

# Dynamic Mass Balance Model of Internal Phosphorus Loading in St. Albans Bay, Lake Champlain



Lake Champlain  
Basin Program

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for  
Lake Champlain Management Conference

March 1994

**DYNAMIC MASS BALANCE MODEL  
OF INTERNAL PHOSPHORUS LOADING  
IN ST. ALBANS BAY, LAKE CHAMPLAIN**

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**Prepared for**

**Lake Champlain Management Conference  
Lake Champlain Basin Program  
54 West Shore Rd.  
Grand Isle, Vermont 05458**

**March 1994**

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This technical report is the seventh in a series of reports prepared under the Lake Champlain Basin Program. Those in print are listed below.

#### Lake Champlain Basin Program Technical Reports

1. *A Research and Monitoring Agenda for Lake Champlain.* Proceedings of a Workshop, December 17-19, 1991, Burlington, VT. Lake Champlain Research Consortium. May, 1992.
2. *Design and Initial Implementation of a Comprehensive Agricultural Monitoring and Evaluation Network for the Lake Champlain Basin.* NY-VT Strategic Core Group. February, 1993.
3. (A) *GIS Management Plan for the Lake Champlain Basin Program.* Vermont Center for Geographic Information, Inc., and Associates in Rural Development. March, 1993.  
  
(B) *Handbook of GIS Standards and Procedures for the Lake Champlain Basin Program.* Vermont Center for Geographic Information, Inc. March, 1993.  
  
(C) *GIS Data Inventory for the Lake Champlain Basin Program.* Vermont Center for Geographic Information, Inc. March, 1993.
4. (A) *Lake Champlain Economic Database Project. Executive Summary.* Holmes & Associates. March 1993.  
  
(B) *Socio-Economic Profile, Database, and Description of the Tourism Economy for the Lake Champlain Basin.* Holmes & Associates. March 1993  
  
(C) *Potential Applications of Economic Instruments for Environmental Protection in the Lake Champlain Basin.* Anthony Artuso. March 1993.  
  
(D) *Conceptual Framework for Evaluation of Pollution Control Strategies and Water Quality Standards for Lake Champlain.* Anthony Artuso. March 1993.
5. *Lake Champlain Sediment Toxics Assessment Program. An Assessment of Sediment - Associated Contaminants in Lake Champlain - Phase 1.* Alan McIntosh, Editor, UVM School of Natural Resources. February 1994.  
  
*Lake Champlain Sediment Toxics Assessment Program. An Assessment of Sediment - Associated Contaminants in Lake Champlain - Phase 1. Executive Summary.* Alan McIntosh, Editor, UVM School of Natural Resources. February 1994.
6. (A) *Lake Champlain Nonpoint Source Pollution Assessment.* Lenore Budd, Associates in Rural Development Inc. and Donald Meals, UVM School of Natural Resources. February 1994.  
  
(B) *Lake Champlain Nonpoint Source Pollution Assessment. Appendices A-J.* Lenore Budd, Associates in Rural Development Inc. and Donald Meals, UVM School of Natural Resources. February 1994.

7. *Internal Phosphorus Loading Studies of St. Albans Bay. Executive Summary.* VT Dept of Environmental Conservation. March 1994.

(A) *Dynamic Mass Balance Model of Internal Phosphorus Loading in St. Albans Bay, Lake Champlain.* Eric Smeltzer, Neil Kamman, Karen Hyde and John C. Drake. March 1994.

(B) *History of Phosphorus Loading to St. Albans Bay, 1850 - 1990.* Karen Hyde, Neil Kamman and Eric Smeltzer. March 1994.

(C) *Assessment of Sediment Phosphorus Distribution and Long-Term Recycling in St. Albans Bay, Lake Champlain.* Scott Martin, Youngstown State University. March 1994.

This report was funded and prepared under the authority of the Lake Champlain Special Designation Act of 1990, P.L. 101-596, through the U.S. Environmental Protection Agency (EPA grant #EPA X 001840-01). Publication of this report does not signify that the contents necessarily reflect the views of the States of New York and Vermont, the Lake Champlain Basin Program, or the U.S. Environmental Protection Agency.

March 1994

## **ABSTRACT**

### **DYNAMIC MASS BALANCE MODEL OF INTERNAL PHOSPHORUS LOADING IN ST. ALBANS BAY, LAKE CHAMPLAIN**

St. Albans Bay, Lake Champlain has a long history of severe phosphorus enrichment and eutrophication. An upgrade of the City of St. Albans Wastewater Treatment Facility and implementation of agricultural best management practices during the 1980's have not yet resulted in significant improvements in water quality in the bay. Recent water quality modeling analyses indicated that the reason for the delayed recovery is continued internal loading from phosphorus stored in the system. This study was designed to identify the timing and location of internal phosphorus loading in the St. Albans Bay and wetland system so that appropriate lake management steps could proceed. The central question for the study was whether the internal phosphorus loading was confined primarily to the Stevens Brook wetland, or whether extensive areas of the bay were involved as well. The sampling program for this study involved chloride and phosphorus measurements made throughout the bay and wetland during April-November 1992, and included data used to estimate flows and loadings to the system. A dynamic (time-dependent) mass balance model using the STELLA II ® object-oriented modeling program was calibrated to the field data and used to simulate the effects of reducing internal phosphorus loading in various parts of the system. The modeling analysis showed that reducing internal loading in the wetland alone would do little to eliminate the late summer phosphorus maxima and algae blooms that consistently plague the bay each year. Internal loading reductions would need to occur throughout a 700 acre region of the bay for water quality to improve. Internal phosphorus loading in St. Albans Bay during the critical summer period appears to be a complex interaction of mechanisms including aerobic sediment phosphorus release at high temperature and pH, coupled with seasonally low wind-induced exchange mixing rates that render the bay more vulnerable to the effects of internal loading during the late summer. Sediment phosphorus inactivation using aluminum salts may be a practical lake management technique to control internal loading in St. Albans Bay, but treating 700 acres of bay would cost approximately \$350,000 to \$525,000. Before such a project could proceed, additional pilot scale treatments and other feasibility studies would be necessary to determine the stability and longevity of the aluminum floc layer, and to assess adverse environmental effects of the treatment.

## INTRODUCTION

St. Albans Bay, located in the Northeast Arm of Lake Champlain in Vermont (Figure 1), has a long history of severe water quality problems related to excessive phosphorus enrichment and resulting algae blooms. Studies have shown that prior to 1987, phosphorus discharged from the City of St. Albans Wastewater Treatment Facility was the major source of phosphorus to St. Albans Bay, particularly during the summer (Little, 1977; Bogdan, 1978; U.S. Environmental Protection Agency, 1974; Henson and Gruending, 1977; Smeltzer, 1983). Agricultural nonpoint sources also contributed to the problem. Water quality models (Laible, 1985; Smeltzer, 1983) predicted that upgrading the St. Albans Treatment Plant to remove phosphorus from the effluent would dramatically reduce phosphorus and algae levels in St. Albans Bay.

As a result of these studies, major efforts were undertaken during the 1980's to control point and nonpoint source phosphorus loading to St. Albans Bay. The upgrade of the St. Albans Treatment Plant was completed in 1987 at a cost of \$2.3 million for phosphorus removal processes. Phosphorus loading from this facility was reduced by 90%, and the plant has consistently met its discharge permit limit of 0.5 mg/l for phosphorus, which is the lowest phosphorus limit required for any municipal treatment facility in Vermont. The St. Albans Bay Rural Clean Water Program was completed in 1991 at a cost of \$2.2 million for the implementation of agricultural best management practices throughout the bay's watershed.

Water quality monitoring data (St. Albans Bay Rural Clean Water Program, 1991; Picotte and Lohner, 1993) have shown that no significant phosphorus or algae reductions have occurred in St. Albans Bay since the treatment plant upgrade was completed in 1987. Recent updated modeling analyses (Smeltzer, 1991) suggest that the reason for the lack of water quality improvement is continued internal phosphorus loading from historical residues stored in the bay and/or wetland sediments. A delay of several years in the recovery of St. Albans Bay following the treatment plant upgrade had been previously expected because of this stored phosphorus (Smeltzer, 1983; Henson and Gruending, 1977).

This study was initiated in 1992 to analyze the location, seasonal timing, and mechanisms of internal phosphorus loading in the St. Albans Bay and wetland system. A central question for this study was whether a major portion of the internal loading is derived from sources within the Stevens Brook wetland which has received much of the historical phosphorus loadings (see Figure 1), or whether the bay sediments are also an important internal source of phosphorus. A better understanding of the location and mechanisms of internal phosphorus loading in St. Albans Bay was necessary before management techniques to control the internal loading could be considered.

This report presents the results of a water quality sampling program conducted in St. Albans Bay during 1992. The data were used to develop a phosphorus mass balance model for the bay in order to identify specific locations in the system where internal loading would need to be reduced in order to restore better water quality.

This study was supported by a grant from the Lake Champlain Basin Program and by the Vermont Department of Environmental Conservation. Related projects presented in Hyde *et al.* (1994) and Martin *et al.* (1994) documented the long-term history of phosphorus loadings to St. Albans Bay and developed a long-term water quality model for the bay to estimate the time that may be required before water quality in the bay improves in response to external phosphorus loading reductions. The combined results from these studies are intended to promote sound

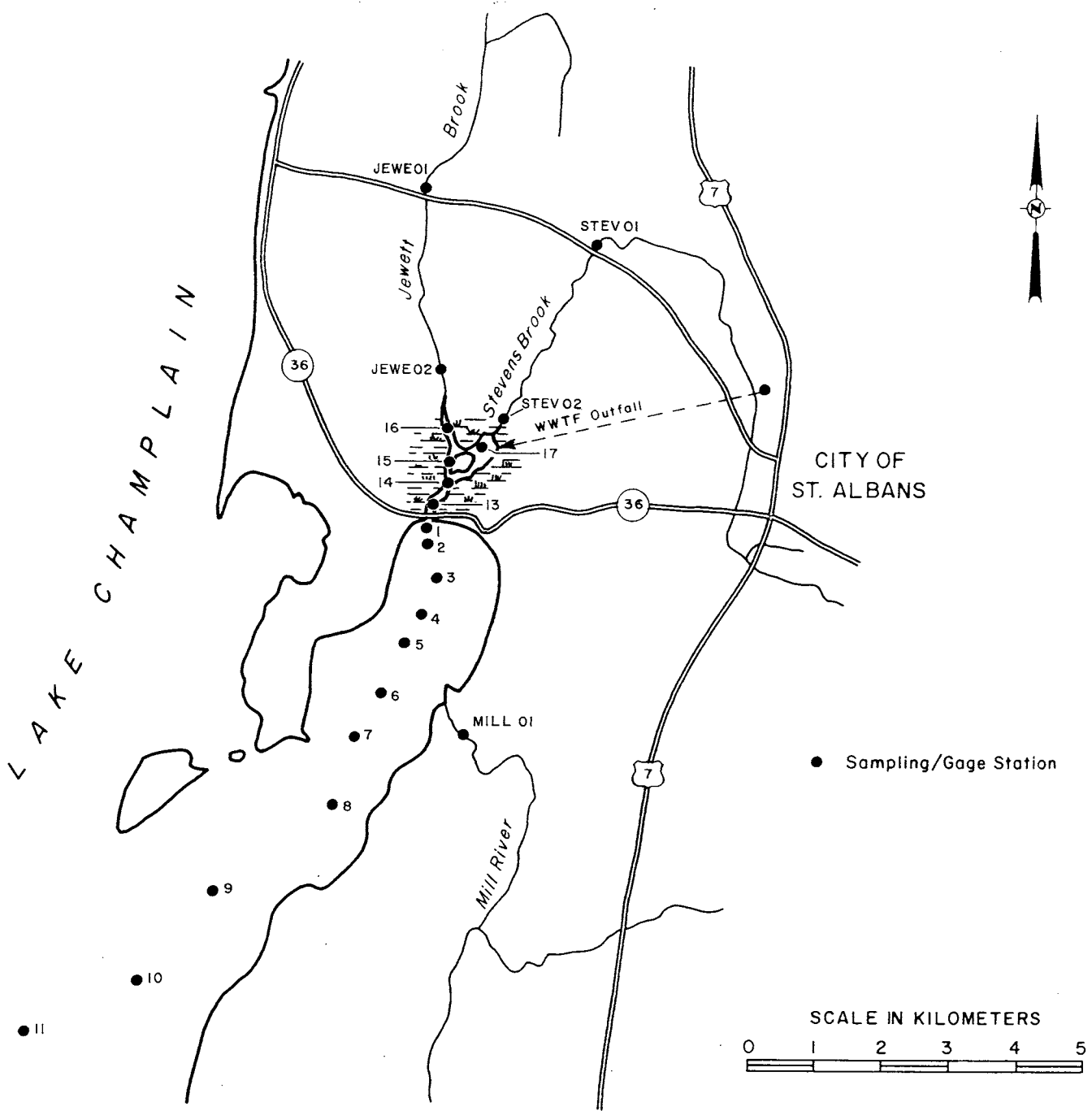


Figure 1. Map of St. Albans Bay area showing location of bay, wetland, and tributary sampling stations.

decisions regarding possible future lake management intervention to control internal phosphorus loading in St. Albans Bay.

## **SAMPLING AND ANALYTICAL METHODS**

### **Bay and Wetland Sampling**

St. Albans Bay and the Stevens Brook wetland were sampled at 17 stations during 1992 at locations shown in Figure 1 and listed in Table 1. The sampling stations were established in a linear sequence along the central axis of the wetland and bay. The distance intervals between stations were shorter in the wetland and inner bay where spatial concentration gradients were known to be greater. Wherever possible, sampling stations were co-located with sites used in previous studies (e.g. St. Albans Bay Rural Clean Water Program, Vermont Lay Monitoring Program, Lake Champlain Diagnostic-Feasibility Study), in order to promote data comparability.

The bay and wetland stations were sampled approximately weekly from mid-April through November, 1992. A total of 26-32 samples were obtained at each station.

At stations where the total depth was 7 meters or less (see Table 1) and unstratified conditions were known to exist from previous studies, a single sample was collected at the 0.5 meter depth using a Kemmerer sampler. At stations where the total depth was greater than 7 meters, a single depth-integrated composite sample was generated by combining equal-volume samples obtained at 5 meter depth increments using a Kemmerer sampler, down to a maximum depth of 20 meters.

All bay and wetland samples were analyzed for total phosphorus, total dissolved phosphorus, and chloride, using analytical methods given in Table 2. In addition, discrete depth samples were obtained near the bottom of the water column using a Kemmerer sampler at all wetland stations (13-17) and at the inner bay stations (01-04). The bottom samples were analyzed for dissolved oxygen, pH, and temperature, using methods given in Table 2. The Secchi depth was also recorded at all stations on each sampling date.

### **Tributary and Wastewater Treatment Plant Sampling**

Tributary sampling stations were established on the three major inflows to St. Albans Bay (Stevens Brook, Jewett Brook, Mill River) at locations shown in Figure 1 and listed in Table 1. Tributary stations were co-located with sites sampled during previous studies as indicated in Table 1. New downstream stations were established on Jewett and Stevens Brooks to monitor any additional loadings to these streams that occurred before they entered the wetland.

The tributary stations were sampled a total of 9-28 times during April-November 1992, with emphasis on the sampling of higher flow conditions when they occurred. Samples of the water column at the center of the stream were obtained using vertically-integrating DH-48 or DH-59 "suspended sediment samplers" (Edwards and Glysson, 1988). Tributary samples were analyzed for total phosphorus, dissolved phosphorus, and chloride according to methods given in Table 2. This sampling program was designed to support tributary load estimation procedures available using the FLUX program (Walker, 1987, 1990).

Table 1. Location of bay and tributary sampling stations.

<u>Station</u>	<u>Latitude (deg min)</u>	<u>Longitude (deg min)</u>	<u>Total Depth (m)</u>	<u>Co-Located Stations<sup>1</sup></u>
<b>Bay/Wetland</b>				
01	44 48.60	73 09.15	1	RCWP-14
02	44 48.47	73 09.18	3	RCWP-12
03	44 48.12	73 08.97	4	
04	44 47.80	73 09.14	6	LCDFS-41
05	44 47.42	73 09.47	6	
06	44 47.12	73 09.73	7	LMP-17, LCDFS-40
07	44 46.78	73 09.98	7	
08	44 46.42	73 10.15	9	RCWP-11
09	44 45.97	73 11.20	20	LCDFS-37
10	44 45.13	73 12.13	22	
11	44 44.22	73 13.10	32	
13	44 48.71	73 09.08	1	
14	44 48.99	73 08.76	1	
15	44 49.18	73 08.80	1	
16	44 49.46	73 08.80	1	
17	44 49.27	73 08.50	1	
<b>Tributaries</b>				
STEV01	44 50.95	73 06.18		RCWP-22, LCDFS-STEVO1
STEV02	44 49.33	73 08.28		
JEWE01	44 51.37	73 09.08		RCWP-21
JEWE02	44 50.18	73 09.00		
MILL01	44 46.78	73 08.67		RCWP-24, LCDFS-MILL01
WWTF				RCWP-25, LCDFS-SAM

1. RCWP = St. Albans Bay Rural Clean Water Program (1991)  
LMP = Vermont Lay Monitoring Program (Picotte and Lohner, 1993)  
LCDFS = Lake Champlain Diagnostic-Feasibility Study (Vermont DEC and New York State DEC, 1992)

Table 2. Analytical methods (Vermont DEC, 1992; U.S. Environmental Protection Agency, 1983).

<u>Parameter</u>	<u>Database Code</u>	<u>Method Description</u>	<u>U.S.E.P.A. Method Number</u>
Total Phosphorus	TP	Colorimetric, ascorbic acid, persulfate digestion	365.1
Dissolved Phosphorus <sup>1</sup>	DP	Colorimetric, ascorbic acid, persulfate digestion	365.1
Chloride	CL	Colorimetric, automated ferric thiocyanate	325.1
Dissolved Oxygen	DO	Winkler titration	360.2
pH	PH	Field, potentiometric	150.1
Temperature	TEMP	Field, thermistor	170.1

<sup>1</sup> 0.45  $\mu$ m filtration in field

Samples from 24-hour composites of the final effluent from the St. Albans City Wastewater Treatment Facility were obtained on a monthly frequency during the study with the assistance of the plant operator. The treatment plant samples were analyzed for total phosphorus and chloride. The discharge to the wetland from the Northwest State Correctional Facility was not sampled during this study because previous results (Vermont DEC and New York State DEC, 1992) had shown this plant to be an extremely small phosphorus source.

### **Tributary and Wastewater Flow Measurements**

Staff gages were maintained on Stevens Brook, Jewett Brook, and the Mill River at locations used for sampling and flow gaging during previous studies (see Table 1). Daily instantaneous stream stage observations were recorded at each site during April-November 1992 with the assistance of Mr. Don Jarrett. The daily stage measurements were converted to estimates of average daily flow using stage-discharge relationships previously established for the sites by the U.S. Geological Survey and by the St. Albans Bay Rural Clean Water Program. The validity of the stage-discharge relationships was confirmed by a limited number of direct flow measurements made during 1992 using standard in-stream current meter techniques.

Daily flows estimated at each tributary gage station were pro-rated for the additional drainage area downstream of the gage station. Flows recorded at the gage stations were multiplied by the drainage area factors given in Table 3 to estimate the flows at the mouths of each stream. The combined 118 km<sup>2</sup> drainage area of Stevens Brook, Jewett Brook, and the Mill River represents about 87% of the total 135 km<sup>2</sup> drainage area of St. Albans Bay. Loadings from the small ungaged areas were considered to be of minor significance for the purposes of this study, and were not accounted for.

Average daily effluent flows from the St. Albans Wastewater Treatment Facility were continuously recorded by the plant operators during the study period and filed at the Vermont Department of Environmental Conservation. The reported flows included a significant portion of recirculated wastewater as a result of an improperly located flow meter, causing an over-estimate of the actual effluent flow. A correction factor of 0.67 was applied to the effluent flow data recorded during the study period, based on a calibration with a new, properly located flow meter operated concurrently during a two week period of dry weather flows in March 1993.

### **Bay Segmentation and Morphometry**

For purposes of data reduction, and in anticipation of the requirements of the mass balance modeling analysis, the bay and wetland were divided into a linear sequence of several discrete segments, as shown in Figure 2. The segments included two main stem wetland segments and four segments in the bay, bounded by the outer lake region. Boundaries between segments were established so that chloride and phosphorus concentration gradients within each segment were relatively slight, based on a review of the sampling results. Each segment contained one to three sampling stations, as shown in Figure 2.

The surface areas and volumes of each segment given in Table 4 were determined by areal planimetry of a 1:40,000 scale N.O.A.A. bathymetric chart for Lake Champlain and a 1:5,000 scale Vermont Mapping Program orthophoto base map of the wetland. The wetland includes a distinct open-water channel area bounded on the sides by emergent vegetation. The values given in Table 4 for wetland segments 1 and 2 are for the channelized area only. The volumes of the two wetland segments were estimated by assuming that the segment mean depths were

Table 3. Tributary drainage areas.

<u>Tributary</u>	<u>Drainage Area at Gage (km<sup>2</sup>)</u>	<u>Drainage Area at Mouth (km<sup>2</sup>)</u>	<u>Mouth/Gage Factor</u>
Stevens	26	39 <sup>1</sup>	1.50
Jewett	14	20 <sup>1</sup>	1.43
Mill	58	59	1.02
Other		17	
Total to Bay		135	

<sup>1</sup> at confluence with wetland

Table 4. Bay and wetland segment morphometry.

<u>Segment</u>	<u>Sampling Stations</u>	<u>Area (m<sup>2</sup>)</u>	<u>Volume (m<sup>3</sup>)</u>
1	14, 15	1.47 E5	1.19 E5
2	13	3.06 E4	4.44 E4
3	1, 2	5.47 E5	1.12 E6
4	3, 4	2.04 E6	6.63 E6
5	5, 6, 7	4.33 E6	1.17 E7
6	8, 9	7.03 E6	7.21 E7
Lake	10, 11		

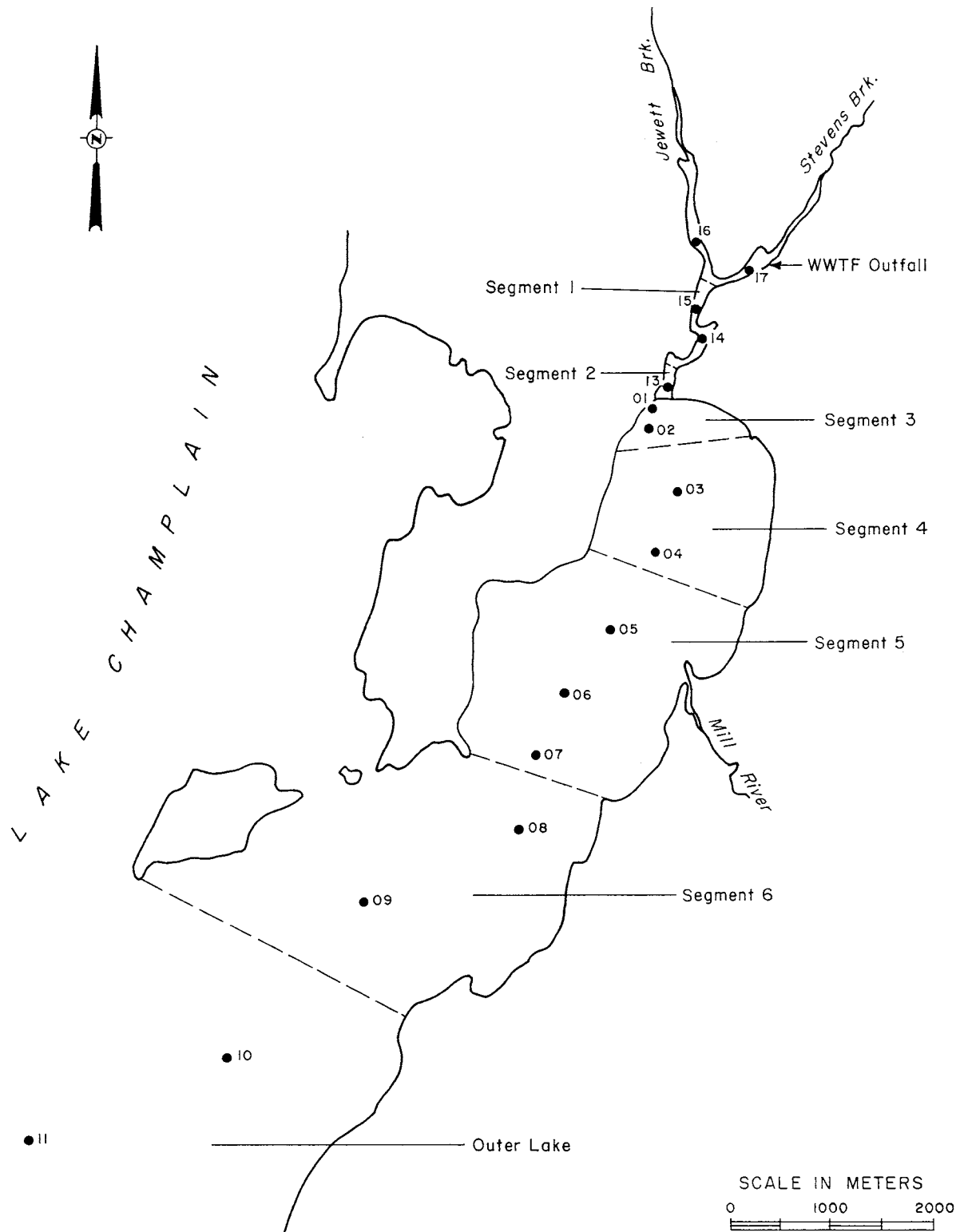


Figure 2. St. Albans Bay modeling segments and sampling stations.

approximately equal to half of the maximum total station depth measured on each sampling date during the study.

The area and volume of the Stevens Brook wetland are affected by seasonal variations in the level of Lake Champlain (Clausen and Johnson, 1989), which during April-November 1992 varied over a range of 1.4 meters. Thus, the mean values given in Table 4 for the volumes of the channelized areas of wetland segments 1 and 2 actually varied during the study by  $\mp$  50-90%.

## **SAMPLING RESULTS**

### **Bay and Wetland**

Segment mean values for each water quality parameter on each sampling date were calculated by averaging the results obtained at each sampling station within the segment. Averaging by segment was done for purposes of data reduction and presentation, and for consistency with the input data requirements of the subsequent mass balance modeling analysis. However, all original water quality data collected during this study are available on request in a personal computer database maintained at the Vermont Department of Environmental Conservation.

#### *Chloride*

The chloride concentration results are shown in Figure 3. A strong spatial gradient of chloride concentration was evident, as seen during previous studies (Smeltzer, 1991), with highest values occurring in the wetland and inner bay segments.

Chloride concentrations were somewhat higher and more variable in the spring and fall than during the summer in most segments. Variability from one sample date to the next was greatest in the two wetland segments (1 and 2). The variability within the wetland was primarily the result of rapidly changing hydrodynamic mixing conditions between the wetland and the bay in the vicinity of Black Bridge. Laible (1985) documented periodic oscillations in flow direction and velocity at this site that occurred on an hourly time scale in response to wind events. Such periodic inflow of relatively low concentration bay water into the wetland results in rapid concentration fluctuations within the wetland that are not fully characterized by the weekly sampling results shown in Figure 3.

#### *Phosphorus*

The total and dissolved phosphorus concentration results are shown in Figure 4. A strong spatial gradient of phosphorus concentration was observed. Total phosphorus concentrations in the wetland were in the extremely eutrophic range of 0.1-0.8 mg/l. Total phosphorus levels declined through the system to background values in the outer lake in the range of 0.010-0.020 mg/l.

Figure 4 shows that there were two periods of maximum phosphorus concentrations observed in the bay during 1992, with the first peak occurring in late August and the second maximum occurring in early November. These phosphorus maxima were observed in all wetland and bay segments. The August peak was particularly significant because it corresponded with a major algae bloom during the height of the recreational season. Water clarity and aesthetic

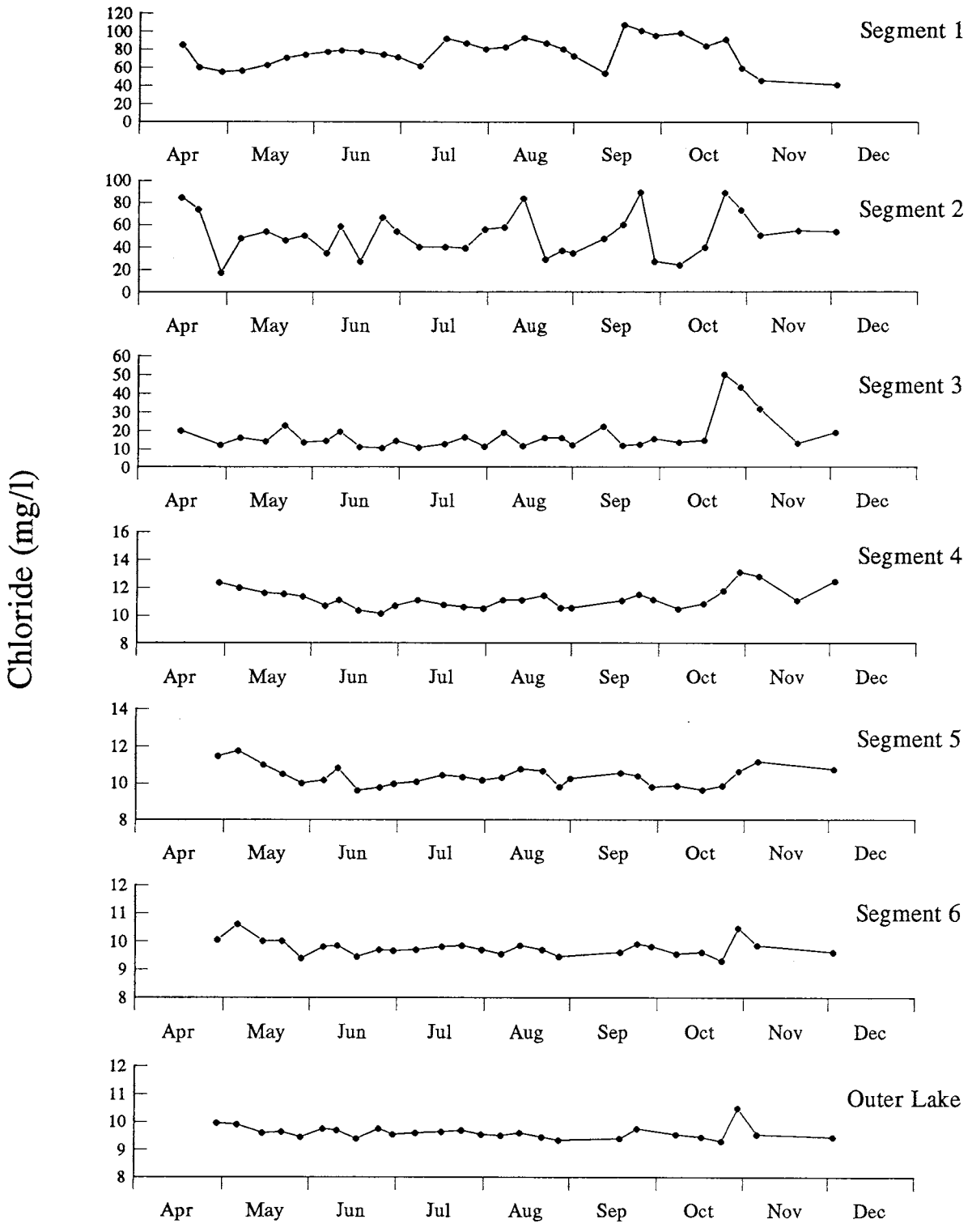


Figure 3. Bay and wetland chloride sampling results, 1992.

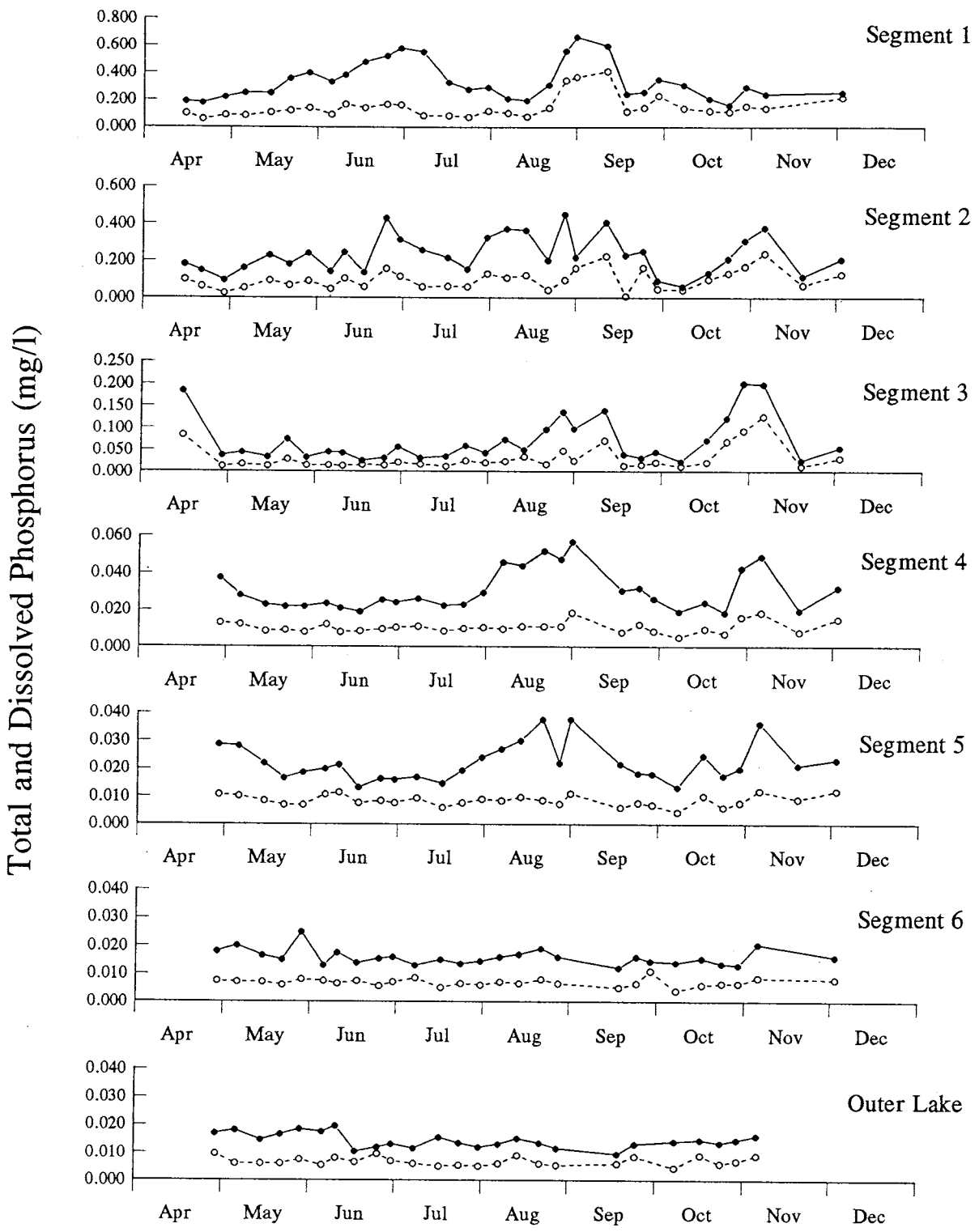


Figure 4. Bay and wetland total phosphorus (●) and dissolved phosphorus (○) sampling results, 1992.

quality were substantially reduced in the inner bay areas, as evidenced by the low Secchi disk transparency values recorded in August (Figure 5).

Dissolved phosphorus levels in most of the bay segments were usually less than half of the total phosphorus values, indicating that most of the phosphorus in the water column in the bay was associated with algae or other particulate matter. However, dissolved phosphorus represented the majority of the total phosphorus present in the wetland and inner bay segments (1, 2, and 3) during the August and November phosphorus maxima. This observation suggests that the loading source causing these maxima contributed primarily dissolved phosphorus.

A review of long-term water quality monitoring data for St. Albans Bay indicated that maximum phosphorus concentrations have typically occurred in August in the years since the 1987 wastewater treatment plant upgrade. The data from three long-term monitoring stations in the inner bay are summarized in Figure 6. Late summer phosphorus maxima have consistently occurred during this period.

#### *Dissolved Oxygen, Temperature, and pH*

Plots of bottom dissolved oxygen, temperature, and pH are shown in Figures 7, 8, and 9. Figure 7 shows that widespread anoxic conditions were never observed in these segments during the 1992 study period. There were two days (May 5 and September 11) on which anoxic samples were obtained at station 16 in the Jewett Brook arm of the wetland. The anoxic episode in May at this station corresponded with the occurrence of highly colored, brown water having an obvious manure smell, and a localized fish kill. Temporarily heavy organic and/or ammonia loading to Jewett Brook from agricultural waste was the apparent cause of the temporary oxygen depletion observed in May at station 16. The cause of the September anoxic episode is unknown.

With the exception of the two anoxic episodes at station 16, dissolved oxygen levels in the wetland and the bay remained high enough throughout the 1992 sampling season (Figure 7) to avoid severe reducing conditions in the water column. One set of pre-dawn dissolved oxygen samples was obtained on July 30 to check for the possibility of overnight oxygen depletion, but no water column anoxia was observed at that time. These results are consistent with other studies on St. Albans Bay which have reported generally well-mixed and unstratified conditions throughout the shallow water regions of the bay (Henson and Gruendling, 1977).

Figure 8 shows the seasonal pattern of temperature recorded in the wetland and inner bay stations during the study. The period of highest temperature corresponded with the August phosphorus maxima measured throughout the wetland and bay (see Figure 4).

Seasonal variations in bottom pH are shown in Figure 9. Elevated pH values in the range of 8.0-9.0 were frequently measured. Maximum pH values were observed in late August in the inner bay and in early October in the wetland.

#### **Tributary and Wastewater Flows and Loadings**

Hydrographs of average daily flow for Stevens Brook, Jewett Brook, Mill River, and the St. Albans City Wastewater Treatment Plant are shown in Figure 10. Most of the stream runoff during the 1992 study period occurred in the early spring and in the fall. Stream flows during

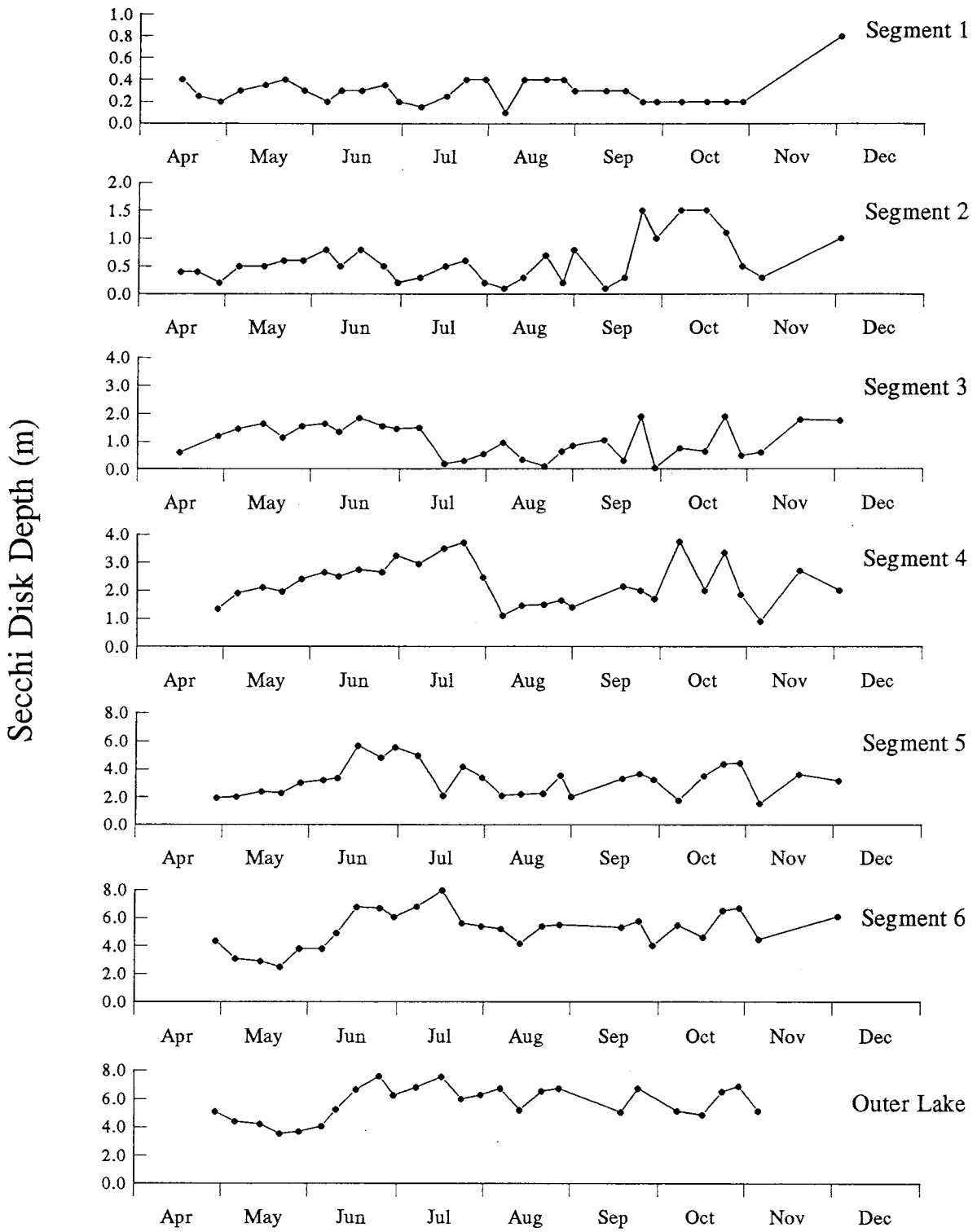


Figure 5. Bay and wetland Secchi disk sampling results, 1992.

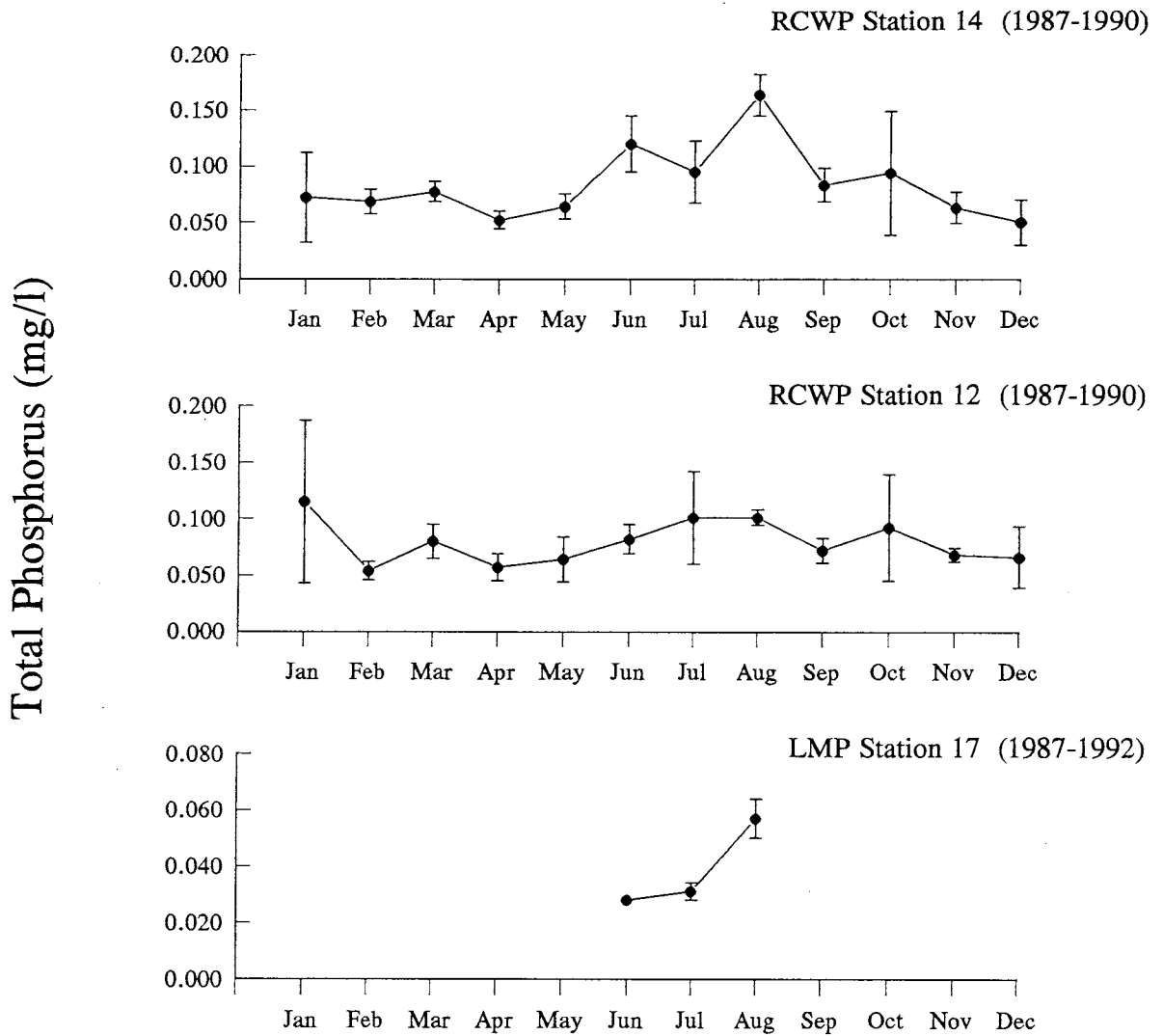


Figure 6. Monthly mean total phosphorus concentrations at three long-term sampling stations in St. Albans Bay for the period since the wastewater treatment plant upgrade was completed.

RCWP = St. Albans Bay Rural Clean Water Program (1991)  
 LMP = Vermont Lay Monitoring Program (Picotte and Lohner, 1993)

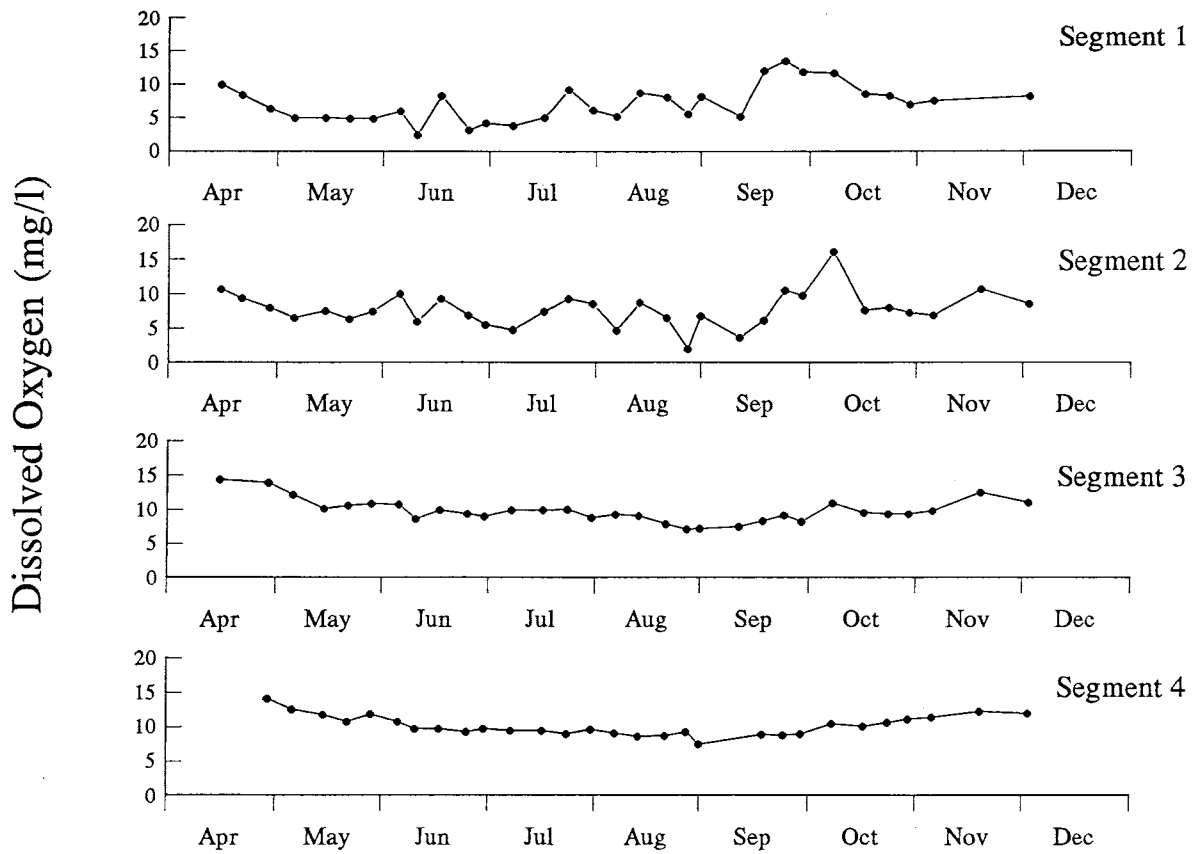


Figure 7. Bay and wetland bottom dissolved oxygen sampling results, 1992.

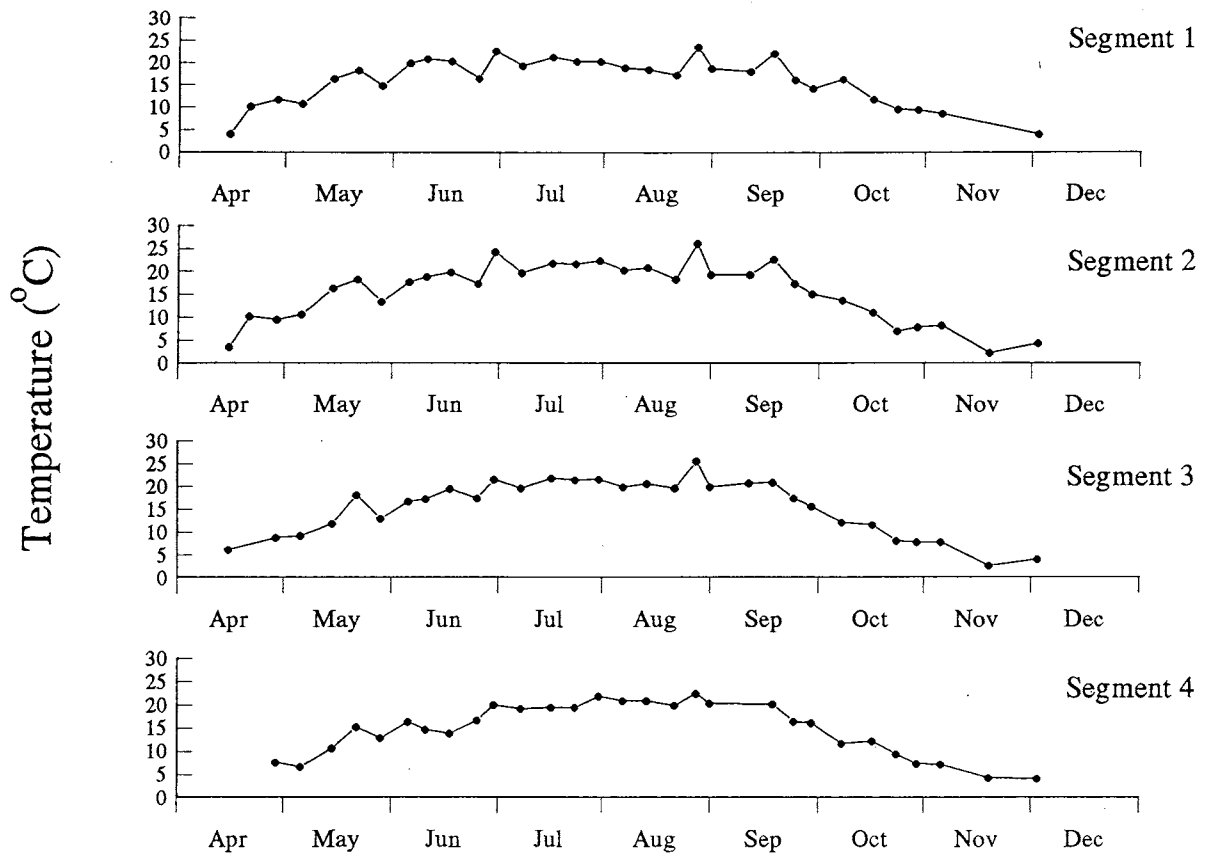


Figure 8. Bay and wetland bottom temperature sampling results, 1992.

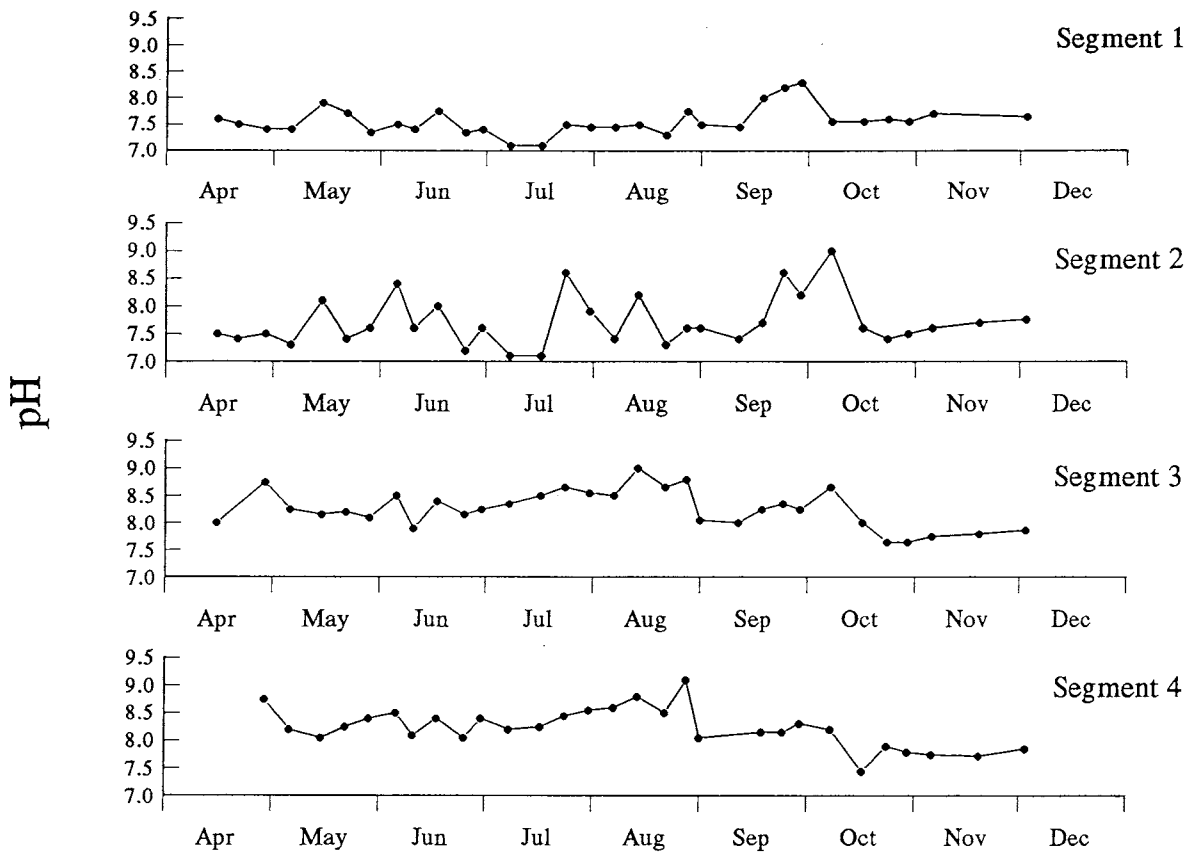


Figure 9. Bay and wetland bottom pH sampling results, 1992.

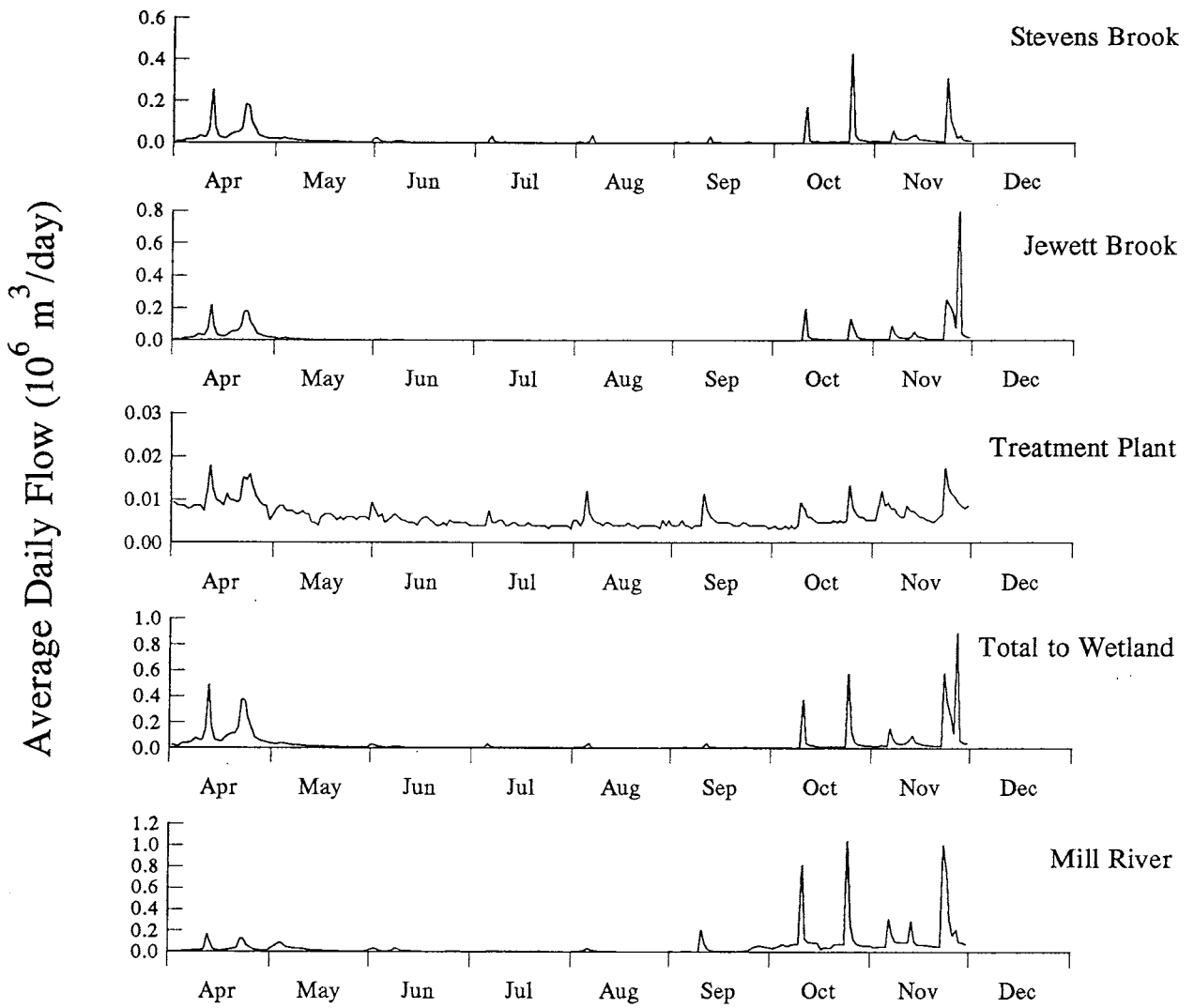


Figure 10. Average daily flow hydrographs for St. Albans Bay tributaries during 1992.

the summer months were low and stable, with only a few minor runoff events. Jewett Brook ran dry for much of the summer.

The chloride and phosphorus sampling results for the tributary and treatment plant stations are summarized in Table 5. The original intent was to use the results from the upstream stations on Jewett and Stevens Brooks (JEWE01 and STEV01) as a basis for loading estimates, supplemented with the much larger data set previously obtained at these stations under the Lake Champlain Diagnostic-Feasibility Study (Vermont DEC and New York State DEC, 1992) and the St. Albans Bay Rural Clean Water Program (1991). However, Table 5 shows that concentration differences existed between the upstream and downstream stations on Jewett and Stevens Brooks during the 1992 study period. Although these differences were generally not statistically significant, chloride and phosphorus loadings were estimated for this study based on samples obtained only at the most downstream stations (JEWE02 and STEV02) in order to provide the most accurate possible estimates of actual inputs to the wetland.

Chloride and total phosphorus loadings from the three tributary streams and the wastewater treatment plant were estimated using the FLUX tributary load estimation program (Walker, 1987, 1990). The FLUX program loading estimates were based on the daily average flow records shown in Figure 10 and the discrete concentration measurements obtained at each site.

Chloride and total phosphorus concentrations varied systematically as a function of flow in all three tributary streams, and so a concentration vs. flow regression method was used for load estimation (Walker, 1987, Method 6). Stratification with respect to flow interval was found not to be necessary to improve the precision of the loading estimates, except for total phosphorus at Jewett Brook, for which two flow intervals were used. No systematic variation between concentration and flow was observed for the wastewater treatment plant samples, and a load estimation technique based on the flow-weighted average concentration was used for this site (Walker, 1987, Method 2).

Mean flows and loadings from each tributary and the wastewater treatment plant during the April-November study period are compared in Figure 11, adjusted for the additional watershed area downstream of the flow gage stations according to the drainage area factors given in Table 3. As seen in Figure 11, the wastewater treatment plant contributed about 27% of the total phosphorus loading to the wetland during this period, and only about 16% of the phosphorus loading to the bay as a whole.

The appropriate concentration vs. flow relationships developed by the FLUX program were applied to the daily flow record for each stream and for the treatment plant to produce a daily time series of chloride and total phosphorus loadings for each site, as shown in Figures 12 and 13. The daily flow and loading values shown in Figures 10, 12, and 13 were used as inputs to the mass balance model, except that the daily chloride loading estimates to the wetland were first modified for reasons discussed below.

## **MODEL DEVELOPMENT**

### **Modeling Approach**

The central purpose of this study was to develop a phosphorus mass balance model for the St. Albans Bay and wetland system to identify the specific timing and location of major internal

Table 5. Summary of tributary and wastewater treatment plant sampling results.

<u>Parameter</u>	<u>Station</u>	<u>Mean (mg/l)</u>	<u>Standard Error (mg/l)</u>	<u>Sample Number</u>
Total Phosphorus	STEV01	0.192	0.035	18
	STEV02	0.145	0.015	27
	JEWE01	0.694	0.109	12
	JEWE02	0.723	0.135	17
	MILL01	0.149	0.032	13
	WWTF	0.519	0.102	8
	Dissolved Phosphorus	STEV01	0.082	0.010
STEV02		0.069	0.008	26
JEWE01		0.394	0.072	9
JEWE02		0.270	0.026	13
MILL01		0.061	0.011	11
WWTF				0
Chloride		STEV01	71.1	6.4
	STEV02	89.4	5.0	28
	JEWE01	23.1	1.0	12
	JEWE02	28.5	1.5	17
	MILL01	34.4	2.1	13
	WWTF	81.1	13.9	8

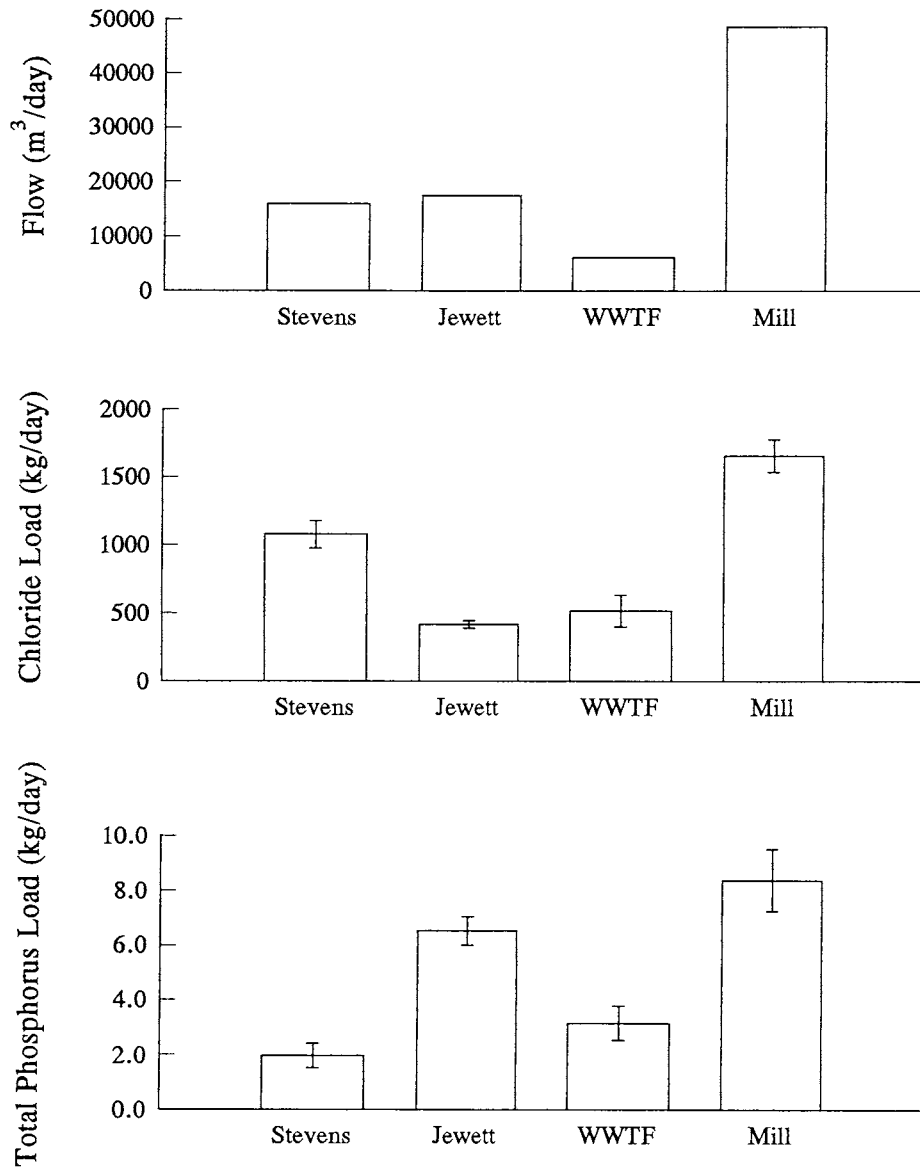


Figure 11. Mean flows, chloride loadings, and total phosphorus loadings to St. Albans Bay during the period of April to November 1992. Error bars indicate plus or minus one standard error.

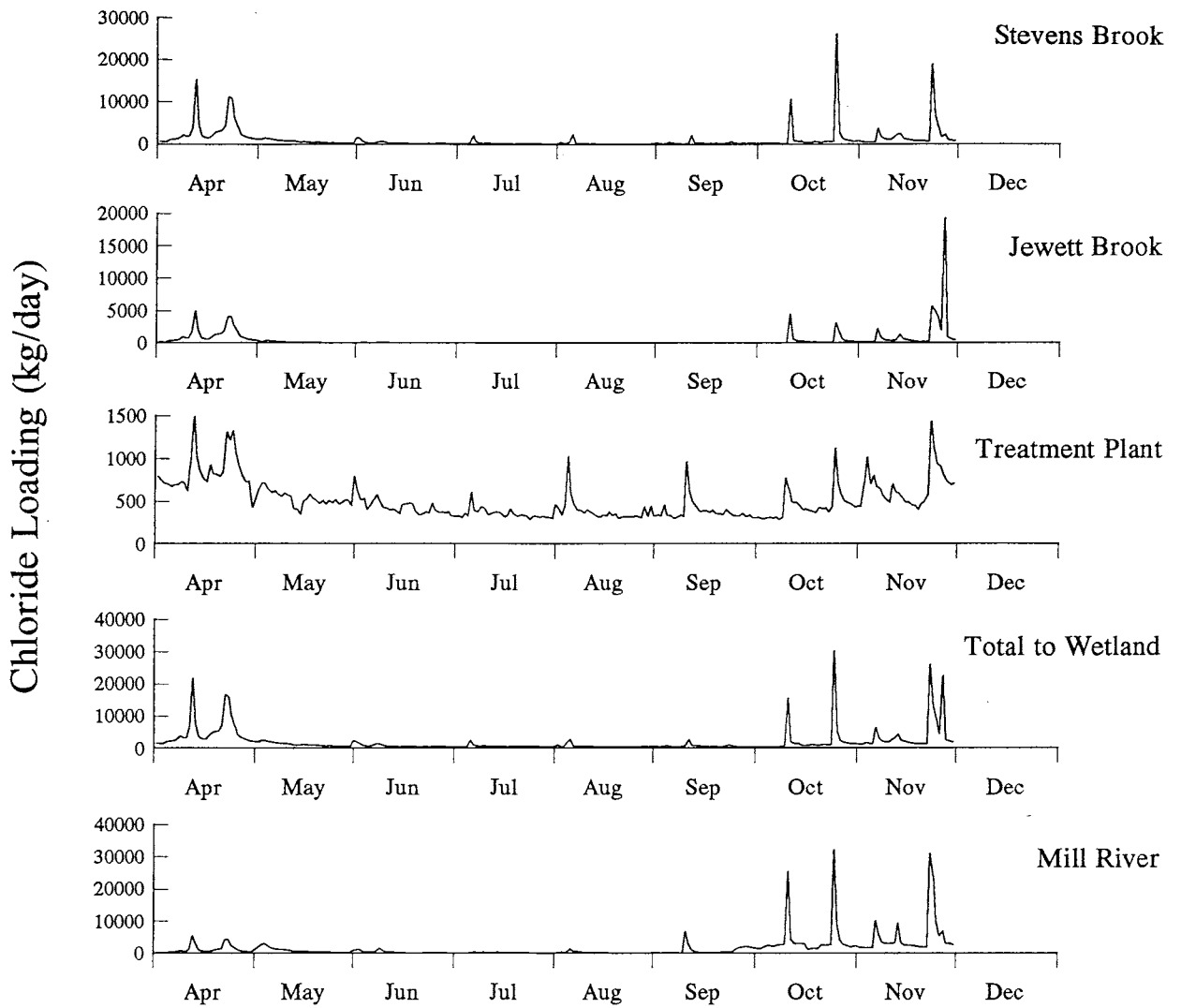


Figure 12. Daily chloride loadings to St. Albans Bay during 1992.

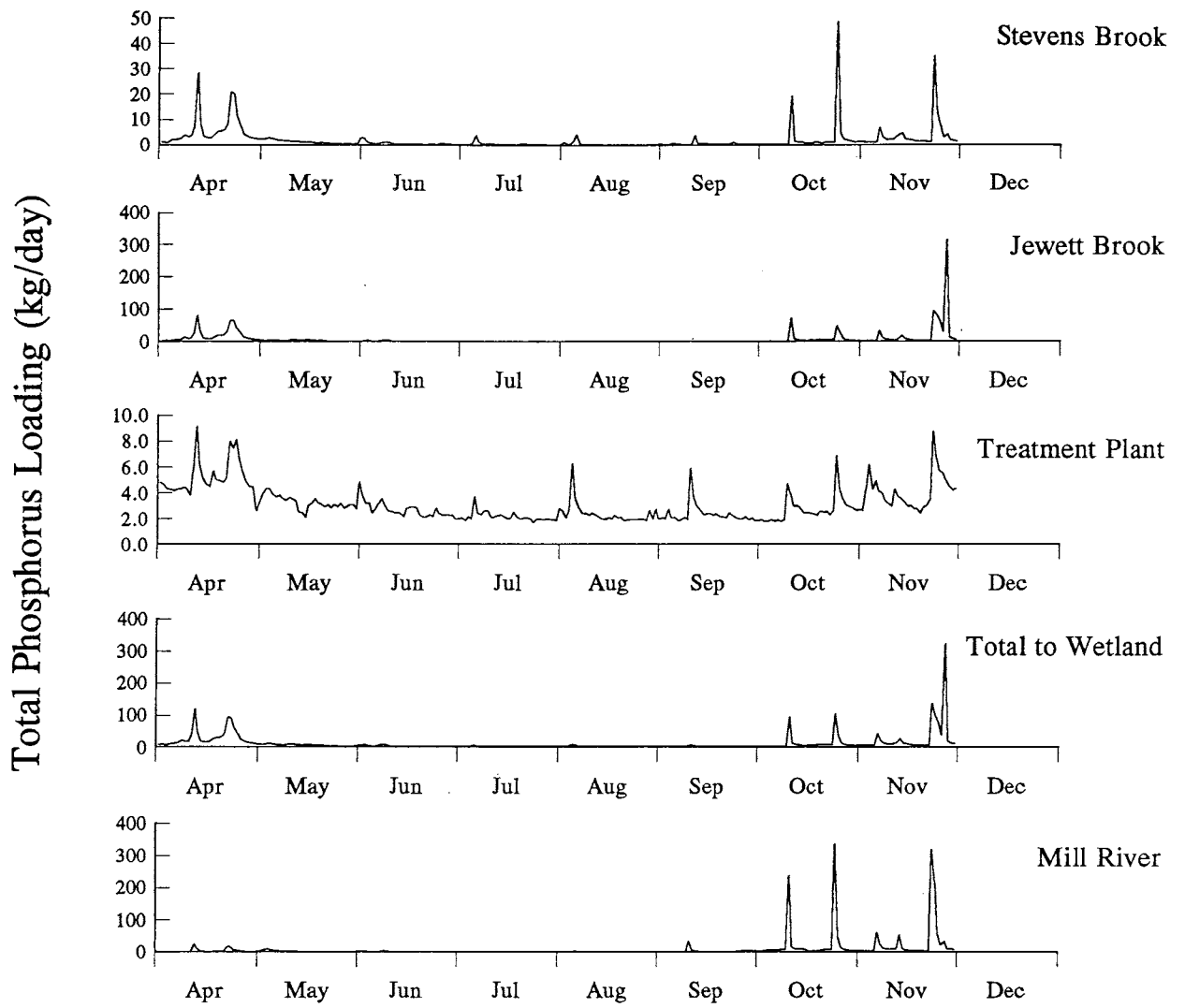


Figure 13. Daily total phosphorus loadings to St. Albans Bay during 1992.

loading episodes. A mass balance model that accounted for phosphorus variations in both space and time was therefore required. The modeling approach chosen for this analysis was a time-dependent, multiple-segment, one-dimensional embayment model, adapted from equations and methods described in Chapra and Reckhow (1983). The general mass balance equation for a single model segment can be written as follows:

$$\text{VOL}_i \frac{dc_{i,t}}{dt} = W_{i,t} + Q_{i-1,t} c_{i-1,t} - Q_{i,t} c_{i,t} + E_{i-1,t} (c_{i-1,t} - c_{i,t}) + E_{i,t} (c_{i+1,t} - c_{i,t}) - v_{i,t} A_i c_{i,t} \quad (1)$$

Where,

$\text{VOL}_i$  = volume of segment  $i$  ( $\text{m}^3$ )

$c_{i,t}$  = concentration in segment  $i$  at time  $t$  ( $\text{g}/\text{m}^3 = \text{mg}/\text{l}$ )

$W_{i,t}$  = external loading to segment  $i$  at time  $t$  ( $\text{g}/\text{day}$ )

$Q_{i-1,t}$  = cumulative advective inflow from all upstream segments at time  $t$  ( $\text{m}^3/\text{day}$ )

$Q_{i,t}$  = advective outflow from segment  $i$  at time  $t$  ( $\text{m}^3/\text{day}$ )

$E_{i-1,t}$  = diffusive exchange flow at upstream boundary of segment  $i$  at time  $t$  ( $\text{m}^3/\text{day}$ )

$E_{i,t}$  = diffusive exchange flow at downstream boundary of segment  $i$  at time  $t$  ( $\text{m}^3/\text{day}$ )

$v_{i,t}$  = net settling velocity in segment  $i$  at time  $t$  ( $\text{m}/\text{day}$ )

$A_i$  = area of segment  $i$  ( $\text{m}^2$ )

For the first (upstream) segment,  $Q_{i-1,t}$  and  $c_{i-1,t}$  and  $E_{i-1,t} = 0$ .

For the last (downstream) segment,  $c_{i+1,t} =$  constant outer lake background concentration.

A series of six linear mass balance equations were established for the multiple segment bay-wetland system shown in Figure 2, with one equation representing each model segment. The modeling procedures involved the simultaneous solution of these equations for particular variables or terms on a time-dependent basis, using finite-difference techniques.

One of the major challenges in developing a water quality mass balance model for St. Albans Bay is accounting for the transport of materials within the bay by wind-driven currents and other hydrodynamic mixing processes. Previous studies by Laible (1985) and Smeltzer (1991) have shown that such processes dominate the bay's hydrodynamics and strongly influence the spatial concentration gradients in the system. Exchange of phosphorus between adjacent segments of St. Albans Bay and the wetland was estimated for this study by means of mass balance calculations for the conservative substance chloride, for which the settling velocity terms ( $v_{i,t}$ ) are zero. By measuring chloride loadings and in-lake concentrations in each segment, the mass balance model equations could be solved for the exchange flow terms ( $E_{i,t}$ ) at each segment boundary.

Once the exchange flow terms were calibrated in this manner using the chloride data, the mass balance equations were used with the phosphorus loading and in-lake concentration data to solve for the apparent settling velocity terms ( $v_{i,t}$ ) in each segment. In this way, the model was used to identify the seasonal and spatial pattern of internal phosphorus settling or release within the bay-wetland system necessary to account for the observed phosphorus concentrations in each segment. The model was also used to simulate the effects of hypothetical in-lake treatments to reduce internal phosphorus loading in certain parts of the system by altering the calculated settling coefficients and solving the equations for the new phosphorus concentrations in each segment.

The mass balance models necessary for this analysis were constructed and run using the object-oriented modeling program STELLA II® (High Performance Systems, Inc., 1992). The STELLA II program object diagram for a six-segment mass balance model based on equation 1 is illustrated in Figure 14. The differential equations were solved within the STELLA II program using a finite (backward) difference approximation by Euler's method. An integration time step (dt) of 0.125 days was found to be the maximum time step that would yield stable results, and this integration step was used for all model runs. Model output was generated at one-day time intervals.

## **Evaluation of Model Terms**

### *Flow and Loading Terms*

Daily time series of flows and loadings of chloride and total phosphorus were available from the FLUX program estimates for the three major tributary streams and the wastewater treatment plant, as shown in Figures 10, 12, and 13. The continuous daily record of flows and loadings for the 244 day period of April 1 to November 30, 1992 was used as tabular input to the STELLA II model program. Flows and loadings to the first wetland segment from Jewett and Stevens Brooks and the wastewater treatment plant were summed for each day and consolidated into the model input terms  $Q_1$  and  $W_1$ , as shown in Figure 14. Daily flows and loadings from the Mill River were routed to segment 5 (see Figure 14).

### *Concentration Terms*

Chloride and total phosphorus concentration measurements were obtained at approximately a weekly frequency during the study period, as shown in Figures 3 and 4. However, it was apparent that concentrations in the system varied in response to changing loading or mixing conditions on a much finer time scale of days or less, particularly in the smaller volume wetland and inner bay segments with short residence times. For this reason, the time-dependent mass balance model was run in a manner that produced segment concentration output on a daily time interval.

To accommodate the need for daily model output data, the weekly total phosphorus concentration measurements were interpolated linearly to produce daily time series data for the concentrations in each segment. The interpolated values fell along the lines interconnecting the actual sampling measurements, as indicated in Figure 4. The daily phosphorus concentration time series data were used for the purpose of calculating daily net settling velocities, as described below.

### *Segment Morphometric Terms*

The volume and surface area of each model segment were treated as constant values, as given in Table 4. In reality, segment volumes varied during the study period in response to changing lake levels, as discussed earlier. However, the variation in segment volume was generally small in relation to other variables affecting the model terms containing the morphometric values. The effect on the model results of the simplifying assumption of constant segment morphometry was therefore expected to be minor.

The minor effect of the assumption of constant segment morphometry was confirmed during the phosphorus modeling process described below by creating an alternative STELLA II model

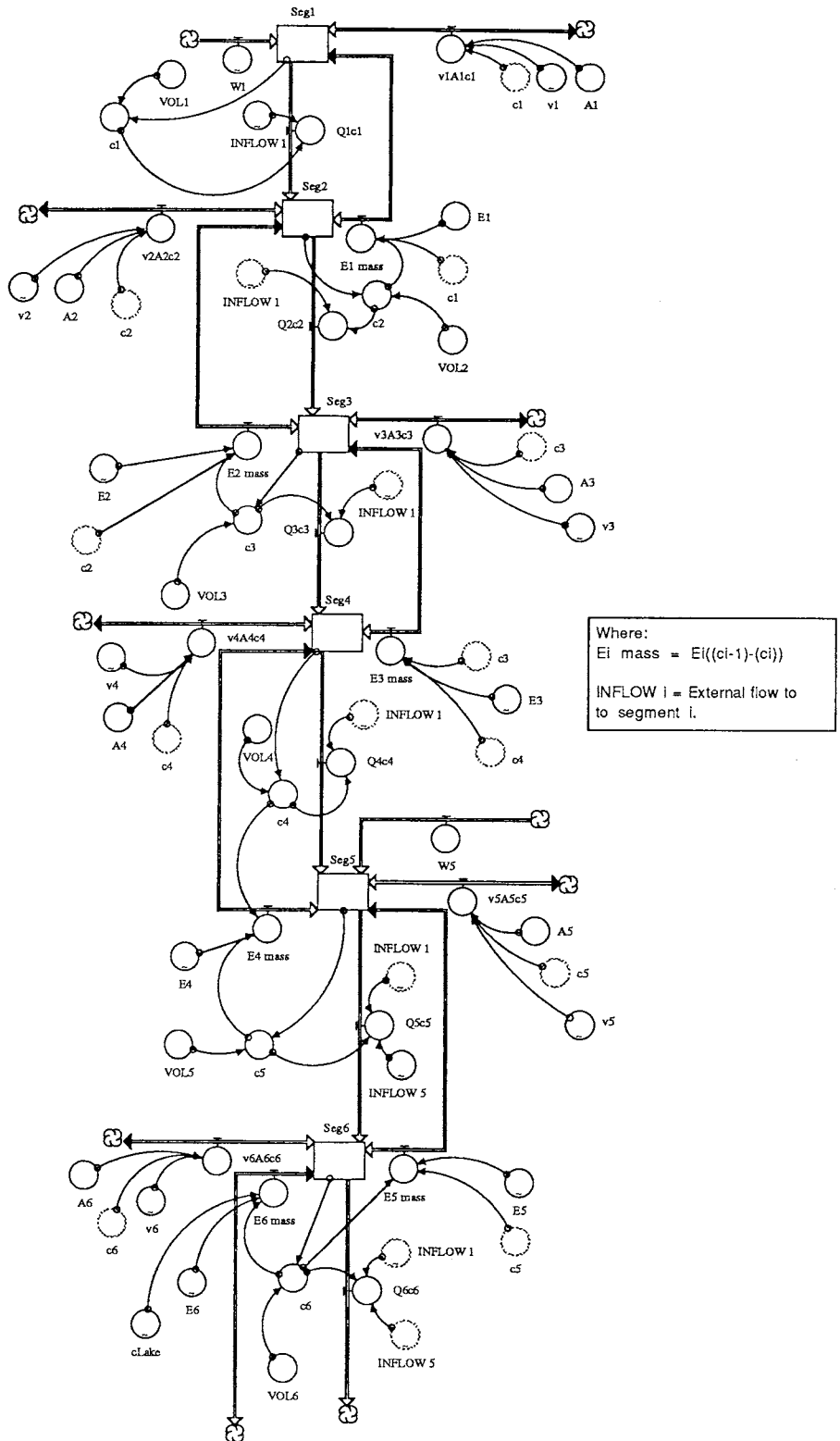


Figure 14. STELLA II model diagram.

which included daily varying segment volumes estimated from 1992 lake level records. The alternative model also added a "change in mass storage" term to equation 1 ( $dVOL_{i,t}/dt c_{i,t}$ ) which accounted for daily mass variations related to segment volume changes. When the variable segment volume model results were compared with the results of the constant volume model, the differences between the two models were found to be negligible. Therefore, the simpler constant volume approach was employed for all model runs.

### *Exchange Flow Calibration*

Equation 1 suggests that the exchange flow terms ( $E_i$ ) can be solved for on a daily basis if daily records of flows, chloride loadings, and segment chloride concentrations are available. An initial attempt was made to solve for the exchange flow terms on a daily basis, using the daily segment chloride concentration values interpolated from the weekly measurements. However, this approach proved to be unworkable because of measurement error or fine scale temporal variability in the data that on some days caused a departure from a monotonic spatial concentration gradient through the six wetland and bay segments to the outer lake. This situation led to cases of physically unrealistic negative or infinite exchange values when the model equations were solved.

To overcome this problem, the chloride model input terms (flows, loadings, concentrations) were averaged by month for each segment and the model equations (see equation 1) were algebraically solved on a steady-state basis ( $dc_i/dt = 0$ ) for each month during the study period. This procedure produced an estimate of a steady-state average exchange rate at each segment boundary for each month from April to November, 1992. One minor data smoothing adjustment was applied to the November mean chloride concentrations in segments 1 and 2 in order to achieve a monotonic exponential concentration decline through these two segments.

An additional modification was necessary to avoid some remaining calculated negative monthly exchange values for segment 1 within the wetland. This problem was caused by the fact that the monthly flow-weighted average chloride concentrations of the combined inflow to segment 1 (from Stevens Brook, Jewett Brook, and the wastewater treatment plant) estimated from the FLUX program procedures were generally close to, and sometimes slightly below the monthly average concentrations in segment 1, presumably as a result of data variability. To overcome this situation, alternative chloride loading rates to segment 1 were calculated using the observed monthly mean chloride concentrations in segment 1 as estimates of the flow-weighted mean concentration in the combined inflow. The segment 1 concentrations were multiplied by the monthly mean combined flow rate to the wetland ( $Q_1$ ) to estimate the monthly loading rates ( $W_1$ ) to this segment.

This procedure effectively assumes that water movement through segment 1 is solely an advective process and sets the monthly exchange rates for segment 1 ( $E_1$ ) at zero. The assumption that segment 1 is essentially a riverine flow-through environment is consistent with the sampling results showing generally minimal dilution of the inflow with water from downstream segments.

The monthly input data values and the calibrated monthly exchange flow estimates are listed in Table 6. Table 6 shows that averaging the chloride data by month using the calculation procedures discussed above resulted in non-negative and otherwise physically realistic estimates of the exchange flow rates for each segment. Lower exchange rates were estimated for the smaller wetland and inner bay segment boundaries. Higher exchange flows were generally

Table 6. Summary of monthly chloride data and exchange rate calibration results.

<u>Segment</u>	<u>Month</u>	<u>Concentration (mg/l)</u>	<u>Tributary Inflow (m<sup>3</sup>/day)</u>	<u>Tributary Loading (g/day)</u>	<u>Exchange Rate (m<sup>3</sup>/day)</u>
1	Apr	67.32	1.12 E5	7.56 E6	0
	May	66.78	1.81 E4	1.21 E6	0
	Jun	77.24	8.92 E3	6.89 E5	0
	Jul	82.13	7.55 E3	6.20 E5	0
	Aug	85.05	7.61 E3	6.47 E5	0
	Sep	91.35	7.78 E3	7.10 E5	0
	Oct	84.85	4.90 E4	4.15 E6	0
	Nov	57.57	1.07 E5	6.14 E6	0
2	Apr	58.40			2.36 E4
	May	49.40			9.61 E3
	Jun	48.00			7.64 E3
	Jul	43.53			9.47 E3
	Aug	48.12			8.46 E3
	Sep	55.83			6.86 E3
	Oct	56.20			5.44 E4
	Nov	39.03			1.18 E5
3	Apr	16.05			1.56 E6
	May	16.61			1.82 E5
	Jun	13.88			1.72 E5
	Jul	12.75			2.60 E5
	Aug	14.89			1.36 E5
	Sep	15.55			1.37 E5
	Oct	30.43			1.41 E5
	Nov	22.35			3.60 E5
4	Apr	12.35			6.99 E6
	May	11.63			1.23 E6
	Jun	10.60			1.11 E6
	Jul	10.74			1.11 E6
	Aug	10.95			9.45 E5
	Sep	11.23			6.23 E5
	Oct	11.54			2.31 E6
	Nov	11.93			6.41 E6
5	Apr	11.47	3.42 E4	1.23 E6	5.02 E6
	May	10.81	2.43 E4	8.93 E5	2.04 E6
	Jun	10.07	1.00 E4	3.80 E5	2.33 E6
	Jul	10.25	4.81 E3	1.85 E5	1.39 E6
	Aug	10.35	6.58 E3	2.50 E5	1.05 E6
	Sep	10.23	2.37 E4	8.50 E5	2.65 E6
	Oct	9.98	1.29 E5	4.28 E6	2.58 E7
	Nov	11.17	1.57 E5	5.19 E6	6.37 E6
6	Apr	10.05			7.32 E7
	May	10.00			4.80 E6
	Jun	9.69			1.48 E7
	Jul	9.76			4.97 E6
	Aug	9.64			4.68 E6
	Sep	9.77			6.54 E6
	Oct	9.73			2.68 E8
	Nov	9.85			2.91 E7
Lake	Apr	9.95			
	May	9.65			
	Jun	9.63			
	Jul	9.63			
	Aug	9.48			
	Sep	9.58			
	Oct	9.70			
	Nov	9.55			

indicated for the spring and fall months than for the summer period, as would be expected from typical seasonal variations in wind intensity.

The exchange flow terms given in Table 6 were used as input values for the STELLA II program model based on Figure 14 to check the accuracy of the chloride model calibration using the monthly exchange values. The exchange flows were entered as tabular input to the STELLA II program on a daily time step, but were held constant within each month. The exchange rate calibration results are shown in Figure 15.

Figure 15 shows that the monthly exchange rate calibration provided a reasonably accurate fit to the weekly chloride data. The calibration was poorest in the two wetland segments (1 and 2) where the weekly concentration measurements varied substantially from the calibrated model results. As discussed earlier, chloride and phosphorus concentrations in the wetland are influenced by hydrodynamic conditions that change rapidly, often on an hourly time scale. It would therefore have been unrealistic to expect an accurately detailed calibration fit of a model for these segments based on monthly average exchange rates and weekly sampling data.

The monthly calibrated exchange flows given in Table 6 were used as tabular input terms in the subsequent STELLA II modeling analysis for phosphorus. The exchange rate calibration results shown in Figure 15 indicate that much of the variation in the modeled daily phosphorus concentrations in the two wetland segments may be derived from errors in the exchange rate terms applied for a given date. However, phosphorus model variability derived from errors in the exchange rate estimates should be much less for the other bay segments.

## **PHOSPHORUS MODEL RESULTS**

### **Conservative Phosphorus Model**

The first step in the phosphorus modeling process was to run the STELLA II model illustrated in Figure 16 using the phosphorus model input data, with a net settling velocity ( $v_i$ ) of zero for all segments. The daily flows and phosphorus loading estimates shown in Figures 10 and 13 were used with the monthly calibrated exchange rates given in Table 6 as input data for the STELLA II model to predict the daily total phosphorus concentrations in each segment that would occur if phosphorus behaved conservatively in the system (i.e. no net internal release or sedimentation). The results of the conservative phosphorus model are compared with the actual sampling measurements in Figure 16.

Figure 16 shows that the conservative phosphorus model fit the observed data reasonably well during the period of April through June. However, the phosphorus peaks observed throughout the system in August and early November were not accounted for by the conservative model. This finding indicates that these phosphorus maxima were caused by net internal loading processes, and not by external loading sources, which were included in the conservative phosphorus model.

### **Net Settling Velocities**

The next phosphorus modeling step was to calculate the net settling velocity values necessary to produce the observed deviations in Figure 16 from the conservative model predictions for each segment. The net settling velocity terms ( $v_i$ ) in equation 1 were solved algebraically for

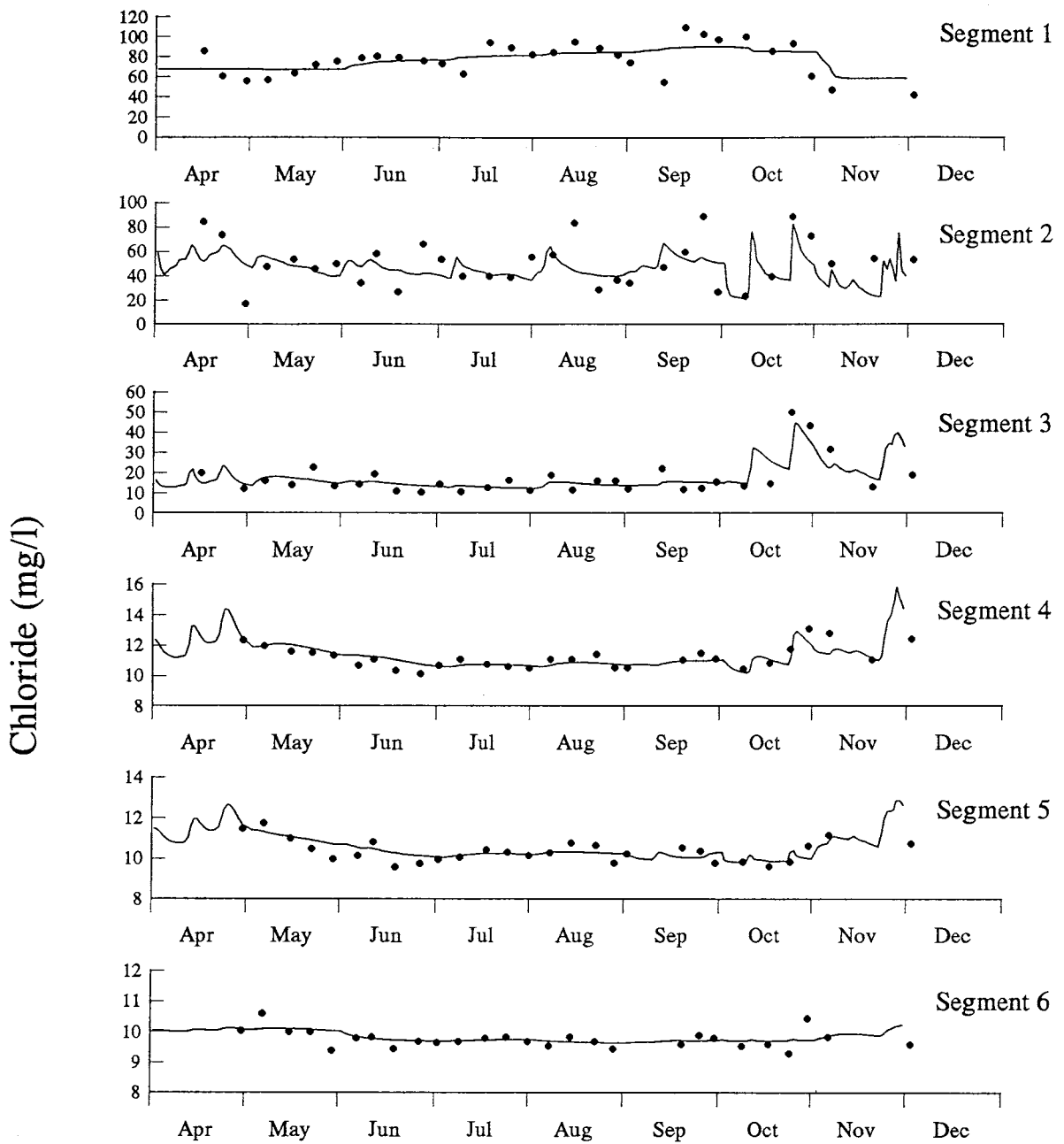


Figure 15. Chloride model calibration results, comparing the model-predicted values (lines) with the observed data (points).

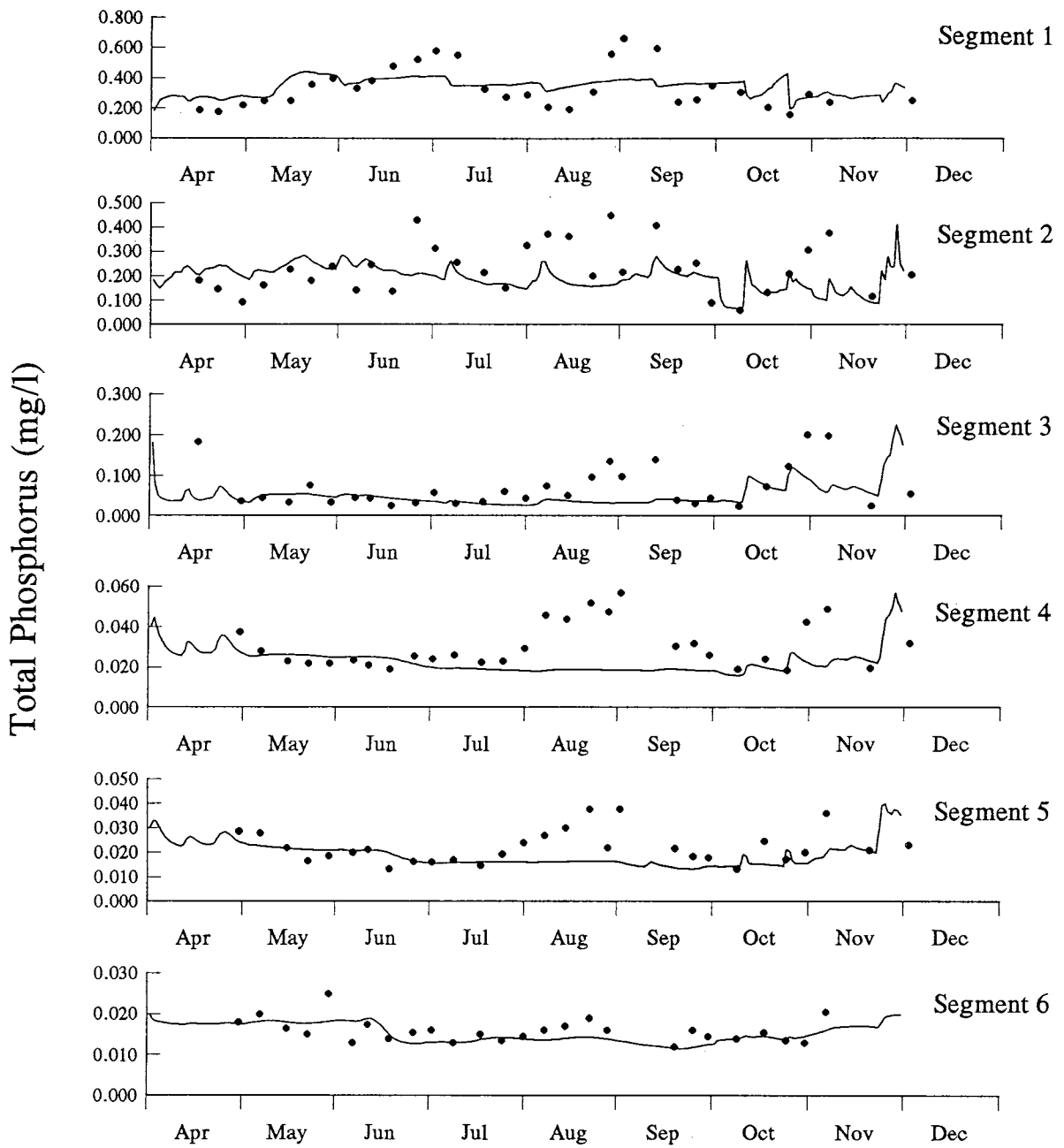


Figure 16. Conservative total phosphorus model results, comparing the model-predicted values (lines) with the observed data (points).

each segment on a daily time step. The daily flow and loading values shown in Figures 10 and 13 and the daily interpolated total phosphorus concentrations shown in Figure 4 were used for this purpose. The daily concentration rate of change in each segment was calculated as a finite backward difference (e.g.  $dc_{i,t} = c_{i,t} - c_{i,t-1}$ ). The April period was excluded from these calculations because of the scarcity of phosphorus sampling measurements during April in most segments (see Figure 4) and the resulting questionable accuracy in the calculation of daily  $dc_i/dt$  values for April.

The daily net settling velocities calculated in this manner include the net effects of all internal phosphorus loading and sedimentation mechanisms. The values also incorporate the net effects of measurement errors in all other model terms. In spite of these uncertainties, however, it was expected that when the calculated daily net settling velocity values were plotted for each segment, some informative spatial and seasonal patterns would emerge.

The calculated net settling velocity values are plotted for each segment in Figure 17. Negative settling velocities indicate a net internal phosphorus load into the water column. It can be seen from Figure 17 that the greatest internal phosphorus transfers, both positive and negative, occurred in the fall. However, there were no consistent spatial differences in the magnitude of the net settling velocities between the wetland, inner bay, and outer bay segments.

The August phosphorus maxima measured throughout the bay (see Figure 4) corresponded with a sustained period of negative settling rates (i.e. net internal loading to the water column) in most segments, as shown in Figure 17. However, the magnitudes of the net settling velocities in August were not particularly high, relative to rates indicated for other periods of the year. Apparently, the relatively moderate net internal loading rates that occurred during August acted in concert with the seasonally low exchange mixing rates (see Table 6) to produce the phosphorus peaks observed in the bay at that time.

The results shown in Figure 17 indicate that the wetland and inner bay sediments did not generally transfer phosphorus to the water at a higher rate than the middle and outer bay segments. It was not clear from Figure 17 how important internal loading from the wetland and inner bay segments was in the overall phosphorus mass balance for the system. To address this question, it was necessary to conduct additional modeling analyses simulating the response of the bay to internal loading reductions that might be achieved at specific locations in the system.

### **Simulation of Internal Loading Reductions**

The effect of a hypothetical in-lake treatment (e.g. sediment phosphorus inactivation with aluminum salts) to control internal phosphorus loading in the St. Albans Bay and wetland system was simulated by altering the settling velocity terms in certain segments to values more typical of most lakes. Studies reviewed by Reckhow and Chapra (1983) indicate that net settling velocities in lakes are typically in the range of 10-20 m/yr (0.03-0.06 m/day) when calculated from annual lake phosphorus budgets. However, highly variable and frequently negative settling velocities were indicated by the 1992 data for St. Albans Bay (see Figure 17), reflecting the presence of significant internal phosphorus loading.

It is difficult to predict quantitatively how a particular in-lake treatment to control internal loading might affect the internal phosphorus balance of a lake. However, for the purpose of the model simulations of potential internal phosphorus loading reductions in St. Albans Bay, it was assumed that an effective in-lake treatment would increase the net settling velocities in the

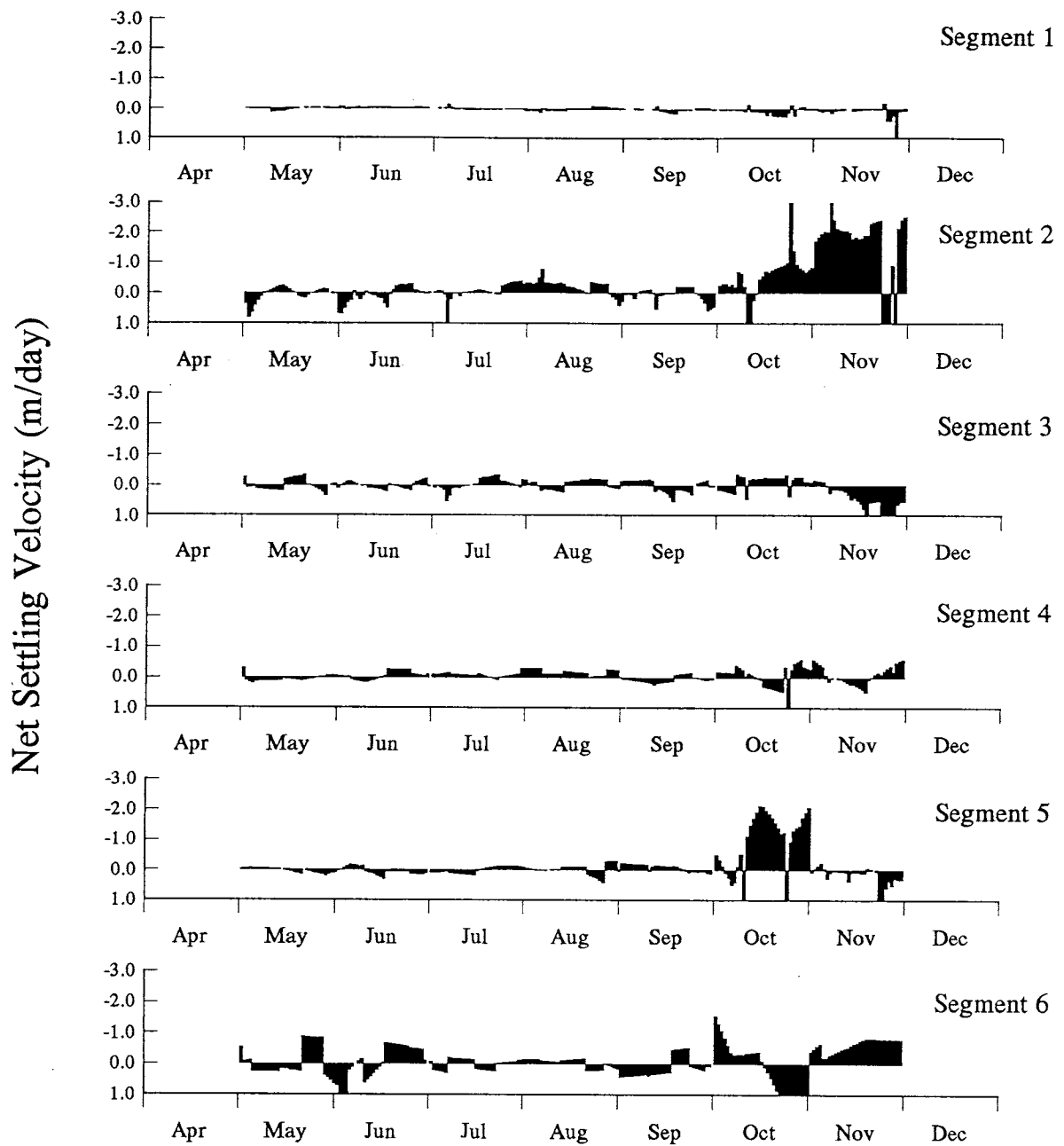


Figure 17. Calculated daily net internal settling velocities. Negative (upward) bars indicate net internal loading to the water column. Note that some values are out of range.

treated segments to values more typical of other lakes. The model was used in this manner to examine the sensitivity of phosphorus concentrations in the bay to realistically possible reductions in internal loading rates in various segments. The model simulations assumed that the net settling velocities would change to a constant value of 0.06 m/day throughout the year in the treated segments, and would remain unchanged in the untreated segments.

Model simulations were conducted with the assumption that an in-lake treatment would begin in the wetland where phosphorus concentrations are highest, and extend progressively outward into the bay to the point where the observed summer and fall phosphorus surges in the bay (Figure 4) were mostly eliminated. Accordingly, five model simulation scenarios involving progressive increases in the treated area were defined as follows:

- (1) No treatment
- (2) Treatment of the wetland segments 1 and 2 only.
- (3) Treatment of the wetland and innermost bay segments 1, 2, and 3
- (4) Treatment of segments 1, 2, 3, and 4.
- (5) Treatment of segments 1, 2, 3, 4, and 5.

The STELLA II phosphorus model for the bay was run for each of these five scenarios, varying the net settling velocities as indicated above. The model simulation results are shown in Figure 18. Treatment of the wetland only (scenario 2 above) would produce only minor benefit in reducing the summer or fall phosphorus maxima in the bay. The simulation results shown in Figure 18 indicate that these phosphorus peaks could be eliminated only by treating bay segments 1, 2, 3, and 4 as well (scenario 4). Treatment extending beyond segment 4 (scenario 5) would produce little additional benefit, however. An alternative model simulation run in which a higher settling velocity value of 0.12 m/day was applied to the treated segments produced essentially the same results, indicating that the findings were not highly sensitive to the choice of a settling velocity value used to represent the effects of an in-lake treatment.

The simulation results shown in Figure 18 indicate that the internal sources of phosphorus responsible for the observed summer and fall phosphorus peaks and algae blooms are not confined to the wetland. Control of the internal loading would involve treatment of a large portion of the inner bay, including the combined 2.8 km<sup>2</sup> (700 acre) area of segments 1, 2, 3, and 4. If internal loading in this area could be controlled to the extent simulated by the model in Figure 18, then average August total phosphorus concentrations in segments 3, 4, and 5 would be reduced to 0.014 - 0.017 mg/l, in contrast with current levels of 0.030 - 0.080 mg/l. Such a reduction would sharply lessen the magnitude of summer algae blooms in the bay.

### **Comparison of Internal and External Loading Rates**

For additional perspective, the total net internal loading rates in the wetland and bay were calculated and compared with the external loadings from the tributaries and wastewater treatment plant. The internal loading terms ( $v_i c_i A_i$  in equation 1) were calculated on a daily basis and summed across segments 1, 2, 3, 4, and 5 to produce an estimate of the total net daily internal phosphorus loading to the inner and middle portions of the bay. These internal loading values are compared with the sum of the external loadings to the system (from Figure 13) in Figure 19.

Figure 19 shows that the phosphorus dynamics of the system during the summer months are dominated by internal processes of sedimentation and phosphorus release from sediments or

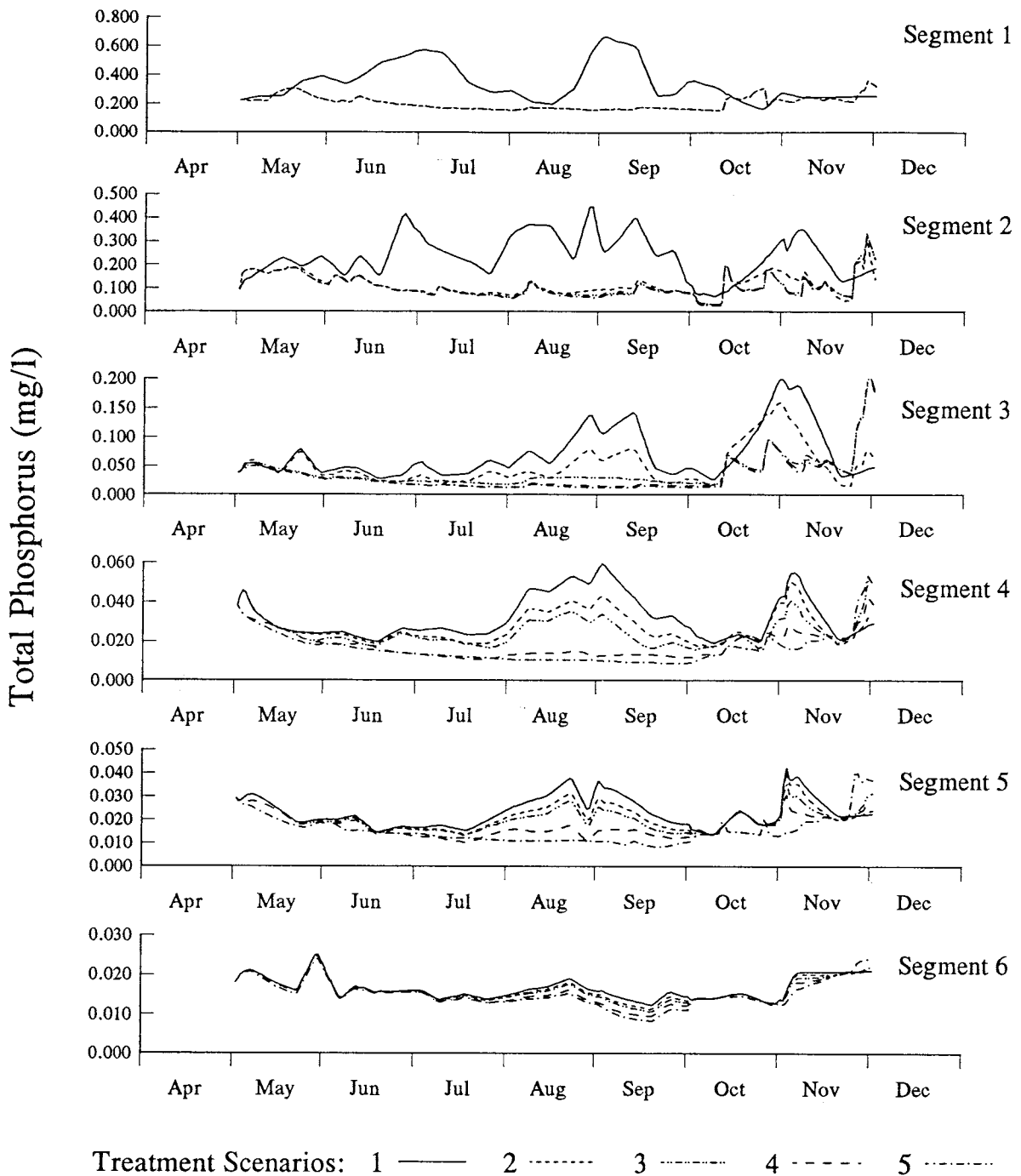


Figure 18. Model simulation results for five internal phosphorus loading reduction treatment scenarios:

1. No treatment.
2. Treatment of segments 1 and 2.
3. Treatment of segments 1, 2, and 3.
4. Treatment of segments 1, 2, 3, and 4.
5. Treatment of segments 1, 2, 3, 4, and 5.

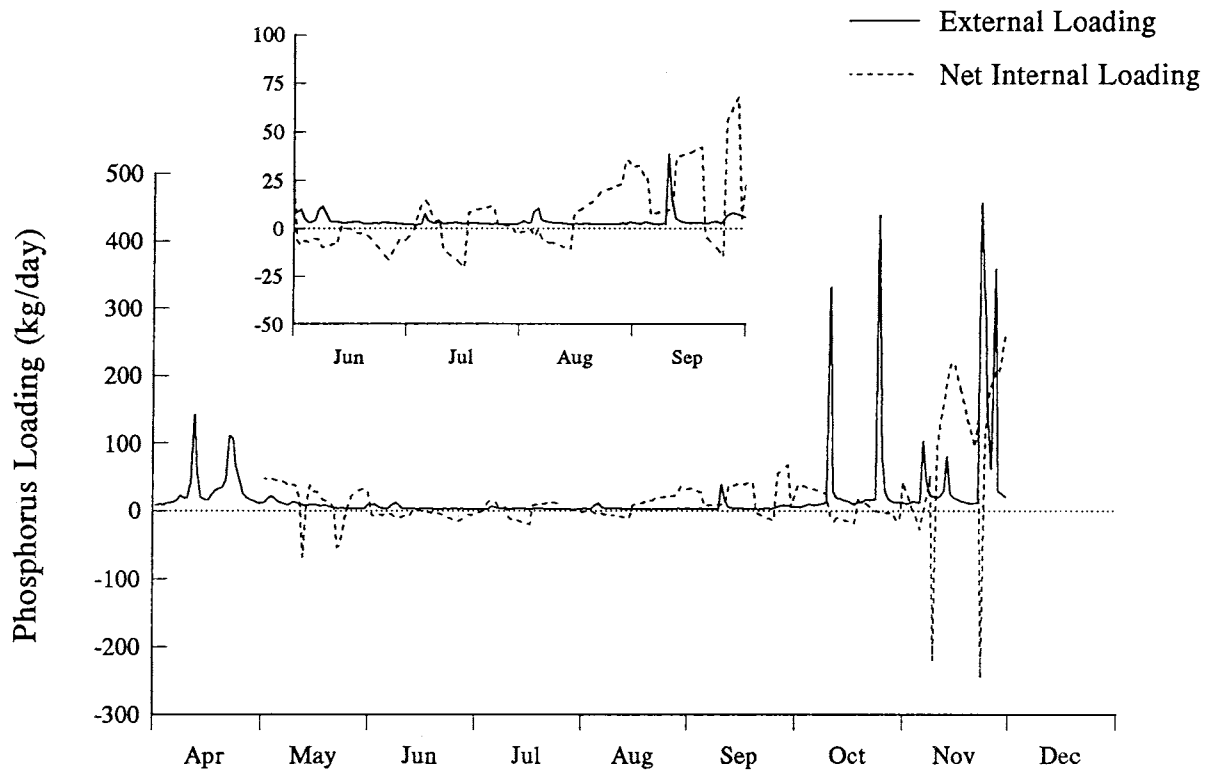


Figure 19. Comparison of external and net internal phosphorus loading rates to St. Albans Bay during 1992 for segments 01-05.

other sources. Consequently, further reductions in external loads would provide minimal near-term benefit in controlling the summer algae blooms that continue to plague the bay. However, to the extent that internal phosphorus sources are ultimately derived from external loadings over the long term, there is good reason to continue to reduce external loadings from the watershed. This is particularly true for nonpoint sources which now contribute the majority of the external phosphorus loading to St. Albans Bay (see Figure 11).

## DISCUSSION

### Mechanisms of Internal Loading

This study was not designed to provide a direct assessment of physical, chemical, or biological mechanisms responsible for internal phosphorus loading in St. Albans Bay. However, it was expected that the seasonal and spatial patterns of the internal loading terms estimated by the mass balance modeling analysis would provide some initial insight into the important underlying mechanisms.

Water column anoxia and associated reducing conditions at the sediment-water interface can cause extensive sediment phosphorus release in some lakes. However, anoxic conditions were rarely observed anywhere in the St. Albans Bay and wetland system during this or previous studies. Therefore, water column anoxia does not appear to be a significant factor causing internal phosphorus loading to St. Albans Bay.

Physical re-suspension of bottom sediments during wind events has been suggested as an important mechanism of internal phosphorus loading in St. Albans Bay (Laible, 1985). However, the sustained periods of net internal loading responsible for the late August phosphorus maxima observed throughout the bay (Figure 19) occurred at a time when wind velocities are typically at their seasonally low values, and generated primarily dissolved, rather than particulate phosphorus in the wetland and inner bay segments. These observations do not rule out physical re-suspension of sediments as an important mechanism causing the August phosphorus peaks, but they suggest that other factors must be involved as well.

Temperature and pH are two factors that have been found to be positively correlated with phosphorus release rates from shallow, aerobic lake sediments (Jensen and Andersen, 1992). Both temperature and pH reached seasonal maximum values in St. Albans Bay during August in 1992 (see Figures 8 and 9). Increased mineralization of sediment phosphorus at higher temperatures, coupled with hydroxide and phosphate ion exchange at high pH, may have caused the sustained period of net internal phosphorus loading observed during August.

Figure 19 shows that the highest internal phosphorus flux rates were indicated for the fall period when neither temperature nor pH were high. Apparently, other mechanisms were involved in the fall. Several major external loading events occurred during the fall period. The large internal loading and sedimentation rates calculated for the fall could have been partly derived from measurement errors in the estimation of these large external loadings.

Release of phosphorus from aquatic vegetation during periods of active growth and during periods of senescence has been identified as an important internal source of phosphorus in some lakes. St. Albans Bay contains extensive beds of submersed aquatic vegetation dominated by Eurasian watermilfoil (*Myriophyllum spicatum*) and *Elodea canadensis*, while the channelized area

of the Stevens Brook wetland is bounded by large areas of emergent plants such as cattail (*Typha angustifolia*) and the giant burreed, *Sparganium eurycarpum*, (St. Albans Bay Rural Clean Water Program, 1991; Clausen and Johnson, 1990).

Release of phosphorus stored in plant tissues into the water during the late summer and fall senescence may have contributed to the internal phosphorus loading events observed at that time (Figure 19). However, a previous STELLA modeling analysis of phosphorus uptake and release by emergent plants in the Stevens Brook wetland (Dingee and MacDonald, 1992) suggested that the peak release rates occurred in late September, somewhat earlier in the season than the maximum internal loading rates indicated for the wetland segments in Figure 17.

A further complication to the understanding of internal loading mechanisms in St. Albans Bay is the finding that internal loading rates calculated as net sedimentation velocities (or alternatively as areal release rates) were not particularly high during the critical August period of elevated water column phosphorus concentrations, in comparison with internal loading rates observed at other times during the sampling period (see Figure 17). Apparently, the seasonally low exchange mixing rates (Table 6) rendered the bay more vulnerable to internal loading during the summer and helped create the conditions that caused the phosphorus concentrations to climb during August.

The role of low summer mixing rates in producing the observed August phosphorus maxima was illustrated by a further model sensitivity run in which the calibrated exchange values (Table 6) were doubled in all segments for the months of July and August, without changing the calibrated settling velocities. This artificial simulation of enhanced summer mixing rates virtually eliminated the August phosphorus peaks in the model results (not shown). Long-term wind records confirmed that average wind velocities in the region are at their minimum values during late summer, as shown in Figure 20.

In summary, the modeling analysis did not reveal any single dominant mechanism for the internal phosphorus loading to St. Albans Bay. Several processes may be important at various times of the year, including sediment re-suspension, increased mineralization of sediment phosphorus at high summer temperatures, phosphate release by metal hydroxide ion exchange at high pH, and secretion from growing or senescent aquatic macrophyte populations. Apparently, one or more of these processes acted in concert with the low summer water exchange rates to cause peak phosphorus and algae levels during the height of the summer recreational season.

### **Management Implications**

A central purpose of this study was to determine whether the internal phosphorus loading to the system was confined primarily to the wetland, or whether the bay sediments and other sources were involved. If the major source was localized in the wetland, then some form of treatment to control the internal loading might be much more feasible and less costly. However, the model simulation results (Figure 18) showed that treating the wetland alone would do little to prevent the summer phosphorus peaks and algae blooms observed in the bay. Treatment of the entire 700 acre inner and middle bay area would be necessary to gain control over the internal loading problem.

A review of lake restoration methods by Cooke *et al.* (1986) describes several techniques that have been effective in reducing internal phosphorus loading in some lakes. Of the methods described by Cooke *et al.*, sediment phosphorus inactivation with aluminum salts (e.g. alum) is

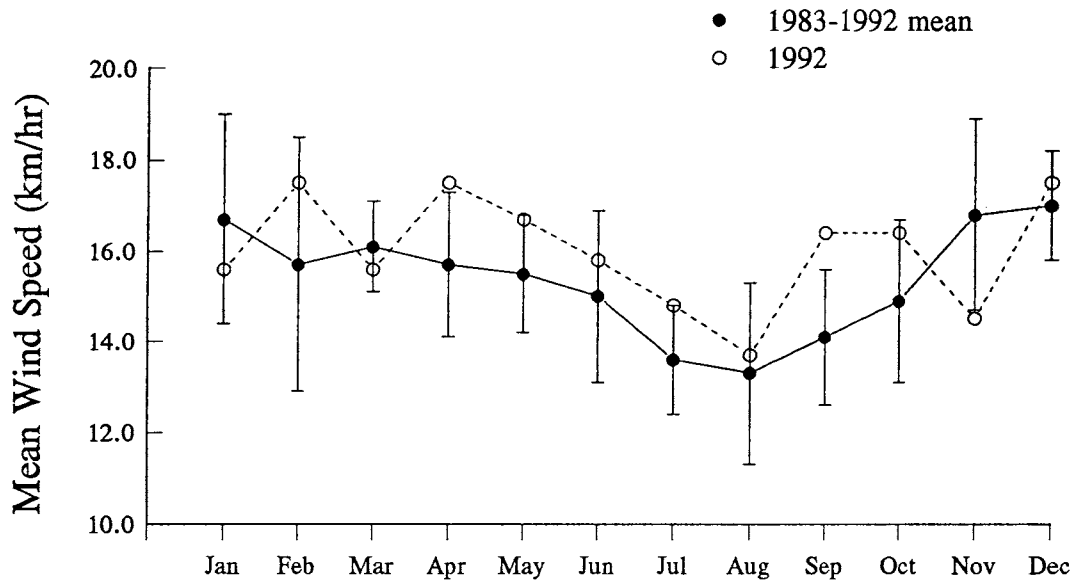


Figure 20. Monthly mean wind speeds recorded at the Burlington, Vermont International Airport during 1983-1992 (National Oceanic and Atmospheric Administration data). Error bars show standard deviations of the 1983-1992 monthly means.

probably the most applicable to St. Albans Bay. Techniques involving aeration or sediment oxidation would probably not be effective because the water column in St. Albans Bay is already well aerated. Removal of phosphorus rich sediments by dredging can sometimes reduce internal phosphorus loading, but this procedure involves high cost and carries major environmental risks and limitations, including sediment re-suspension, release of toxic substances bound in the sediments, destruction of benthic habitat, destruction of wetland habitat, and problems associated with disposal of dredged materials. Harvesting of aquatic plants has not been effective as a means of phosphorus removal in most situations.

A preliminary plan for treating St. Albans Bay with aluminum salts provided to the Town of St. Albans by Sweetwater Technology Corp. (T. Eberhardt, personal communication, 8/28/91) estimated the treatment cost at \$500-750 per acre. An alum/aluminate treatment of Lake Morey, Vermont in 1986 cost \$538 per acre including some donated chemical (Smeltzer, 1990). The cost to treat a 700 acre region of St. Albans Bay might therefore be in the range of \$350,000 to \$525,000.

One important issue relating to the feasibility of an aluminum treatment of St. Albans Bay sediments is the question of whether the aluminum hydroxide floc would be re-suspended and lost from the system as a result of wind induced mixing. Aluminum treatments of shallow lakes have shown a mixed record of successes and failures (Welch *et al.*, 1988; Cooke *et al.*, 1993), and re-suspension and re-distribution of the aluminum floc has sometimes been a factor in treatment failure. In St. Albans Bay, unlike a closed lake, there is potential for loss of aluminum floc out of the system entirely by transport out the bay mouth. A further concern would be for the potential toxicity of aluminum to fish and other organisms at the high pH levels existing in St. Albans Bay. Similar conditions in Lake Morey caused elevated dissolved aluminum concentrations in the water following treatment, and may have resulted in sub-lethal toxic effects on yellow perch in that lake (Smeltzer, 1990).

For these reasons, any aluminum treatment under consideration for St. Albans Bay should be preceded by feasibility studies that include pilot scale chemical applications to limited areas to determine the stability of the floc layer and the potential for toxicity. Feasibility studies for aluminum treatment should also include a complete assessment of all environmental impacts to the bay or the wetland, and provide all information necessary for permitting and other regulatory requirements. It is possible that the feasibility studies would conclude that an aluminum treatment of St. Albans Bay would be ineffective or harmful and that no good treatment alternative exists.

## CONCLUSIONS

Water quality monitoring data for St. Albans Bay indicate that phosphorus concentrations in the bay have remained excessively high in the years since the wastewater treatment plant was upgraded in 1987. Maximum phosphorus levels and algae blooms have consistently occurred during August in recent years, at the height of the summer recreational season. The modeling analysis conducted for this study showed that the 1992 August phosphorus peak was caused by internal phosphorus sources, and not by external loadings from the watershed.

The central question addressed by this study was whether the major portion of the internal phosphorus loading to the system was confined to sources within the Stevens Brook wetland, or whether sediments and other sources in the bay were dominant. The model simulation analysis

provided a clear answer to this basic question. Control of internal phosphorus loading in the wetland alone would do little to reduce the summer phosphorus maxima and algae blooms in the bay. Most of the internal phosphorus loading is derived from sources within the bay.

The sampling data and modeling analysis revealed no single source or location of internal phosphorus loading to the system that was a dominant mechanism. It appears that the August phosphorus maxima in the bay are caused by a complex interaction of factors occurring over an extensive area of the wetland and bay. High temperatures and pH values during late summer, combined with seasonal minimum wind velocities and mixing rates, may interact to render the bay especially vulnerable to the effects of aerobic sediment phosphorus release at that time of year.

The management implications of these findings are that an in-lake treatment to control internal phosphorus loading in St. Albans Bay would need to be a large scale project involving about 700 acres of bay and wetland area. Sediment phosphorus inactivation using aluminum salts appears to be the most practical treatment technology for consideration. Preliminary cost estimates for an aluminum treatment of St. Albans Bay are in the range of \$350,000 to \$525,000. However, additional feasibility studies must be conducted before an aluminum treatment can be seriously considered. Feasibility studies are needed to determine the potential for re-suspension and loss of aluminum floc from the shallow, well-mixed areas of the system, and to evaluate possible adverse effects of a treatment on the bay, wetland, and human users, for environmental permitting purposes. Feasibility studies may find that no practical treatment alternative exists.

A decision on proceeding with feasibility studies for an extensive in-lake treatment to control internal phosphorus loading to St. Albans Bay should consider the long-term historical perspective provided by Martin *et al.* (1994) regarding the time that may be required for the bay to respond to the wastewater treatment plant loading reductions without further management intervention. Furthermore, to be optimally effective, any future in-lake treatment to control internal loading must be coupled with renewed efforts to reduce the continued high rates of nonpoint source phosphorus loading to the system as well.

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