate from that effort and no attempt has been made to generate a cost estimate for any target area within St. Albans Bay.

6.2 Dredging

Dredging the existing soft sediment in St. Albans Bay should provide the desired internal nutrient loading reduction and subsequent algal bloom abatement for at least as long as it takes to accumulate enough phosphorus-rich new sediment to support the internal recycling again. It also will remove a lot of resting stages of algae and seeds and other propagules of rooted plants, shifting community composition in a desirable but not necessarily permanent manner. It would also increase water depth, usually considered a benefit in most aquatic management programs. However, this would be a very large project with substantial financial and disposal area needs.

Assuming a soft sediment depth of only 10 cm (a very thin layer, and not necessarily sufficient in this case) over the 2.6 million m^2 of the inner bay, the amount to be removed would be over 259,000 m^3, or almost 339,000 cubic yards (cy), the units with which dredging contractors work. At a cost range of $5-15/cy, the cost for dredging just 0.1 m (4 inches) over the area of the inner bay would be between $1.7 and 5 million (Table 6). Extended to the entire area of interest (Inner Bay, Middle Bay, Outer Bay and Stevens Brook Wetlands), the cost could approach $28 million.

Table 6. Estimation of potential dredging costs.

Lake Section	IB	MB	OB	SBW
Target Area (ac)	643	1075	1743	44
Target Area (m2)	2592742	4334677	7028226	178000
	Dredging 4 inches of Sediment			
Volume of Sediment to be Dredged (cy)	338977	566719	918875	23272
Low Cost per CY	$5	$5	$5	$5
Low Dredging Cost	$1,694,885	$2,833,595	$4,594,377	$116,360
High Cost per CY	$15	$15	$15	$15
High Dredging Cost	$5,084,655	$8,500,784	$13,783,132	$349,080
	Dredging 1 ft of Sediment			
Volume of Sediment to be Dredged (cy)	1016931	1700157	2756626	69816
Low Cost per CY	$5	$5	$5	$5
Low Dredging Cost	$5,084,655	$8,500,784	$13,783,132	$349,080
High Cost per CY	$15	$15	$15	$15
High Dredging Cost	$15,253,965	$25,502,352	$41,349,395	$1,047,240
	Dredging 3.3 ft of Sediment			
Volume of Sediment to be Dredged (cy)	3389770	5667189	9188754	232720
Low Cost per CY	$5	$5	$5	$5
Low Dredging Cost	$16,948,850	$28,335,947	$45,943,772	$1,163,600
High Cost per CY	$15	$15	$15	$15
High Dredging Cost	$50,846,550	$85,007,841	$137,831,317	$3,490,800

It is likely that more sediment will have to be removed; the phosphorus level 8 cm deep into the sediment is only 50% of that in the upper 2 cm, but this is still a high value and represents a potential for continued high internal loading. Table 6 provides a range of estimates for removing as much as 1 m (3.3. ft) of sediment from each of the four potential target areas. The costs certainly appear prohibitive.

Even if funding was somehow secured, it is rare to dredge a water body the size and depth of St. Albans Bay. The considered range of sediment volumes is 2-20 million cy. The project would have to be a hydraulic dredging operation, as there is no way to drain the bay and conventional wet dredging would be very messy, with likely impacts to a greater portion of the lake than are likely to be acceptable. The dredged material would not be appropriate for disposal in the lake due to its high water and P content and the likelihood that it would create an enormous turbidity plume as well as water quality problems at the disposal site. Dredged material would need to bepumped as a slurry to fields in the area for drying and disposal, the volume would actually increase by about 25% initially. After drying, the volume could shrink as much as 50%, but even then the dredged material could cover an area of 620 acres with 1 to 10 ft of material (depending on whether 2 or 20 million cy were dredged). Generally, material is not piled more than 3 ft high even when dry, so it is more likely that about 2000 acres of disposal area would be needed if the larger quantity of sediment was to be dredged. No costs for land acquisition or use have been included in this cost estimate. Environmental impacts will be considered separately, but it is hard to envision dredging going forward on a cost basis alone.

6.3 Phosphorus Inactivation

Phosphorus inactivation is another viable option given the technical feasibility of inactivating the large quantity of reactive phosphorus in the sediment. Aluminum salts are the most commonly used chemical to inactivate phosphorus in lake sediments. Costs of phosphorus inactivation vary depending on the extent of application and quantity of aluminum needed (Tables 7 and 8). The use of aluminum sulfate (alum) only, as opposed to a buffered combination of alum and sodium aluminate (aluminate), does not make a large difference in cost, although there may be some storage and application costs not inherently captured in this analysis when both alum and aluminate are used. The most critical factor is the ratio of aluminum to phosphorus, which would never be less than 1:1 and is usually at least 10:1. The cost of inactivating the top 4 cm of sediments of only the inner bay would cost approximately $500,000 to $3.6 million, depending upon the dose and assuming that estimates for design, permitting and monitoring are appropriate for Vermont. Extending the treatment to all four areas of interest, the cost of inactivating the phosphorus in the top 4 cm of sediments ranges from approximately $3.6 million to $29 million, depending on the dose. While this is expensive, it is far less expensive than dredging.

Even with in lake phosphorus inactivation, the continued influx of phosphorus from the watershed is expected to continue to cause algal blooms. It is possible that during the normally low flow late summer-fall period, when most blooms occur, that the internal load is the primary source of phosphorus for fueling those blooms. It is also possible that occasional summer storms load enough phosphorus to the bay to support algal blooms. In either case, continued loading from the watershed will replace the inactivated phosphorus in the surficial sediments over a period of years, minimizing the longevity of the treatment effect and lowering the benefit:cost ratio in any economic analysis of management of St. Albans Bay with aluminum.

While alum treatments in shallow systems have provided as much as a decade of relief with limited management activity in the watershed (Welch et al. 1988, Cooke et al. 1993b, Welch and Cooke 1999), the known loading from the St. Albans Bay watershed is among the higher quantities encountered in our experience. Even a decade of relief may not be considered economically favorable at the estimated cost, but we suspect that benefits would last even less than a decade. Reduction of tributary phosphorus loads prior to treatment therefore will be necessary to protect any investment made in phosphorus inactivation in the bay.

Table 7. Estimate of probable cost for sediment phosphorus inactivation at a 1:1 ratio of Al to P.

Cost of In-Lake Alum Application for the top 4 cm of sediment at a 1:1 Al to P ratio	**Inner Bay**	**Middle Bay**	**Outer Bay**	**Black Creek Wetland**	
Mean Available Sediment P* (mg/kg Dry Weight)	293	273	585	979	
Mass of P to be Treated (g/m2)	18	16	35	59	
Target Area (ac)	643	1075	1743	44	
Aluminum Application for Alum only					
Dose (gal alum) @ 11.1 lb/gal and 4.4% Al	205317	319829	1111219	46944	
Aluminum Application for Alum + Aluminate					
Dose (gal alum) @ 11.1 lb/gal and 4.4% Al	89822	139918	486134	20537	
Dose (gal aluminate) @ 12.1 lb/gal and 10.38% Al	44911	69959	243067	10269	
Unit Cost					
Alum (per gallon)	$1.00	$1.00	$1.00	$1.00	
Aluminate (per gallon)	$2.75	$2.75	$2.75	$2.75	
Chemical Cost					
Alum only	$205,317	$319,829	$1,111,219	$46,944	
Alum + Aluminate	$213,326	$332,305	$1,154,567	$48,775	
Labor Cost					
Planning/Design (assumes 5-10 samples and no access issues)	$6,000	$6,000	$6,000	$6,000	
Permitting (assumes no overly complicated/protracted process)	$10,000	$10,000	$10,000	$10,000	
Application (assumes 10,000 gal/day)	$102,659	$159,914	$555,610	$23,472	
Mobilization/Contingencies (assumes 1 day/25 ac)	$128,600	$215,000	$348,600	$8,800	
Monitoring (assumes 1 day/trtmt day + 12 days + 20% for lab costs)	$39,038	$52,779	$147,746	$20,033	
					All Sections
Cost Summary (alum only)	$491,614	$763,523	$2,179,176	$115,250	**$3,549,562**
Cost Summary (alum + aluminate)	$499,623	$775,999	$2,222,523	$117,081	**$3,615,226**

*Source: Druschel et al. 2005

Table 8 Estimate of probable cost for sediment phosphorus inactivation at a 10:1 ratio of A1 to P

Cost of In-Lake Alum Application for the top 4 cm of sediment at a 10:1 Al to P ratio	**Inner Bay**	**Middle Bay**	**Outer Bay**	**Black Creek Wetland**	
Mean Available Sediment P* (mg/kg Dry Weight)	293	273	585	979	
Mass of P to be Treated (g/m2)	18	16	35	59	
Target Area (ac)	643	1075	1743	44	
Aluminum Application for Alum only					
Dose (gal alum) @ 11.1 lb/gal and 4.4% Al	2,053,171	3,198,289	11,112,195	469,442	
Aluminum Application for Alum + Aluminate					
Dose (gal alum) @ 11.1 lb/gal and 4.4% Al	898,216	1,399,180	4,861,336	205,370	
Dose (gal aluminate) @ 12.1 lb/gal and 10.38% Al	449,108	699,590	2,430,668	102,685	
Unit Cost					
Alum (per gallon)	$1.00	$1.00	$1.00	$1.00	
Aluminate (per gallon)	$2.75	$2.75	$2.75	$2.75	
Chemical Cost					
Alum only	$2,053,171	$3,198,289	$11,112,195	$469,442	
Alum + Aluminate	$2,133,264	$3,323,052	$11,545,674	$487,755	
Labor Cost					
Planning/Design (assumes 5-10 samples and no access issues)	$6,000	$6,000	$6,000	$6,000	
Permitting (assumes no overly complicated/protracted process)	$10,000	$10,000	$10,000	$10,000	
Application (assumes 10,000 gal/day)	$1,026,586	$1,599,145	$5,556,097	$234,721	
Mobilization/Contingencies (assumes 1 day/25 ac)	$128,600	$215,000	$348,600	$8,800	
Monitoring (assumes 1 day/trtmt day + 12 days + 20% for lab costs)	$260,781	$398,195	$1,347,863	$70,733	
					All Sections
Cost Summary (alum only)	$3,485,138	$5,426,628	$18,380,756	$799,696	**$28,092,218**
Cost Summary (alum + aluminate)	$3,565,230	$5,551,391	$18,814,235	$818,009	**$28,748,865**

*Source: Druschel et al. 2005

6.3.1 Dosing of tributary streams

In addition to source control or as an interim measure as source control is being implemented, watershed phosphorus loads from tributaries can also be inactivated by injecting doses of alum into the tributaries. This dosing can be done seasonally as the greatest phosphorus loads appears to occur during spring storm events. Following from the analysis of expected spring flows and phosphorus loads, cost estimates for inactivating the incoming load from each of the tributaries can be generated (Tables 9 and 10). Given additional variables of days of treatment, location of the dosing station, and the concentration of phosphorus that must be treated, this is less straightforward than estimating sediment phosphorus inactivation. If we assume an inflow concentration of 0.1 mg P/L and individual dosing stations for each of three major tributaries and one additional site to handle all other inflows, costs range from $291 to $1048 per day of treatment, depending on the tributary and whether aluminate is used to buffer the alum.

Of course, one additional dosing station can't practically handle all inflows not in the three major tributaries, and no treatment of these minor inflows is likely. It also seems likely that Stevens and Jewett Brooks would be treated after their confluence and conversion to Black Creek, most likely at the bridge at the inlet to the bay. In that case, there would be only two dosing stations, one on Black Creek and one on Mill Creek. The treatment need for the combined Stevens and Jewett Brooks (Black Creek) is coincidentally the same as for Mill Creek, so the cost for treating Mill Creek is applicable to Black Creek at up to $1048/day.

For 45 days of treatment during spring, with dosing stations at the Black Creek bridge and on Mill Creek, the cost for establishing these stations (design, permitting, construction) and treating for phosphorus inputs at 0.1 mg/L would be roughly $515,000 (twice the Mill Creek total cost in Table 9). If it is assumed that the inflow phosphorus level was 0.2 mg/L, the corresponding cost would be $700,000 (twice the Mill Creek value in Table 10); only the chemical costs increase when assuming a larger inflow concentration, and assumption of the higher value seems appropriate in this case, as additional inactivation of bay sediment phosphorus is desired.

The tributary treatment option is considerably less expensive than sediment inactivation in the bay, but does not supply the same level of internal load control. It does limit incoming phosphorus, and may represent a viable interim measure for controlling the load from the watershed. Yet it may not prevent blooms in the short-term if internal loading is the primary phosphorus source, and the benefit:cost ratio does not seem favorable as a long-term management approach for watershed loading.

Table 9. Estimate of Probable Cost for Tributary Phosphorus Inactivation at an Inflow of 0.1 mg/L (Note that annual costs after the first year may be somewhat lower as some cost of the dosing station components as well as some design and permitting costs may not be incurred every year).

Inflow Treatment Assuming a P Concentration of 0.1 mg/L	**Tributary**			
	Stevens	**Jewett**	**Mill**	**Other**
Daily Aluminum Dose				
Necessary Al load/day (kg/day)	148	76	224	64
Necessary Al load/day (lb/day)	326	167	492	142
Gal Alum Needed (alum only trtmt)	666	342	1008	291
Gal Alum Needed (alum + aluminate trtmt)	292	150	441	127
Gal Aluminate Needed (alum + aluminate trtmt)	146	75	221	64
Daily Chemical Cost Estimate				
Cost for treatment with Alum only	$666	$342	$1,008	$291
Cost for treatment with Alum + Aluminate	$692	$355	$1,048	$302
Total Cost				
Days of Treatment	45	45	45	45
Chemical Cost for Alum only	$29,991	$15,380	$45,371	$13,073
Chemical Cost for Alum + Aluminate	$31,161	$15,980	$47,141	$13,583
Dosing Station	$80,000	$80,000	$80,000	$80,000
Additional Design and Permitting Cost	$40,000	$40,000	$40,000	$40,000
Additional Monitoring Cost	$45,000	$45,000	$45,000	$45,000
Total Cost of One-Year Program	$226,152	$196,360	$257,513	$191,656

Table 10. Estimate of Probable Cost for Tributary Phosphorus Inactivation at an Inflow of 0.2 mg/L (Note that annual costs after the first year may be somewhat lower as some cost of the dosing station components as well as some design and permitting costs may not be incurred every year).

Alum Inflow Treatment Assuming a P Concentration of 0.2 mg/L	Tributary			
	Stevens	**Jewett**	**Mill**	**Other**
Daily Aluminum Dose				
Necessary Al load/day (kg/day)	296	152	448	129
Necessary Al load/day (lb/day)	651	334	985	284
Gal Alum Needed (alum only trtmt)	1333	684	2017	581
Gal Alum Needed (alum + aluminate trtmt)	583	299	882	254
Gal Aluminate Needed (alum + aluminate trtmt)	292	150	441	127
Daily Chemical Cost Estimate				
Cost for treatment with Alum only	$1,333	$684	$2,017	$581
Cost for treatment with Alum + Aluminate	$1,385	$710	$2,095	$604
Total Cost				
Days of Treatment	45	45	45	45
Chemical Cost for Alum only	$59,983	$30,760	$90,743	$26,146
Chemical Cost for Alum + Aluminate	$62,322	$31,960	$94,283	$27,166
Dosing Station	$80,000	$80,000	$80,000	$80,000
Additional Design and Permitting Cost	$40,000	$40,000	$40,000	$40,000
Additional Monitoring Cost	$45,000	$45,000	$45,000	$45,000
Total Cost of One-Year Program	$287,305	$227,720	$350,025	$218,312

7.0 MANAGEMENT IMPACTS

Each of the viable techniques for managing internal loading in St. Albans Bay would have quantifiable impacts on the bay if implemented. The intended impact is beneficial; phosphorus concentrations should be reduced and algal growth should be limited, preventing algal blooms that currently impair uses of the bay. Associated negative impacts are also possible, however, and must be considered in any management analysis.

The intended benefit of any method to directly lower internal loading would be a lower summer phosphorus concentration in St. Albans Bay. Concentrations of phosphorus in the bay between 1992 and 2004 have averaged about 30 ug/L (Druschel et al. 2005), although values approaching 50 ug/L are not uncommon and values as high as 80 ug/L have been recorded. Values going into the summer period tend to average closer to 20 ug/L, suggesting that internal recycling could be providing a net increase of 10 ug/L by very rough assessment. More detailed analysis of just one year of data, from 1992 (Smeltzer et al. 1994), suggests that high phosphorus levels are found throughout the year in the Black Creek wetland and the portion of the inner bay closest to Black Creek, but that values in most of the inner and middle bays begin the summer not far above the targeted value of 17 ug/L, climbing during the summer to values as high as 60 ug/L. This would seem to suggest that as much as 40 ug/L is being supplied by internal loading. If internal loading can be eliminated, phosphorus could decline to a level close to the desired target. At the range of phosphorus values encountered, and decrease should translate into less frequent and/or severe algal blooms.

7.1 Mixing

The mixing approach may foster greater phosphorus binding and lower concentrations somewhat, but the primary mechanism at work is physical disruption, with altered light regime and possible chemical shifts that do not favor blue-green algae. Mixing in St. Albans Bay is not a technique for the reduction of internal phosphorus loading. Whether or not a few mixing devices can make a difference in one part of a large bay with strong wind influence remains to be seen. There does not appear to be any obvious negative impact from such mixing. However, there is the potential for the same algae to be present, just not as a concentrated surface scum. This would improve aesthetics, but might encourage use of those areas by swimmers, with resultant potentially greater exposure to the toxins associated with algal blooms in the past in St. Albans Bay. A thorough monitoring program should accompany the planned mixing test in summer of 2007.

7.2 Dredging

Dredging is a very effective way to reduce internal loading, as well as to remove reserves of seeds, spores and other resting stages of plants. It has the potential to dramatically change the character of the lake bottom, usually for the better from the perspective of the majority of lake uses and users. Removal of all soft sediment would essentially eliminate the internal load, although complete removal of all soft sediment is rarely accomplished with hydraulic dredging. As any dredging in St. Albans Bay is likely to be by hydraulic means, assumption of a 90% reduction in internal load might be appropriate, putting dredging on par with inactivation by aluminum in this regard.

However, the cost of dredging suggests that only a small portion of the soft sediment would be likely to be removed. Removal of the upper 10 cm (about 4 inches) would expose a sediment layer with about half the available phosphorus concentration of the current surficial layer. Whether or not that would translate into a 50% reduction in internal loading is unknown, and further testing would be advised before planning a dredging program. Sediment cores can be sectioned and tested for phosphorus release in the laboratory, providing some estimate of the likely release rate after any given level of dredging.

The negative side of dredging is a function of direct impacts on organisms that might be sucked up in the dredging operation or indirect impacts on organisms through changing substrate conditions and food resources. Organisms with minimal mobility will be entrained in the hydraulic dredging pipe and transported to the containment area for dredged material. Mortality is possible by direct impact from the cutterhead, abrasion in the pipeline, or by adverse conditions in the containment area (most often low oxygen, high turbidity, and/or drying). Many benthic invertebrates (e.g., dragonfly or damselfly larvae) survive this ordeal, but some do not (many bivalve mollusks). The change in bottom features will affect which organisms choose to dwell there after dredging; shifts in the composition of biological communities are to be expected. If water clarity is sufficient and not all soft sediment has been removed, growth of rooted aquatic plants is likely to increase. While this may be a beneficial result, it could also involve invasive species such as Eurasian watermilfoil that can negatively affect habitat and water uses.

7.3 Phosphorus Inactivation

Use of aluminum to inactivate sediment phosphorus could reduce internal recycling by at least 50% and by even more than 90%, based on experience elsewhere (Welch and Cooke 1999). The highest reductions tend to match up with deep lakes with strong hypolimnetic anoxia, where aluminum breaks a cycle of phosphorus and iron releases and co-precipitation. While there is no anoxic hypolimnion in St. Albans Bay, the iron-phosphorus cycle appears to occur in the sediments, and may still be an important force in internal recycling. Data from Druschel et al. (2005) indicates that here is about five times as much iron as phosphorus, lower than desired for optimal binding stoichiometry. The addition of aluminum could be a great aid in lowering internal loading.

Oxic release rates tend to be much lower than anoxic release rates, but the change in phosphorus concentration in the bay could be accounted for by applying very typical oxic release rates. Considering how large the available phosphorus pool is in the surficial sediments, chemical equilibrium might be expected to drive the water column concentration up to observed levels, and the effect of wind may also enhance the transfer. While inputs from the watershed represent a threat in association with storms at any time of year, it is entirely possible that internal loading supplies the bulk of the measured phosphorus in the bay in late summer. If it is assumed that inactivation with aluminum will counteract 90% of that loading, the concentrations should not exceed 25 ug/L, and could be lower than 20 ug/L. It is not clear that the target level of 17 ug/L will be reached, but the probability of algal blooms should be greatly decreased.

The negative side of aluminum treatment is related mainly to potential toxicity at the time of treatment and possible burial and smothering of non-mobile fauna during the period of floc settling and reaction with the surficial sediments. With the experience of multiple New England lakes in hand and newer methods for dose determination and impact minimization verification available, the risk of fishkills and other adverse biological side effects can be minimized. The impact of smothering is transient, and recovery can be rapid, but there is a trade-off to be accepted between long-term benefits and short-term risks when applying aluminum compounds.

Toxicity is largely a function of interaction of aluminum with gills. Some mortality may not be actual toxicity at all, but rather clogging of gills, as aluminum is a coagulant. Organisms that breathe air are not at significant risk of direct impact. The direct risk is also transient, as once the aluminum reacts (it undergoes a one-way hydrolysis reaction), it is inert. Later changes in pH, oxygen, or other water quality variables do not appreciably alter the reacted aluminum. Reactions are active for several hours, then decline gradually until there is virtually no reactivity about three months later. Risk to most aquatic organisms passes within hours of treatment.

The toxicity from aluminum is avoidable with an appropriate ratio of buffering agent. An alum:aluminate ratio of 2:1 is usually used. This has been determined from experience and is consistent with assay results (ENSR 2001). However, traditional approaches that seek to maintain a pH of 6 to 8 may be inadequate, as it appears that aluminate may cause toxicity or at least stress at pH levels as low as 7.5. It also appears that toxicity occurs in a short period; brief exposure to an elevated pH at a high aluminum dose (e.g., 50 mg/L) can cause mortality. Some adjustment of buffer ratio may be justified, depending on local conditions. Assays and jar tests with a finer range of ratios than usually employed are recommended for verification prior to any treatment.

Other toxicity avoidance mechanisms include applying the dose sequentially, such that the concentration of aluminum is never >10 mg/L, with <5 mg/L preferred. Also, as only about 25 to 50 acres are treated in a day, a patchwork treatment scheme can be established that minimizes the risk to organisms in one portion of the targeted treatment area. By avoiding the treatment of adjacent areas in succession, refuges are maintained for sensitive fauna.

The burial of some sessile or minimally motile organisms like mollusks in a thick floc layer is likely in a treatment of the magnitude envisioned for St. Albans Bay. Organisms may smother in the time it takes for the floc to compress, react with surficial sediments, and become part of the upper sediment layer. Recolonization is then possible, and has been observed in the one well studied Vermont lake treatment at Lake Morey (Smeltzer et al. 1999), but short-term impact is likely. Assessment of benthic fauna, with a focus on protected species of mollusks, is essential prior to treatment.

Plants are largely unaffected by aluminum, although it is possible that the floc may coat them and limit the use of light in photosynthesis. However, as the goal of an alum treatment is to get the aluminum to interact with sediment, more than sparse coverage by rooted plants is an impediment to successful treatment. Treatment at a time when plants are absent or minimal, or removal of plants prior to treatment, is recommended.

The option to add aluminum to tributaries will affect internal load by reducing the addition of new available phosphorus to the surficial sediments, and by inactivating at least some of the available phosphorus already in that sediment, albeit only a fraction of what would be inactivated by a direct sediment treatment in any one year. It is not, however, intended to provide an immediate and obvious reduction in internal load. Rather, tributary treatment would be an interim treatment until watershed P sources are controlled and would demonstrate the relative role of watershed inputs during spring in determining summer conditions in the bay while starting the process of sediment phosphorus inactivation at a lower cost than for complete treatment of surficial sediments. It provides a benefit while maximizing the probability that this benefit will not be quickly erased by watershed loading prior to the implementation of watershed P controls. It is not clear, however, that tributary treatments will be adequate to significantly reduce internal loading in one or several years; if the internal load is really the primary force in algal blooms, no apparent benefit may accrue from tributary treatment for multiple years.

The risks with tributary treatment are similar to those with direct treatment of the bay, except that the risk is focused on the tributaries and immediately adjacent areas of the bay. Dilution alone will minimize impacts bay-wide. The dose for tributaries is considerably lower than for the bay treatment, and will not exceed the guidelines for protecting aquatic life from toxicity (the maximum concentration of aluminum would be 4 mg/L under the suggested scenarios). However, conditions in the tributaries will be changed to some degree, at least visually, and this may have an impact on migrating fish or other fauna. Most treatments of this type have been on systems heavily impacted by storm water runoff, so biotic effects have been considered of secondary importance. Careful monitoring would be needed during test runs to determine the efficacy of this approach on an environmental impact basis.

8.0 MANAGEMENT RECOMMENDATIONS

Restoring designated uses currently impaired by algal blooms and other eutrophication symptoms will require a comprehensive program of management strategies. Past studies have implicated internal loading of phosphorus as a contributing factor to the high phosphorus concentrations that persist in the bay despite reductions in point and non-point phosphorus sources in the watershed. Techniques that could be considered for reducing internal phosphorus loading include circulation, dredging and phosphorus inactivation. Given that St. Albans Bay continues to have large external phosphorus loads, watershed management techniques must be utilized in conjunction with any of these in-lake strategies to provide lasting relief from algal blooms.

When accomplished in conjunction with a reduction in watershed P loading, the most cost-effective in lake strategy to reduce the algal bloom problems in St. Albans Bay appears to be inactivation of phosphorus both in the surficial bay sediments and potentially in the tributary waters. By phasing the implementation of the treatment and completing the dosing station(s) first, assessment of the potential impact of significant reduction in phosphorus loading from the one or more tributaries could be accomplished. This may provide a bridge to allow in lake treatment to proceed prior to full implementation of watershed P controls. The dosing stations would not be necessary if substantial reductions in watershed phosphorus loading were realized prior to treatment of the bay using BMPs and other measures. Following up with inactivation of available phosphorus in the surficial sediments would directly counteract internal loading. The aluminum chemicals carry some risk of short-term toxicity to aquatic life, although recent advances in pre-treatment planning should minimize that risk. Some burial and related mortality may occur when the floc initially settles to the bottom, but the impact would be transient. As a consequence of high costs and some risk of non-target impacts, some sort of phasing and areal limitation on approaches seems prudent. Implementation also requires control over watershed inputs to prolong the benefits of any reduction in internal loading.

It may therefore be advantageous to focus efforts on only one part of the bay, such as inner bay. The 1992 data, if considered representative of the gradient of conditions in the bay during summer, indicate that Black Creek (in the wetland upstream of the bay) and the immediately adjacent portion of the inner bay have high levels of phosphorus that may be linked to continued watershed inputs. Here a dosing station may be most appropriate until watershed P inputs are reduced. The remainder of the inner bay and all of the middle bay show distinct signs of summer increase in phosphorus consistent with internal loading, and here the direct treatment of sediment for inactivation of available phosphorus would seem appropriate. Perhaps a sediment treatment in the inner bay with a dosing station at the Black Creek inlet to the inner bay would be the most appropriate combination to address internal loading while controlling external loading to a primary target portion of the bay prior to the full implementation of watershed P control.

While inactivation of phosphorus in the channels within the Black Creek wetland may eventually be necessary, dosing phosphorus above the Black Creek wetland in Stevens and/or Jewett Brooks may be difficult to permit because of perceived risk to the fish and wildlife in the diverse wetland habitat. Consequently, a dosing station at the mouth of Black Creek appears more desirable at this time, and combined with treatment within the inner bay to inactivate available phosphorus in the surficial sediments, would make an excellent test case for controlling algal blooms in a part of the bay which needs it the most. If successful the approach can be extended to other parts of the bay. Ultimately, the tributary treatments should be discontinued as watershed management becomes more comprehensive and successful, but such treatment appears to be a critical interim step to protect the investment in sediment treatment if control of internal P loading is desired in the short term.

Based on the cost estimates provided in previous sections, the inner bay sediment treatment would cost between $500,000 and $3.5 million, depending upon the stoichiometry of the inactivation process, which needs to be tested in the next phase of this process. The dosing of Black Creek at its inlet to the inner bay would cost between $257,000 and $350,000 in the first year, with only chemical and monitoring costs in subsequent years ($92,000 to $139,000). For a three year program, this suggests a rather wide range of $1,033,000 to $4,267,000. The most critical variable is the necessary aluminum dose for adequate inactivation of available phosphorus in surficial sediments, a fairly easy value to obtain.

Recommended steps for the Phase II implementation are outlined in the following section.

9.0 RECOMMENDED PHASE II STEPS

9.1 Tributary Phosphorus Inactivation Protocols

The following steps are recommended to perform alum dosing in tributaries:

- Confirm the location of the alum dosing station. Consider the phosphorus contribution of the tributary (Black Creek, including all of Stevens and Jewett Creeks), the seasonal timing of loading, access issues, and potential risk to ecological habitat. The estimated cost of consulting aid to accomplish this task is $1000.
- Collect storm samples in Black Creek and possible Mill Creek to characterize the current load of P in the spring and summer to the bay from these sources. Data from this effort would be useful in designing the alum dosing stations and estimating costs. It is assumed that these data can be collected by state or local residents and therefore, no consulting costs are associated with this task at this time.
- Collect any past hydrologic records and create a stage/discharge relationship at the chosen location if one does not exist. If a dosing station is placed at the mouth of the Black Creek, it may be difficult to generate an accurate relationship due to the influence of lake level, particularly when the lake and/or tributary levels are high; an upstream location for flow assessment may be preferable. The estimated cost of consulting aid to accomplish this task is $3000. It may be more expensive if new data must be collected.
- Determine the necessary dose of aluminum. Available data indicate a dose of between 2 and 4 mg Al/L, but some sampling during storms would be helpful in determining if some large loading spikes might be missed. Assays of storm water with this range of aluminum concentration, to determine available phosphorus remaining after treatment, would be helpful. The estimated cost of consulting and laboratory aid to accomplish this task is $4000.
- Calculate the amount of aluminate necessary for buffering. The same assays used to determine effectiveness of the treatment can be applied to determine resulting pH control needs and the best ratio of alum to aluminate to minimize toxicity risk. The estimated cost of consulting and laboratory aid to accomplish this task is $2500.

- Permitting: The estimated cost of consulting aid to accomplish this task is $5000 and includes agency consultation, completion of an Aquatic Nuisance Control Permit and attendance of an informational meeting but not efforts associated with appeals or challenges to the permit or any federal permitting.
- Develop a monitoring program for the dosing process. This could involve automated analysis for some variables (e.g., pH) and automated sampling for others with lab analysis (total and dissolved P and Al) at an upstream and downstream site. The estimated cost of consulting aid to accomplish this task is $1000. Actual monitoring equipment will cost about $30,000 for automated sampling.

The cost of consulting and laboratory support for Phase II of this project with regard to tributary dosing for phosphorus control is approximately $16,500, exclusive of permitting costs beyond those described above and any automated sampling equipment that might be purchased for Phase III. Note, however, that not all tasks require consulting aid; the state agencies or university could conduct some of the needed tasks for a lesser cost. This cost estimate does not include costs associated with inventory of rare, threatened or endangered species in the study area.

9.2 In-lake Phosphorus Inactivation Protocols

The following steps are recommended to perform an in-lake phosphorus inactivation treatment:

- Make a final determination of the treatment area(s). Due to the size of the entire bay, it may be more economical to apply alum to sections of the bay rather than the entire bay. A treatment of just the inner bay is advised, in conjunction with a dosing station in Black Creek. No cost is associated with this task.
- Review the results of the P flux measurements being conducted on the bay this summer. Consider adding sufficient replicates in each zone of the bay (wetland, IB, MB and OB) to estimate flux to the water column. These data are expected to confirm earlier model estimates of internal P loading (Smeltzer et al 1994). The preliminary work is being conducted by researchers at University of Vermont (UVM) so no cost is currently associated with this task. Follow up work, if desired, could be conducted by either UVM or the consultant.
- Sample sediment in the section(s) of the bay chosen for alum inactivation. Test for available phosphorus again and conduct dosing tests with alum to estimate the dose necessary to adequately inactivate available phosphorus. The “optimal” dose is defined as a point on a curve of aluminum dose vs. reduction in phosphorus availability, such that higher doses result in diminishing returns while

lower doses inactivate too little phosphorus to meet water quality goals. The estimated cost of consulting and laboratory aid to accomplish this task is $5000.

- Measure pH and dissolved oxygen, in the water column and the sediments in order to check whether the water is aerobic and identify the depth of the redox zone in the sediment. Diurnal measurements may be advantageous. The estimated cost of consulting aid to accomplish this task is $3000.
- Once the aluminum dose has been determined, perform bioassays with St. Albans Bay water to which alum and aluminate solutions are added at ratios of 1.6:1, 1.8:1, 2.0:1 and 2.2:1 at the maximum dose expected in mg/L and increments of half the next higher dose until the aluminum concentration is <10 mg/L. This will verify the preferred alum:aluminate ratio and investigate any need for a split dose (mulitple applications separated by several days) or a spread dose (application at multiple levels to reduce maximum concentration). Current knowledge suggests that the 2:1 dose will be appropriate, and that an aluminum concentration of 10 mg/L will be tolerable at that ratio. The areal dose is likely to be between 17.6 and 176 g/m^2, which for an area averaging 10 ft deep, equates to 6 to 60 mg/L, a wide range for which toxicity testing is needed. The estimated cost of consulting aid to accomplish this task is $12,000. This cost could decline once the target dose is known, as it may not be the highest level anticipated in this assessment.
- Divide the treatment area into sectors based on any changes in dose to the sediment, substantial differences in depth, and any known sensitive resources that may require special attention. Divide these sectors into daily target treatment areas of about 25 acres. A larger area can be used if the dose is low, but that seems unlikely in this case. The estimated cost of consulting aid to accomplish this task is $1000.
- Calculate the gallons of alum and gallons of aluminate to be added to each sector, both in total and during any given day of application (where split treatment is to be performed). The estimated cost of consulting aid to accomplish this task is $1500.
- Determine the application depth. It may be acceptable to treat at the surface, and may be a necessity in shallow parts of the bay. Treatment well below the surface will not remove the phosphorus from that upper water volume, and may slow results, but shallow depth and substantial mixing limits the need to consider depth of application in this instance. No cost is associated with this task.
- Determine the time of application. Treatment in the spring as soon as the water warms to at least 10^{o}C (50^{o}F), will provide the most immediate beneficial results, but late summer treatment is possible and minimizes potential impacts on spring spawning fish. Floc formation is effective and efficient above 10^{o}C. The primary

impediment to fall treatment is rooted plant densities, which if high can inhibit even distribution of floc and effective inactivation of sediment phosphorus. No cost is associated with this task.

- Develop a monitoring program to support the project, including elements to be conducted prior to, during, and after treatment. Measure pH, alkalinity, dissolved aluminum, and phosphorus at selected stations prior to treatment, just after treatment, and monthly through the following summer. Monitor water clarity and chlorophyll at each station as well. Monitor pH and alkalinity each day of treatment at multiple depths within the treatment zone and at an untreated control station. Observe fish behavior visually from the surface and with underwater video equipment. The key target conditions include a pH between 6 and 7.5 at all times and no fish mortality. Some benthic invertebrate mortality (such as mollusks) may be unavoidable, and a monitoring program to quantify any losses may be needed. The estimated cost of consulting aid to develop the monitoring program is $1500.
- Permitting - The estimated cost of consulting aid to accomplish this task is $5000 and includes agency consultation, completion of an NPDES Permit and attendance of an informational meeting but not efforts associated with appeals or challenges to the permit or any federal permitting.

The cost of consulting and laboratory support for Phase II of this project with regard to inactivation of phosphorus in surficial sediments is approximately $29,000, exclusive of permitting, although it is likely that bioassay costs will be lower; and estimate of $25,000 is suggested. Note, however, that not all tasks require consulting aid; the state agencies or university could conduct some of the needed tasks for a lesser cost. This cost estimate does not include costs associated with inventory of rare, threatened or endangered species in the study area.

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