A PHOSPHORUS BUDGET, MODEL, AND LOAD REDUCTION STRATEGY FOR LAKE CHAMPLAIN

Lake Champlain Diagnostic-Feasibility Study Final Report

Vermont
Department of Environmental Conservation

New York State
Department of Environmental Conservation

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Final Report

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The phosphorus load reduction strategy for Lake Champlain recommended in this report developed as a result of extensive review and discussion by the Lake Champlain Management Conference, the Lake Champlain Technical Advisory Committee and its Eutrophication and Nonpoint Source Subcommittees, and the staff of the Lake Champlain Basin Program.

The cover photograph was taken by Neil Kamman.

INTRODUCTION

Description of Lake

Lake Champlain is an interstate and international body of water that is one of the largest freshwater lakes in North America. Like the North American Great Lakes, Lake Champlain is part of the St. Lawrence River drainage system. Lake Champlain is a morphologically complex water body with numerous embayments and well-defined segments, as shown in Figure 1. A wide variety of physical and water quality conditions exists within the lake. Detailed discussions of the lake's limnology, environment, and cultural uses can be found in Myer and Gruendling (1979) and Lake Champlain Basin Study (1978, 1979). Morphometric data for Lake Champlain and its drainage basin are given in Table 1.

History of Eutrophication Management of Lake Champlain

Phosphorus enrichment and eutrophication has been a major water quality management issue on Lake Champlain for many years. Efforts to deal comprehensively with eutrophication issues in Lake Champlain began in the 1970's. Lake Champlain was included in the National Eutrophication Survey during which an initial phosphorus budget for the lake was developed by the U.S. Environmental Protection Agency (1974). A study by Henson and Gruendling (1977) produced a second estimate of phosphorus loadings to the lake, based on a tributary sampling program. The results of these studies were reviewed and extended by Bogdan (1978) and incorporated into the Lake Champlain Basin Study (1979), a comprehensive data compilation and planning effort for the lake that addressed eutrophication and several other natural resource issues.

The Lake Champlain Basin Study (1979) produced a plan that included a number of phosphorus management recommendations designed to hold constant or reduce phosphorus inputs to the lake until 1990. The Lake Champlain Basin Study supported existing state phosphorus control policies including the continuation of phosphorus detergent bans in Vermont, New York, and the Province of Quebec, and the construction of phosphorus removal facilities at a number of Vermont municipal wastewater treatment plants located near the lake. The Lake Champlain Basin Study also recommended that nonpoint source phosphorus control programs be pursued, particularly for agricultural runoff.

In the years since the Lake Champlain Basin Study was completed, all the major eutrophication management recommendations have been pursued and at least partly implemented. Phosphorus detergent bans continue to exist basin-wide. Phosphorus removal facilities have been constructed at most of the originally designated Vermont municipal wastewater treatment plants, and also at a small number of municipal and industrial discharges in New York and Quebec. Phosphorus removal requirements in Vermont were legislatively expanded in 1992 to include 29 municipal treatment plants in Vermont, with full implementation expected to occur over the next several years. In the nonpoint source area, the U.S. Soil Conservation Service has initiated agricultural water quality protection projects in several sub-watersheds of Lake Champlain.

The phosphorus management actions taken following the completion of the Lake Champlain Basin study probably accomplished the purpose of preventing further general increases in phosphorus loadings and eutrophication of the lake until 1990. However, current phosphorus levels in Lake Champlain are too high in some areas (Smeltzer, 1992) and substantial reductions are needed.

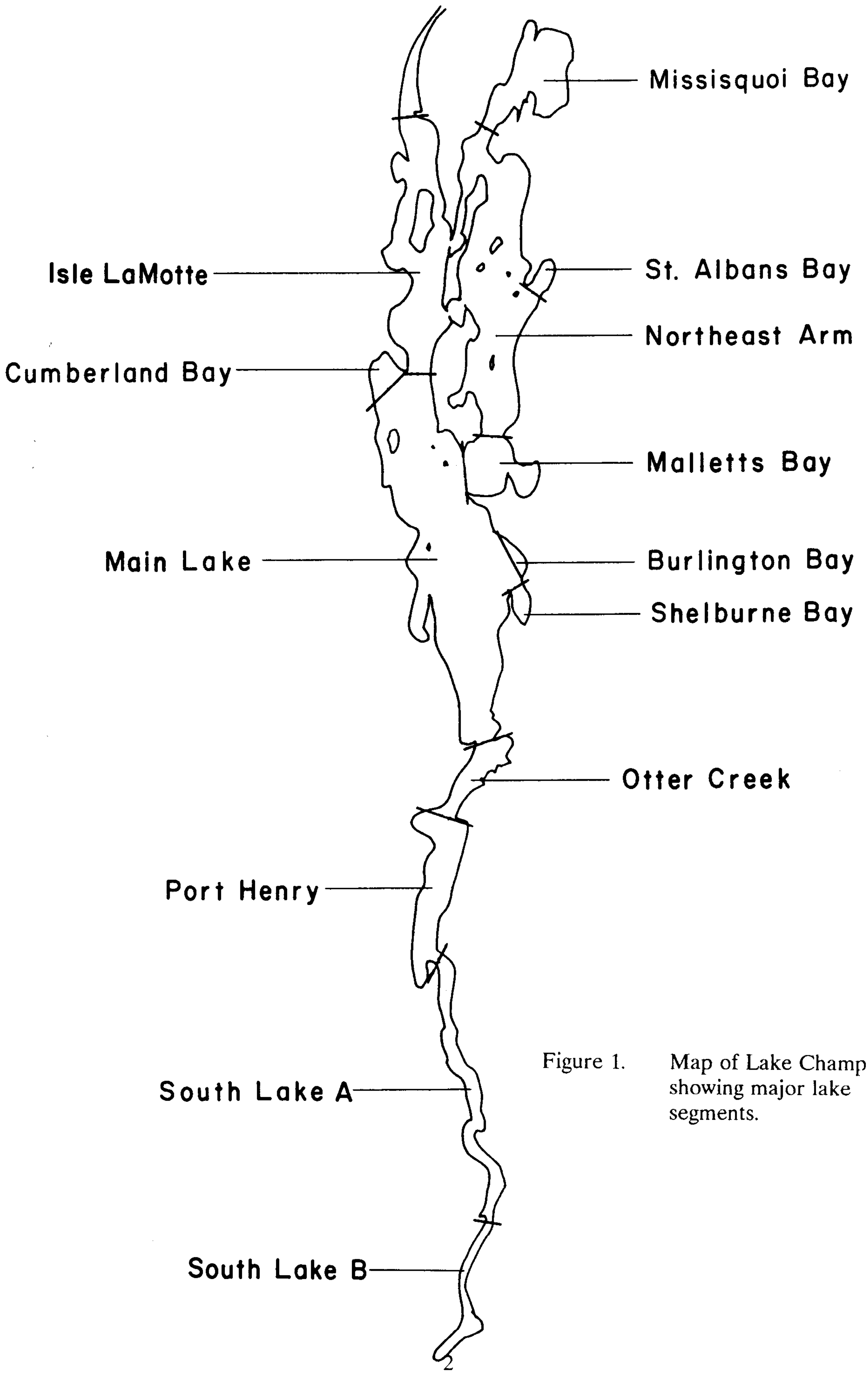

Figure 1. Map of Lake Champlain showing major lake segments.

Table 1. Morphometric data for Lake Champlain and its drainage basin, from Myer and Gruendling (1979).

Length	**170 km**
Maximum Width	**20.2 km**
Maximum Depth	**122 m**
Mean Surface Elevation (NGVD)	**29.0 m**
Shoreline Length	**808 km**
Surface Area (excluding islands)	**1,130 km^2**
Surface Area (including islands)	**1,269 km^2**
Volume	**25.8 km^3**
Mean Depth	**22.8 m**
Drainage Area (excluding lake surface)	**19,881 km^2**
Drainage Area (at lake outlet)	**21,150 km^2**

Cooperative interstate and international efforts on Lake Champlain were renewed in 1988 when the States of Vermont and New York and the Province of Quebec signed a Memorandum of Understanding on Environmental Cooperation on the Management of Lake Champlain. A Vermont and New York Workplan prepared at the time of this agreement (Vermont Agency of Natural Resources and New York State Department of Environmental Conservation, 1988) addressed eutrophication and other lake management issues. The 1988 work plan recognized that eutrophication of the general body of Lake Champlain is the result of the cumulative impact of many individually small phosphorus sources, with no single source being dominant on a lakewide basis. Management of such cumulative impacts must involve the development of policies to be broadly applied throughout the basin aimed at various categories of phosphorus sources such as municipal and industrial discharges, nonpoint source runoff from agricultural and urban areas, and other sources.

The 1988 Vermont and New York work plan outlined a phosphorus management strategy for Lake Champlain involving the following steps:

1. **Establish numeric in-lake eutrophication water quality criteria for each segment of the lake.**

2. **Measure the phosphorus loadings to the lake and develop a whole-lake water quality model predicting the lake's eutrophication response to its phosphorus loadings.**

3. **Use the lake model to conduct a point and nonpoint source phosphorus load allocation and set basin-wide phosphorus management policies and priorities in a manner to attain the in-lake water quality criteria.**

In 1990, the federal Lake Champlain Special Designation Act was passed by the United States Congress and signed into law. The Act established the Lake Champlain Management Conference and charged it with the responsibility of preparing a Comprehensive Pollution Prevention, Control, and Restoration Plan for Lake Champlain. The plan produced by the Lake Champlain Managment Conference (1996) endorses the need for phosphorus reduction through the steps outlined above, and uses the present Lake Champlain Diagnostic-Feasibility Study as a basis for determining the phosphorus load reductions needed for Lake Champlain.

The first step in the phosphorus management process for Lake Champlain was largely accomplished by the signing in 1993 of a New York, Quebec, and Vermont Water Quality Agreement on in-lake phosphorus criteria for Lake Champlain. This Water Quality Agreement resulted from the report and recommendations of the Lake Champlain Phosphorus Management Task Force (1993) which reviewed current eutrophication management programs and policies in New York, Quebec, and Vermont. The Water Quality Agreement endorsed a specific set of interim total phosphorus concentration criteria for 13 segments of the lake in order to establish a consistent approach to phosphorus management in Lake Champlain among the three jurisdictions. The in-lake phosphorus criteria are listed in Table 2.

The 1993 Water Quality Agreement also committed New York, Vermont, and Quebec to the development of basin-wide phosphorus load allocations for both point and nonpoint sources using the data and modeling capability developed by the present Lake Champlain Diagnostic-Feasibility Study. Phosphorus load allocations for Lake Champlain, and any future modifications to the in-lake criteria which guide the allocation process, will be made with joint participation from all three jurisdictions.

Table 2. New York, Quebec, and Vermont interim in-lake total phosphorus concentration criteria for 13 segments of Lake Champlain, as adopted in the 1993 Lake Champlain Water Quality Agreement (Lake Champlain Phosphorus Management Task Force, 1993). Lake segments are as shown in Figure 1.

Lake Segment	Total Phosphorus Criterion (Fg/l)
Main Lake	10
Malletts Bay	10
Shelburne Bay	14
Burlington Bay	14
Cumberland Bay	14
Northeast Arm	14
Isle LaMotte	14
Otter Creek	14
Port Henry	14
St. Albans Bay	17
Missisquoi Bay	25
South Lake A	25
South Lake B	25

Previous Studies

There have been several previous efforts to estimate phosphorus loadings to Lake Champlain. The U.S. Environmental Protection Agency (1974) sampled 21 tributaries to Lake Champlain once per month for a one year period during 1972-1973 and used the sample data to calculate loading rates. Henson and Gruendling (1977) identified 34 tributaries with drainage areas greater than 10 square miles distributed along both sides of the lake, and developed phosphorus loading estimates based on an independent sampling effort conducted during 1970-1974. Bogdan (1978) developed phosphorus export estimates for the Lake Champlain Basin based on population and land use characteristics in the basin, and compared the estimates from the previous studies. Bogdan estimated the total loading of phosphorus to Lake Champlain to be in the range of 536-804 mt/yr (metric tons per year), with approximately half of the total derived from point sources.

The earlier tributary sampling programs conducted by the U.S. Environmental Protection Agency (1974) and Henson and Gruendling (1977), while providing the basis for some first estimates of total lake loading, involved too few samples per tributary to avoid problems of poor precision and possible bias. The reported number of phosphorus samples obtained per tributary was generally less than 20, and the loading calculation method involving the use of median phosphorus concentrations could have introduced considerable bias into the estimates by failing to adequately account for variations in phosphorus with flow rate. Preferred methods for calculating mass loadings to lakes generally require more samples appropriately distributed over the full range of stream flow conditions with particular emphasis on sampling the high flow events (Verhoff *et al.*, 1980).

Van Benschoten (1979) evaluated the suitability of phosphorus mass balance models for use on Lake Champlain, given the data existing at the time. Van Benschoten concluded that there was insufficient information to support the development and use of a phosphorus model for lake management decision-making purposes on Lake Champlain. Information needs identified by Van Benschoten included a better knowledge of water movements and phosphorus transport between lake segments, more data on in-lake phosphorus concentrations, more precise estimates of hydraulic and phosphorus loadings including their seasonal variation, a better understanding of phosphorus sedimentation and other internal processes in various regions of the lake, and more knowledge of the role of the various forms of phosphorus in the eutrophication process in the lake. The present study is designed to fill many of the data and modeling needs identified by Van Benschoten (1979).

Scope of This Study

The Lake Champlain Diagnostic-Feasibility Study was initiated in 1989 as a joint project between the States of Vermont and New York. The project was funded by the U.S. Environmental Protection Agency's Clean Lakes Program and the States of Vermont and New York, with cooperative assistance supplied by the U.S Geological Survey.

The general purpose of a Clean Lakes Program diagnostic-feasibility study is to determine the nature and causes of water quality problems in a lake and evaluate the feasibility of alternative lake restoration methods. The present Lake Champlain Diagnostic-Feasibility Study was focused on the issue of phosphorus and eutrophication, as described in the original project proposal (Vermont Department of Environmental Conservation and New York State Department of Environmental Conservation, 1989). Most of the project effort was devoted to the measurement of phosphorus loadings to Lake Champlain and the modeling analysis of the lake's water quality response to phosphorus loadings.

Phosphorus loadings to Lake Champlain were measured over a two year period by frequent sampling of sources including tributary inflows, direct wastewater discharges, and precipitation to the lake surface. A network of new and pre-existing stream flow gages was maintained by the U.S. Geological Survey to support the tributary loading measurements. An extensive lake sampling program was conducted concurrently with the loading measurements to provide a basis for the development of a whole-lake phosphorus model for Lake Champlain. Loadings and in-lake concentrations of chloride were also determined during the study in order to quantify in-lake mixing and transport of materials. Additional data were obtained on a broader range of limnological parameters in complete vertical profile at selected deep water stations to supplement basic water quality information on the lake.

The data were used to establish a phosphorus budget for Lake Champlain which identifies and ranks all the major phosphorus sources to the lake. A whole-lake phosphorus mass balance model was then developed to analyze the lake's water quality response to its phosphorus loadings. The model was applied to develop alternative point and nonpoint source phosphorus load reduction strategies designed to attain the in-lake phosphorus concentration criteria endorsed by New York, Quebec, and Vermont in the 1993 Lake Champlain Water Quality Agreement (Table 2).

METHODS

Lake Sampling

Lake sampling for this study was done for two purposes. The first purpose was to generate data on total phosphorus and chloride concentrations throughout the lake for use in developing a mass balance model. The second purpose was to obtain more complete limnological parameter coverage in vertical profile at several deep water stations.

A total of 52 sampling stations were established throughout the lake at locations listed in Table 3 and shown in Figure 2. The sampling locations were selected to provide several representative stations within each lake arm or bay likely to be specified as a model segment, and to define concentration gradients in areas where they were known to exist. Station locations were also chosen to correspond, wherever possible, to previously established long-term monitoring stations sampled by the Vermont Lay Monitoring Program (Picotte and Lohner, 1993).

Each station was sampled by boat approximately twice per month from late April to early November during 1990 and 1991. The sampling stations were located in the field using a LORAN-C navigation device.

Five of the stations were sampled at discrete depths in complete vertical profile, for a broad list of limnological parameters. The other 47 stations were sampled using a vertical compositing procedure for a limited number of parameters, as described in more detail below.

Vertical Composite Stations

A single, vertically integrated sample was obtained on each sample date at the 47 "composite" stations listed in Table 3 and Figure 2. A compositing procedure was designed so that the sample concentration results would correspond approximately to the vertical "mixed-reactor" assumption to be used in the lake model. The composite samples were intended to represent the concentration that would exist if the water column were completely mixed vertically.

The sample depths for the vertical composite samples were chosen to represent the midpoints of lake strata having approximately equal volumes. Analysis of generalized hypsographic relationships for Lake Champlain (Myer and Gruendling, 1979) indicated that, for the lake as a whole, sample depths of 2, 8, 16, 29, and 69 percent of the total water column depth would represent the midpoints of five lake strata, each including 20 percent of the total lake volume. At lake composite stations where the total depth was greater than 10 meters, samples were obtained at these five levels, and mixed together in equal volumes (one liter each) to form the composite. Where the site depth was 10 meters or less, only one level was sampled at 16% of the total depth, representing the point between two strata of equal volume. The sample depths used in forming the composite samples are listed for each station in Table 3.

All lake composite station samples were obtained using a brass Kemmerer water sampler 56 cm in length (Vermont Department of Environmental Conservation, 1989), and combined into a clean plastic compositing bucket from which the final mixed sample was drawn. The composite samples were analyzed for total phosphorus and chloride, and the Secchi disk transparency was recorded each time a sample was obtained.

Table 3. Lake station locations and sampling depths ("C" = composite, "P" = vertical profile).

Station	Reference Number[1]	Latitude (deg min)	Longitude (deg min)	Total Depth (m)	Type	Sample Depths (m)
01	500447	43 35.53	73 25.80	5	C	1
02	503387	43 42.89	73 22.98	5	C	1
03	503310	43 51.35	73 22.97	8	C	1
04	503288	43 57.10	73 24.47	10	C	2
05	503280	44 00.59	73 24.38	9	C	2
06	500448	44 04.08	73 25.98	10	C	2
07	500449	44 07.56	73 24.77	50	P	0 5 10 15 20 25 30 35 40 45
08	500450	44 11.05	73 23.42	59	C	1 5 9 17 41
09	500451	44 14.53	73 19.75	97	C	2 8 16 28 67
10	500452	44 18.00	73 19.17	122	P	0 12 24 37 49 61 73 85 98 110
11	500453	44 21.42	73 18.02	42	C	1 3 7 12 29
12	500454	44 21.42	73 19.79	93	C	2 7 15 27 64
13	500455	44 24.84	73 17.54	71	C	1 6 11 21 49
14	500456	44 24.84	73 19.72	43	C	1 3 7 12 30
15	500457	44 25.24	73 24.15	37	C	1 3 6 11 26
16	503506	44 25.55	73 13.92	25	C	1 2 4 7 17
17	503410	44 27.98	73 16.64	47	C	1 4 8 14 32
18	503529	44 28.17	73 22.57	47	C	1 4 8 14 32
19	500458	44 28.26	73 17.95	100	P	0 10 20 30 40 50 60 70 80 90
20	500460	44 28.27	73 15.08	35	C	1 3 6 10 24
21	500459	44 28.49	73 13.90	15	C	0 1 2 4 10
22	500461	44 31.69	73 19.13	77	C	2 6 12 22 53
23	500462	44 31.69	73 21.77	50	C	1 4 8 15 35
24	503498	44 33.90	73 12.50	20	C	0 2 3 6 14
25	503519	44 34.92	73 16.87	32	C	1 3 5 9 22
26	500463	44 35.11	73 21.40	42	C	1 3 7 12 29
27	500464	44 35.11	73 23.90	29	C	1 2 5 8 20
28	500465	44 38.53	73 22.93	61	C	1 5 10 18 42
29	503530	44 40.32	73 14.02	43	C	1 3 7 12 30
30	500466	44 40.32	73 16.07	27	C	1 2 4 8 19
31	503538	44 40.80	73 25.00	14	C	0 1 2 4 10
32	500469	44 41.95	73 22.00	59	C	1 5 9 17 41
33	500468	44 42.07	73 25.09	11	C	0 1 2 3 8
34	503485	44 42.49	73 13.61	50	P	0 5 10 15 20 25 30 35 40 45
35	500467	44 42.49	73 16.32	17	C	0 1 3 5 12
36	500470	44 45.37	73 21.30	50	P	0 5 10 15 20 25 30 35 40 45
37	503520	44 45.97	73 11.20	20	C	0 2 3 6 14
38	500474	44 47.11	73 13.27	17	C	0 1 3 5 12
39	500475	44 47.11	73 15.19	40	C	1 3 6 12 28
40	503488	44 47.12	73 09.73	7	C	1
41	503489	44 47.80	73 09.14	6	C	1
42	500471	44 49.21	73 21.15	25	C	1 2 4 7 17
43	503534	44 51.73	73 12.92	15	C	0 1 2 4 10
44	500472	44 53.06	73 21.85	15	C	0 1 2 4 10
45	501605	44 55.07	73 11.18	8	C	1
46	503535	44 56.90	73 20.40	7	C	1
47	503547	44 58.55	73 12.90	5	C	1
48	503540	45 00.02	73 07.55	4	C	1
49	500473	45 00.65	73 20.71	6	C	1
50	503515	45 00.80	73 10.43	4	C	1
51	500476	45 02.22	73 06.99	5	C	1
52	500477	45 02.68	73 08.60	5	C	1

[1] **Vermont Department of Environmental Conservation database.**

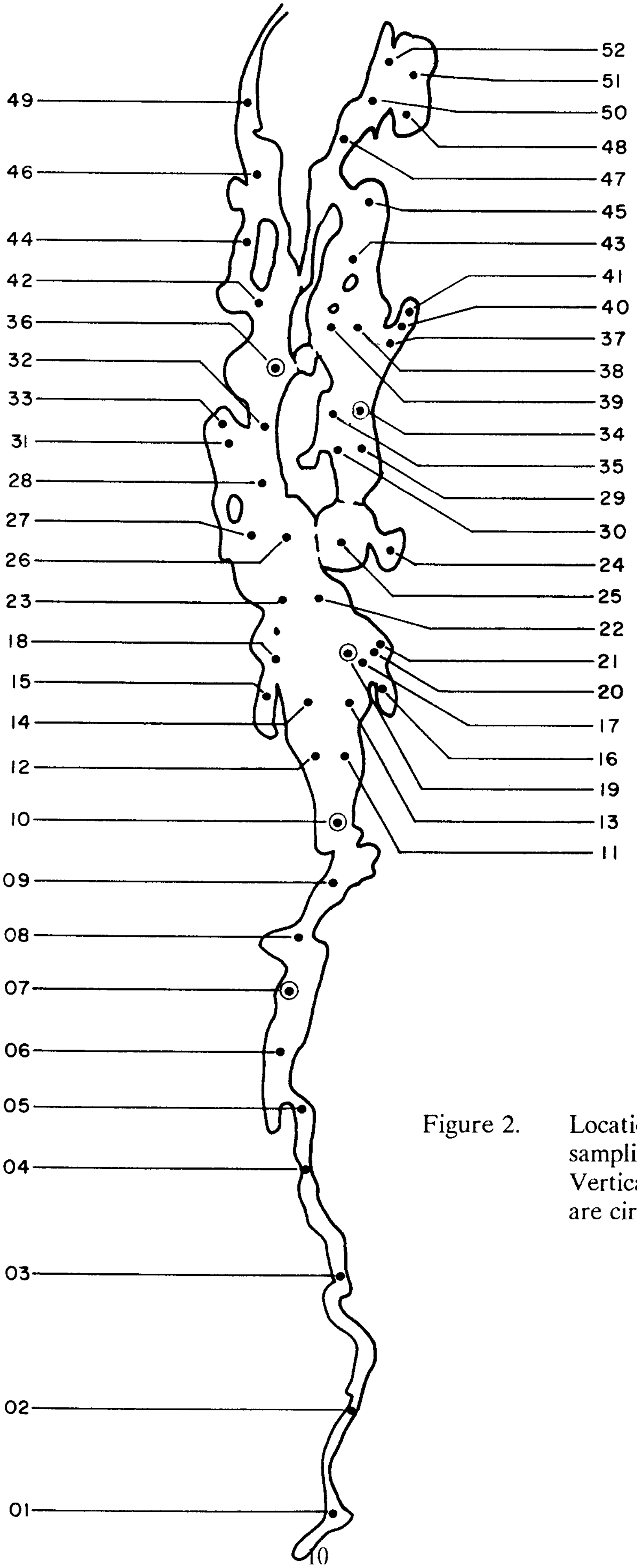

Figure 2. Location of lake sampling stations. Vertical profile stations are circled.

Samples were also obtained for chlorophyll-a analysis at the composite stations on each sampling date. Chlorophyll-a samples were drawn from a separate vertical composite integrated over the euphotic zone only (twice the Secchi disk depth) using a hose sampler. This chlorophyll-a sampling procedure was employed to attain consistency with the methods used in the long-term Vermont Lay Monitoring Program (Picotte and Lohner, 1993).

Vertical Profile Stations

Five lake stations located in deep, central areas of the major arms of the lake were sampled using a Kemmerer sampler at 10 discrete depths throughout the water column, without compositing, to produce complete vertical profiles for a larger set of limnological parameters. Samples were obtained at 0, 10, 20, 30, 40, 50, 60, 70, 80, and 90 percent of the total depth at each station, at the locations and specific depths indicated in Table 3.

The vertical profile samples were analyzed for the following list of parameters, and the Secchi disk transparency was recorded on every sampling visit.

Total Phosphorus
Dissolved Phosphorus
Ortho-Phosphorus (Stas. 7, 10, 34 only)
Chloride
Total Kjeldahl Nitrogen
Nitrate/Nitrite-N
Total Ammonia (1990 only)
Chlorophyll-a
Alkalinity (1990 only)
Total Suspended Solids (1990 only)
Total Silica (1991 only)
Dissolved Silica (1991 only)
Dissolved Oxygen
pH (Stas. 7, 10, 34 only)
Temperature (*in situ*)

Tributary Flow Measurements

Tributary flows to Lake Champlain measured at 31 sites during the period of March 1990 through April 1992 were used for this study. The flow data were obtained for use in developing tributary phosphorus loading estimates, and to provide hydrologic terms needed for the lake model. The gaged streams included all tributaries to Lake Champlain having drainage areas greater than 26 km^2 where flow gaging near the stream's outlet to the lake was determined to be feasible. The streams included in the flow gaging network for this study are listed in Table 4, and their locations are shown in Figure 3. A record of average daily flows for the study period was produced at all gaged sites.

The drainage area values given in Table 4 for the gage stations were provided by the U.S. Geological Survey, the Quebec Ministry of the Environment, and Environment Canada. The drainage areas at the tributary mouths (at the lake) were obtained from the U.S. Geological Survey and from Henson and Gruendling (1977).

At most sites, continuous stream flow gages were operated by the U.S. Geological Survey under cooperative agreements with the states of Vermont and New York. Some sites were pre-existing U.S.G.S. stations with long periods of record. In other cases, gages were newly constructed or re-established for this study. Continuous flow data for the Mill River and Stevens Brook in Vermont for part of 1990 were obtained from the St. Albans Bay Rural Clean Water Program (1991) database maintained at the University of Vermont. Continuous flow data for the Pike River in Quebec were obtained from the Quebec Ministry of the Environment.

Table 4. Tributaries to Lake Champlain with drainage areas greater than 26 km^2 (10 mi^2), and project flow gage network.

Tributary	Gage Location	Reference Number	Drainage Area at Lake (km^2)	Drainage Area at Gage (km^2)	Gage Type*
Vermont					
Winooski	Essex Jct.	04290500	2828	2704	C
Otter	Middlebury	04282500	2462	1627	C
New Haven	Brooksville	04282525		298	P-C
Missisquoi	Swanton	04294000	2223	2202	P-C
Lamoille	E. Georgia	04292500	1909	1777	C
Poultney	Fair Haven	04280000	692	484	C
Lewis	N. Ferrisburg	04282780	209	200	C
Little Otter	Ferrisburg	04282650	185	148	P-C
LaPlatte	Shelburne Falls	04282795	137	116	P-C
East	Orwell	04280800	81	35	P
Mill	Georgia	04292750	59	58	C-P
Stevens	St. Albans	04292800	59	26	C-P
Malletts	Colchester	04290610	76	42	P
Indian	Colchester	04290580	31	28	P
Mud			30		N
Stonebridge	Georgia Plains	04292700	32	22	C
Quebec					
Pike	Bedford	030420	517	404	C
Rock	St. Armand	04294300	152	126	P
New York					
Saranac	Plattsburgh	04273500	1575	1575	C
Ausable	Au Sable Forks	04275500	1323	1160	C
Mettawee	Middle Granville	04280450	1098	433	C
Great Chazy	Perry Mills	04271500	769	640	C
Bouquet	Willsboro	04276500	712	712	C
LaChute	Ticonderoga	04279000	702	606	N
Little Chazy	Chazy	04271815	139	137	C
Salmon	S. Plattsburgh	04273700	175	160	C
Putnam	Crown Point Center	04276842	160	134	C
Little Ausable	Valcour	04273800	189	176	C
Scomotion			104		N
Mt. Hope	Whitehall	04279125	30	30	C
Mill	Port Henry	04276770	73	70	C
Highlands Forge	Willsboro	04276069	30	28	C
Mill	Putnam Sta.	04279040	27	27	C
Hoisington	Westport	04276645	28	17	C
Riley			28		N
Outlet (Richelieu R.)	Fryers Rapids	02OJ007	21150	22000	C

*Gage Type:
C = continuous recording
P = partial record site with daily staff gage readings
P-C = partial record site upgraded to continuous site in Oct. 1990
C-P = University of Vermont continuous site reduced to partial record site after Aug. 1990
N = Gaging not feasible

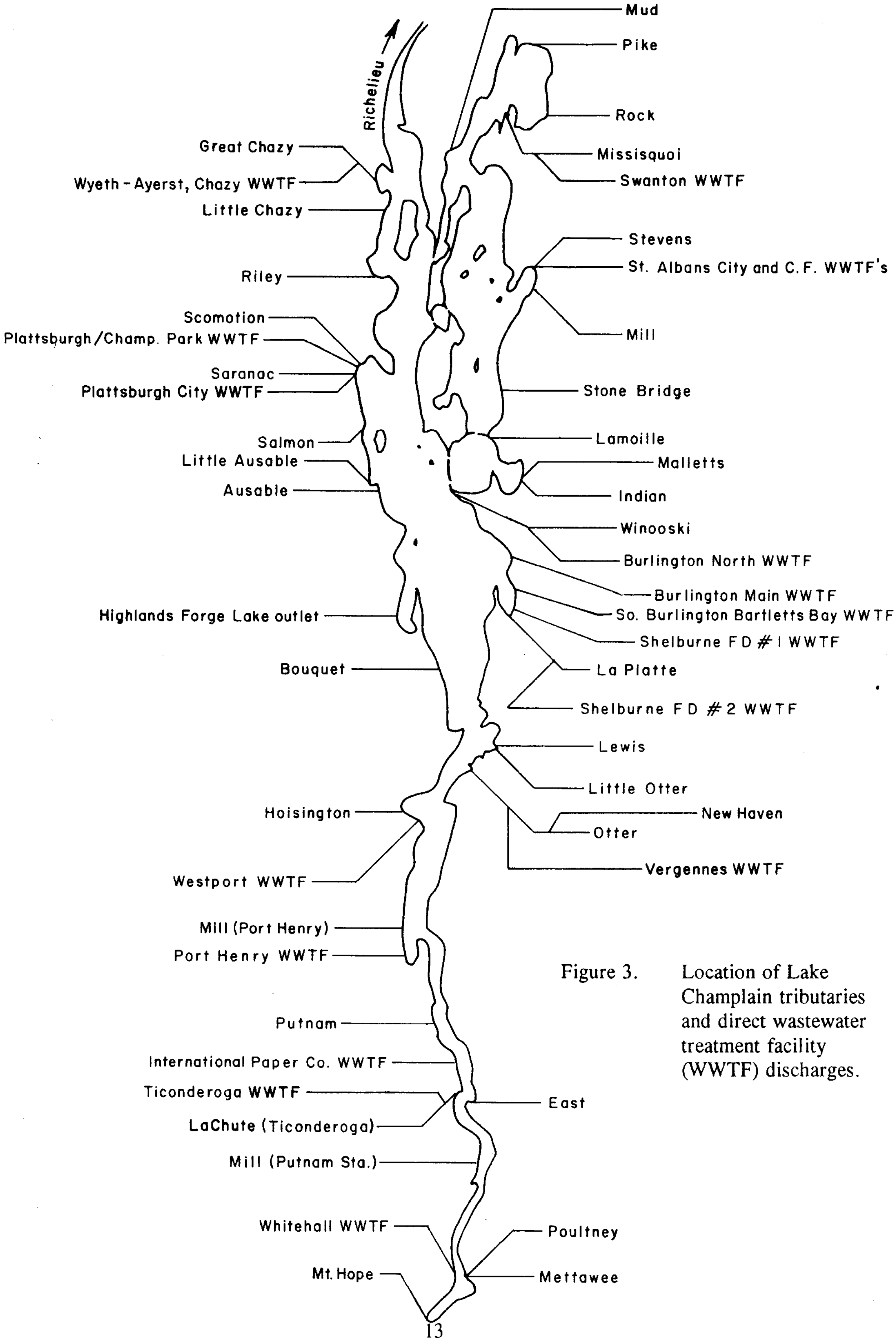

Figure 3. Location of Lake Champlain tributaries and direct wastewater treatment facility (WWTF) discharges.

Continuous flow data for the Richelieu River downstream from the lake's outlet were obtained from the Water Survey of Canada (Environment Canada).

Ten of the stream gages in Vermont and Quebec were operated as "partial record" sites during all or part of the study period (see Table 4). At the partial record sites, staff gages were installed and stage-discharge relationships were established by the U.S. Geological Survey. The staff gages were read daily by trained local residents. The U.S. Geological Survey analyzed the daily instantaneous flow measurements and produced average daily flow estimates for the partial record sites using inter-gage comparison procedures.

A continuous U.S.G.S stream gage on LaChute Creek previously maintained from 1943 to 1978 was not in operation during the study period. However, inter-gage correlation techniques based on the historical gage data for the site were used to estimate annual mean flows for LaChute Creek for 1990-1992.

The stream gage network listed in Table 4 includes a continuous gage established on the New Haven River, which is a tributary to Otter Creek and does not discharge directly to Lake Champlain. The gage on the New Haven was installed to provide greater hydrologic coverage for the Otter Creek watershed. Flow values given in this report for Otter Creek were based on the daily sum of the flows measured at the Otter Creek gage in Middlebury, Vermont and at the New Haven River gage, adjusted for the additional downstream drainage area.

The stream gages listed in Table 4 incorporate a combined drainage area of 16,202 km^2, or about 81% of the lake's 19,881 km^2 total watershed area. The gage sites are located on rivers which, at their mouths, drain a combined area of 18,680 km^2, or about 94% of the lake's watershed area.

Tributary Sampling

Sampling was conducted on 31 tributaries to Lake Champlain, as listed in Table 5. The sampling stations were generally located at the downstream-most bridge crossing on each stream, as near to the lake as possible, but upstream from any possible lake backwater influence. The tributary samples were analyzed for total phosphorus, dissolved phosphorus, and chloride, in order to develop loading estimates for phosphorus budget calculations and lake modeling purposes.

Two sampling stations were established on the Mettawee River, including one on the barge canal in Whitehall, New York. The barge canal site (METT01) was hydraulically influenced to some extent by lake backwater and by the operation of the lock system, but was included because of anticipated water quality differences between the barge canal site and the other Mettawee River station (METT02) located upstream of the confluence with the barge canal.

The number of samples obtained on each tributary ranged from 36 to 115 during the March 1990 to April 1992 study period. The tributary sampling effort concentrated on obtaining as high a proportion of the total number of samples as possible during the relatively rare but critically important high flow conditions. Samples were also obtained under low and moderate flow conditions.

The event-response nature of the tributary sampling program was designed to support the application of flow-stratified loading estimation methods provided by the FLUX program

Table 5. Tributary sampling station locations. The latitude and longitude coordinates were determined at the sampling sites using a Global Positioning System receiver, uncorrected and unlogged, with a precision of approximately 100 m.

Tributary	Station Code	Reference Number[1]	Latitude (deg min)	Longitude (deg min)
Vermont				
Winooski	WINO01	501903	44 31.52	73 15.41
Otter	OTTE01	500509	44 09.94	73 15.40
Missisquoi	MISS01	500505	44 55.23	73 07.63
Lamoille	LAMO01	501794	44 37.96	73 10.39
Poultney	POUL01	POU1	43 34.24	73 23.53
Lewis	LEWI01	500503	44 14.80	73 14.77
Little Otter	LOTT01	501371	44 12.24	73 15.11
LaPlatte	LAPL01	501594	44 22.21	73 13.01
East	EAST01	500504	43 48.13	73 19.13
Mill	MILL01	501563	44 46.80	73 08.68
Stevens	STEV01	501575	44 50.96	73 07.18
Malletts	MALL01	500510	44 33.95	73 09.46
Indian	INDI01	500511	44 33.59	73 10.81
Stonebridge	STON01	500506	44 41.24	73 11.45
Quebec				
Pike	PIKE01	500512	45 07.38	73 04.18
Rock	ROCK01	500513	45 02.00	73 02.54
New York				
Saranac	SARA01	500491	44 41.52	73 27.19
Ausable	AUSA01	500500	44 33.63	73 26.95
Mettawee	METT02	500507	43 31.72	73 23.33
Mettawee/Barge Canal	METT01	500508	43 33.33	73 24.10
Great Chazy	GCHA01	500492	44 58.81	73 25.96
Bouquet	BOUQ01	500498	44 21.84	73 23.41
Little Chazy	LCHA01	500490	44 54.12	73 24.88
Salmon	SALM01	500502	44 38.40	73 29.70
Putnam	PUTN01	500495	43 57.35	73 25.99
Little Ausable	LAUS01	500501	44 35.65	73 29.79
Mt. Hope	HOPE01	500493	43 31.32	73 30.44
Mill (Port Henry)	MILB01	500496	44 03.15	73 28.77
Highlands Forge	HIGH01	500499	44 25.47	73 25.71
Mill (Putnam Sta.)	MILC01	500494	43 44.03	73 23.32
Hoisington	HOIS01	500497	44 11.05	73 26.04

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[1] Vermont Department of Environmental Conservation database.

(Walker, 1987, 1990) used for this study. Sample concentration results distributed over the entire range of flow conditions, but with greatest emphasis on high flow conditions, are required for optimally precise loading estimates. These methods reduce the variance of tributary mass loading estimates by accounting for the sometimes strong relationship between concentration and flow.

Tributary samples were obtained from bridges or by wading using DH-48 or DH-59 "Suspended Sediment Samplers" described in Edwards and Glysson (1988). These sampling devices provided a depth and velocity integrated sample of the entire stream water column.

In smaller, well-mixed streams where lateral concentration gradients were unlikely to exist (based on visual judgement), only one vertically integrated sample was collected on each sampling date at the centroid of flow (point of greatest depth-velocity product). At sites where the stream width was greater, up to five vertical samples were obtained at equal width increments across the stream and proportionately composited into a single sample in a simplified version of the "equal width increment" technique described in Edwards and Glysson (1988).

Wastewater Treatment Facility Sampling

A total of 17 wastewater treatment facilities discharging either directly into Lake Champlain or immediately upstream (i.e. below the tributary sampling stations) were sampled monthly during the study period. An exception was the Wyeth-Ayerst industrial discharge in Chazy, New York which was not discovered to be downstream of the sampling station on the Great Chazy River until late in the study. Only two samples were obtained from this facility.

The sampled facilities are listed in Table 6 and shown in Figure 3. The wastewater samples were either 8-hour or 24-hour composites of the final effluent after chlorination and all other treatment steps, collected with the assistance of the plant operators. All wastewater samples were analyzed for total phosphorus and chloride. Effluent flow rates from these facilities were obtained from plant operation records filed under the conditions of their state discharge permits.

Precipitation Sampling

Four precipitation sampling stations, listed in Table 6, were employed for this study to provide a spatial representation of the phosphorus and chloride content of precipitation falling on Lake Champlain. These stations were operated during the non-winter months of 1990 and 1991, and samples were collected during each rain event in which adequate sample volume was generated. All precipitation samples were analyzed for total phosphorus and chloride. The amount of rainfall measured at an adjacent rain gage was also recorded each time a sample was obtained.

One existing National Atmospheric Deposition Program site in Underhill, Vermont located 27 km east of the lake shore, was used to provide "wet" precipitation samples in which the collection device was open only when precipitation was occurring. Three additional precipitation sampling stations were established specifically for this study, with one in Orwell, Vermont, within 100 meters of the lake shore, another in South Hero, Vermont about 800 m from the lake shore on the largest of the Lake Champlain Islands, and the third in Bloomingdale, New York about 53 km west of the lake shore. These three sites used "bulk" collection techniques in which samples were drawn by trained volunteer samplers from a continuously open, screened, funnelled collection vessel immediately after each precipitation event.

Table 6. Wastewater treatment facility and precipitation sampling stations.

WASTEWATER TREATMENT FACILITY	STATION CODE
Vermont	
Swanton	SWA
St. Albans City	SAM
Northwest Correctional Facility	NWC
Burliington North	BUN
Burlington Main	BUM
South Burlington Bartletts Bay	SBB
Shelburne Fire District #1	SF1
Shelburne Fire District #2	SF2
Vergennes	VER
New York	
Plattsburgh/Champlain Park	PCP
Plattsburgh City	PLA
Westport	WES
Port Henry	PHE
International Paper Co.	IPC
Ticonderoga	TIC
Whitehall	WHI
Wyeth-Ayerst, Chazy	AYC

PRECIPITATION STATION	
Vermont	
Underhill	UND
Orwell	ORW
South Hero	SHE
New York	
Bloomingdale	BLO

A combination of wet and bulk collection methods was used for this study on the assumption that the "true" deposition values fall between the wet and the bulk values. The wet samples exclude the dry component of atmospheric deposition, whereas the bulk samples may include a dry component that is related to local land sources that would not exist over the open lake water. The results of the two types of precipitation samples were used to bracket the likely ranges of total phosphorus and chloride atmospheric loading rates to the surface of Lake Champlain.

Contamination from insects, bird droppings, and other foreign material was sometimes encountered in the precipitation samples. Samples for which field records indicated visually obvious contamination were excluded from the data set. Total phosphorus samples exceeding 0.100 mg/l collected during precipitation events of greater than 1.0 cm in amount were also judged to be contaminated and were excluded from the data set, following the procedure of Murphy and Doskey (1975).

Laboratory Analytical Methods

All water chemistry samples collected for this study were analyzed by the Vermont Department of Environmental Conservation Laboratory using procedures documented in Vermont Department of Environmental Conservation Laboratory (1992). The analytical methods are listed in Table 7.

Quality Assurance Procedures

Field Quality Assurance Program

The quality assurance procedures employed for this study included a field duplicate analysis program and a field blank analysis program. Field duplicates were generated by obtaining two separate samples at the same location and time. The field duplicate results were used to assess combined field and laboratory analytical variability. Field blanks were generated by running laboratory distilled and deionized water through the field sampling equipment. Field blank samples were analyzed to detect possible field or laboratory contamination.

Generally, one field duplicate sample and one field blank sample were processed for each chemical parameter within each batch of samples collected by a field crew on one day of sampling. About 12 percent of the total laboratory analytical effort for this study was devoted to these two quality assurance aspects.

Laboratory Quality Assurance Program

The Vermont Department of Environmental Conservation Laboratory operates a quality assurance and quality control program that is documented in Vermont Department of Environmental Conservation Laboratory (1992). In addition to these standard laboratory procedures, this study also generated samples for use in chemical recovery assessment. Internal spikes of known concentration were added in the laboratory to samples split in the field, and the samples were analyzed for percent recovery. Approximately 0.5 percent of the total laboratory analytical effort for this study was devoted to the chemical recovery assessment program.

Table 7. Laboratory analytical methods (Vermont Department of Environmental Conservation Laboratory, 1992).

Parameter	Lab Code	Method Description	Method Reference[1]	Filtration
Total Phosphorus	TP	Colorimetric, ascorbic acid, persulfate digestion	SM 4500-PF, EPA 365.1	none
Dissolved Phosphorus	DP	Colorimetric, ascorbic acid, persulfate digestion	SM 4500-PF, EPA 365.1	0.45 Fm
Ortho-phosphorus	OP	Colorimetric, ascorbic acid	SM 4500-PF, EPA 365.1	0.45 Fm
Chloride	TCL	Colorimetric, automated ferric thiocyanate	SM 4500-Cl^- E, EPA 325.1	none
Total Kjeldahl Nitrogen	TKN	Potentiometric, ion selective electrode	EPA 351.4	none
Nitrate/Nitrite	TNOX	Colorimetric, automated cadmium reduction	EPA 353.2, SM 4500-NO_3^- F	none
Total Ammonia	TNH3	Colorimetric, automated phenolate	EPA 350.1, SM 4500-NH_3 B	none
Chlorophyll-a	CHA	Fluorometric with pheophytin correction	SM 10200H.3	1.6 Fm GFA
Alkalinity	ALK	Potentiometric titration	SM 2320B	none
Total Suspended Solids	TSS	Total nonfilterable residue	SM 2540-D	1.5 Fm 934AH
Total Silica	TSI	Colorimetric, heteropoly blue, NaOH digestion	SM 4500-Si F	none
Dissolved Silica	DSI	Colorimetric, automated heteropoly blue	SM 4500-Si F	0.45 Fm
Dissolved Oxygen	DO	Winkler titration	SM 4500 OC	none

[1] EPA = U.S. Environmental Protection Agency (1983)

SM = Standard Methods (1989)

Database Documentation and Access

Water Quality Database

All water quality data generated during this study, including both field and laboratory measurements, are stored in a personal computer database maintained by the Vermont Department of Environmental Conservation. These data are available on request in either electronic or hard copy format to other government agencies, university researchers, and the general public. The contents of the data files that are available for distribution are documented in Table 8.

Tributary Flow Database

All tributary gage height and flow records generated during this study are maintained and processed by the U.S. Geological Survey, with a few exceptions as noted previously. Tabulated average daily flow rates and other information for each site are published in annual data reports (e.g. Toppin *et al.* 1992, Firda *et al.* 1992). The data are stored in the U.S. Geological Survey's National Water Data Storage and Retrieval System (WATSTORE).

Table 8. Summary of project water quality database. The data fields listed below are stored at the Vermont Department of Environmental Conservation, Water Quality Division in a Paradox® (Borland International) personal computer database.

FIELD NAME	**DEFINITION**
Sample Identification	
Station Id	**Station reference number, Tables 3, 5, and 6**
Date	**Date sample was taken**
Time	**Time sample was taken**
Type	**Sample type (COMP = depth composite, TIME = time composite, KEMM = Kemmerer, HOSE = hose)**
Depth, m	**Depth of sample**
Program	**Program code, ChampDFS for this project**
Lake Data	
TP, mg/l	**Total phosphorus**
TCL, mg/l	**Chloride**
Chl-a, ug/l	**Chlorophyll-a**
Secc, m	**Secchi disk transparency**
DP, mg/l	**Dissolved phosphorus**
DOP, mg/l	**Dissolved ortho-phosphorus**
TKN, mg/l	**Total Kjeldahl nitrogen**
TNOX, mg/l	**Total nitrate/nitrite-N**
TNH3, mg/l	**Total ammonia-N**
Field pH	**Field pH**
Reg Alk, mg/l	**Alkalinity, measured by EPA standard method**
TSi, mg/l	**Total silica as SiO_2**
DSi, mg/l	**Dissolved silica as SiO_2**
TSS, mg/l	**Total suspended solids**
Temp, C	**Temperature, *in-situ* thermistor method**
DO, mg/l	**Dissolved oxygen, Winkler titration method**
DO meter, mg/l	**Dissolved oxygen, *in-situ* membrane electrode meter method**
River Data	
TP, mg/l	**Total phosphorus**
DP, mg/l	**Dissolved phosphorus**
TCL, mg/l	**Chloride**
Wastewater Data	
TP, mg/l	**Total phosphorus**
TCL, mg/l	**Chloride**
Precipitation Data	
Amount, in	**Amount of precipitation measured for the event that was sampled**
TP, mg/l	**Total phosphorus**
TCL, mg/l	**Chloride**

SAMPLING RESULTS

Lake Sampling

Phosphorus, Chloride, Chlorophyll-a, and Secchi Disk Summary

The lake sampling results for the four water quality parameters measured at all 52 lake stations (total phosphorus, chloride, chlorophyll-a, and Secchi depth) are statistically summarized in Table 9. The chloride and total phosphorus results are illustrated in Figures 4 and 5. The summary statistics presented in Table 9 represent the variations among all sample dates during 1990 and 1991. In developing statistics for the five vertical profile stations, the results were first averaged across all discrete depths in the water column to calculate an average value for each sample date comparable to the composite station values. The chlorophyll-a data at the profile stations were not presented in this summary, however, because a full water column average concentration would not be comparable with the composite station chlorophyll results, which were obtained in the euphotic zone only using different field procedures.

Figure 4 shows that highest chloride concentrations existed in the South Lake at station 4, located just north of the International Paper Co. discharge. This discharge contained very high chloride concentrations (see wastewater treatment facility results below), and apparently affected chloride levels at several lake stations in the area. The lowest chloride levels in the lake were found in Missisquoi Bay which is strongly influenced by the Missisquoi River inflow and its relatively low chloride concentrations (see tributary results below). Chloride differences between lake stations located along the main axis of the lake were very slight, but showed a consistent north-south concentration gradient.

The spatial patterns of total phosphorus Figure 5 were consistent with long-term eutrophication monitoring data for Lake Champlain (Picotte and Lohner, 1993; Smeltzer, 1992). Highest phosphorus concentrations were observed in St. Albans Bay, Missisquoi Bay, and the South Lake. Lowest levels of eutrophication existed in Malletts Bay and in the Main Lake region.

Differences Between Years

Annual mean total phosphorus concentrations at the lake composite stations were generally similar between 1990 and 1991. The maximum absolute mean annual concentration difference between years for any lake station was 0.005 mg/l, and the maximum relative difference was 22%. The number of lake stations that had higher mean total phosphorus levels in 1991 was approximately equal to the number of stations that showed phosphorus declines from 1990 to 1991. There was no statistically significant lakewide difference in annual mean total phosphorus concentrations between 1990 and 1991, based on a paired t-test among all composite stations. This finding indicates that pooling of in-lake data from both 1990 and 1991 would be justified in the development of a steady-state modeling analysis for the lake.

Seasonal Variations

Seasonal variations in chloride and total phosphorus are shown in Figures 6 and 7 for 13 segments of Lake Champlain, using the segment boundaries shown in Figure 1. All data collected during both 1990 and 1991 were averaged by month across all stations within each segment, and the monthly segment mean concentrations are plotted in Figures 6 and 7.

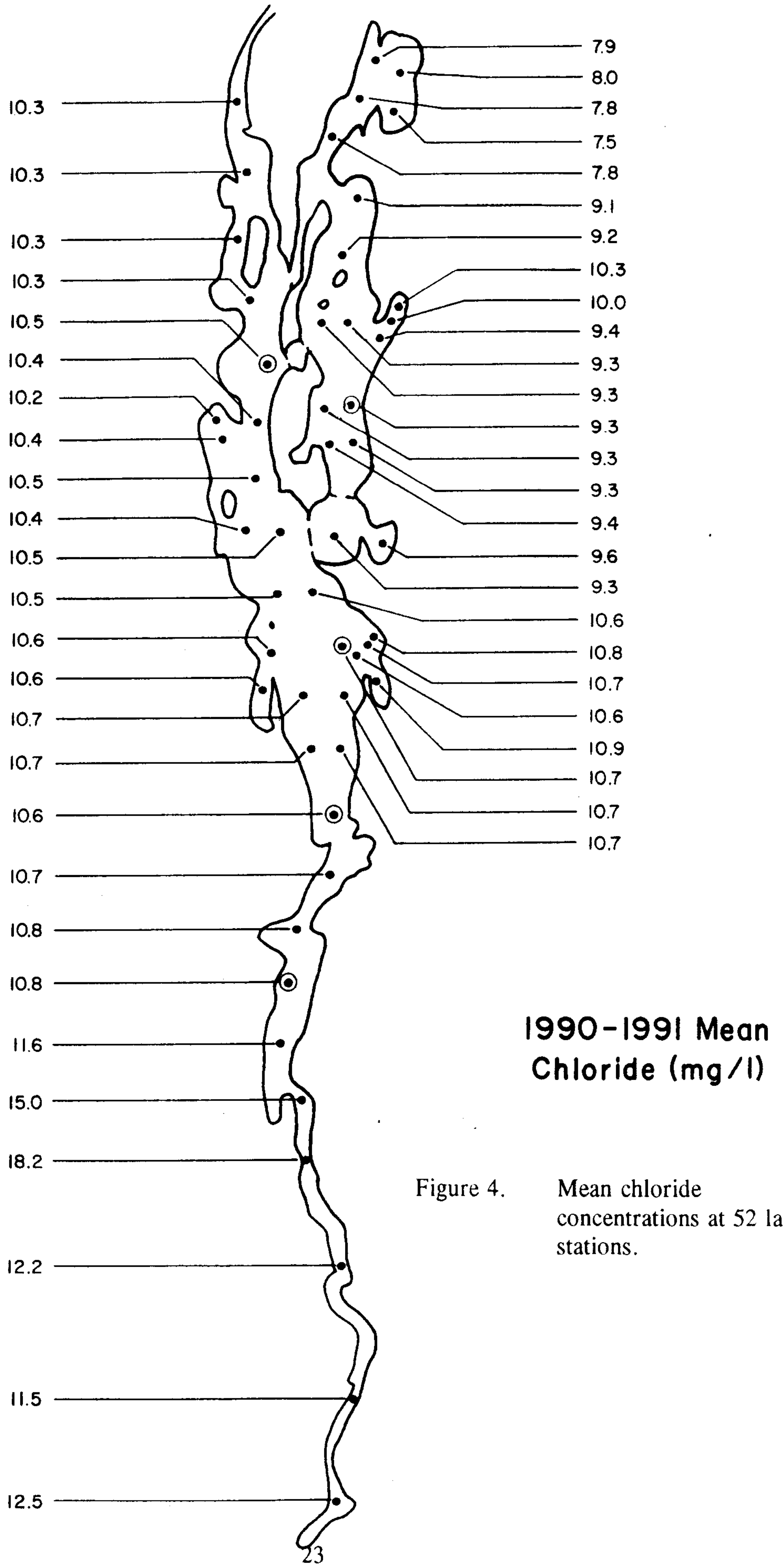


Figure 4. Mean chloride concentrations at 52 lake stations.

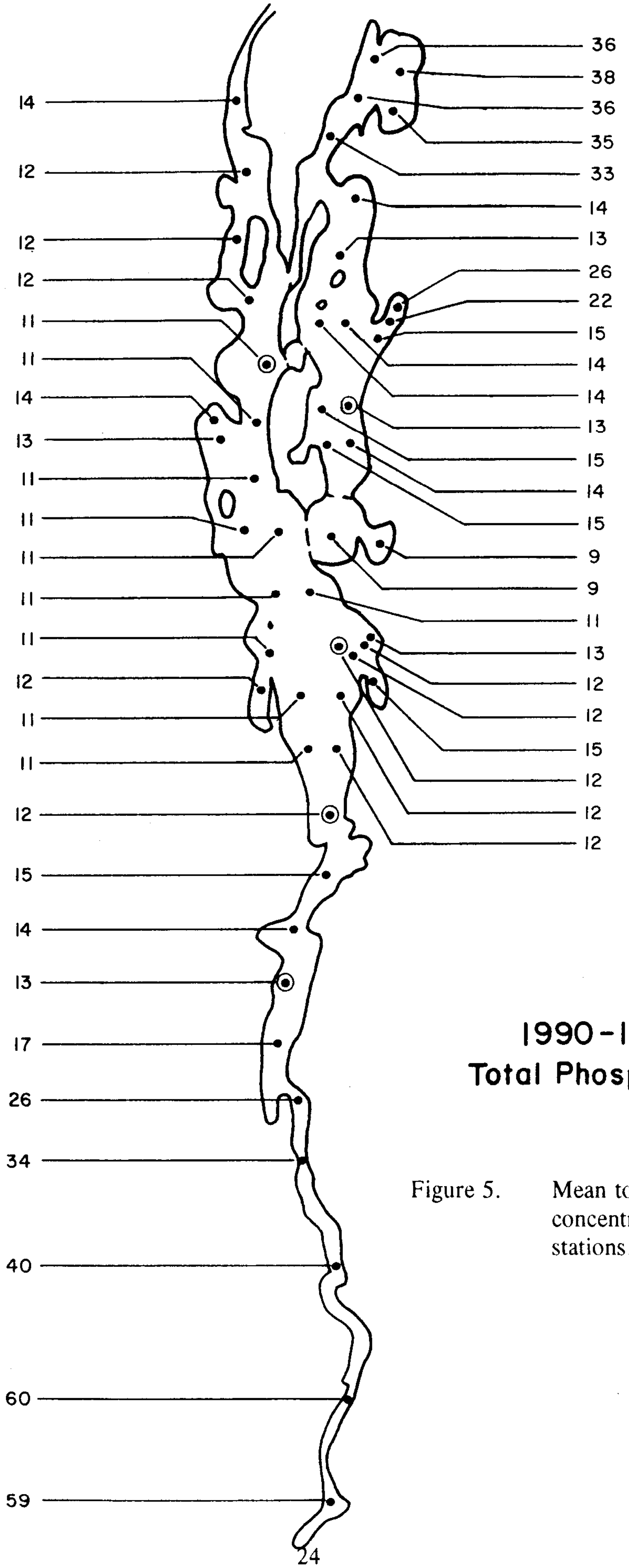


Figure 5. Mean total phosphorus concentrations at 52 lake stations.

Table 9. Lake data statistical summary (1990-1991) for total phosphorus (TP, Fg/l), chloride (CL, mg/l), chlorophyll-a (CHA, Fg/l), and Secchi Depth (SD, m). Statistics are: Arithmetic mean (M), standard deviation (S), and number of sample dates (N).

Station	TP-M	TP-S	TP-N	CL-M	CL-S	CL-N	CHA-M	CHA-S	CHA-N	SD-M	SD-S	SD-N
01	58.79	18.42	14	12.49	2.16	14	6.45	4.36	14	0.33	0.21	14
02	59.68	21.60	25	11.46	2.28	25	13.40	14.83	22	0.51	0.28	25
03	39.72	12.92	25	12.21	3.39	25	8.56	5.85	22	0.83	0.41	25
04	34.23	9.54	26	18.22	4.78	26	6.27	4.13	22	0.89	0.33	26
05	25.68	5.03	25	15.00	3.23	26	5.93	4.50	23	1.35	0.53	26
06	16.54	4.97	26	11.56	0.65	26	4.60	3.05	23	3.31	1.51	26
07	13.40	2.84	26	10.83	0.25	26				3.95	1.08	26
08	13.81	4.11	26	10.77	0.28	26	3.48	2.63	23	4.22	1.25	26
09	14.96	4.53	23	10.65	0.26	23	3.60	2.52	20	3.77	1.47	23
10	11.99	1.85	26	10.63	0.22	26				4.43	1.56	26
11	12.20	3.37	20	10.70	0.19	20	3.78	2.73	19	4.65	1.23	20
12	11.00	1.97	19	10.70	0.21	19	3.14	1.71	18	4.50	1.10	19
13	11.84	3.22	19	10.65	0.21	20	3.58	2.91	19	4.80	1.12	20
14	11.22	2.21	18	10.65	0.19	18	3.36	1.98	17	4.85	1.25	18
15	12.05	2.82	20	10.59	0.16	20	4.08	2.53	19	4.85	0.97	20
16	14.88	4.91	26	10.85	0.41	26	3.76	3.38	23	4.62	1.44	26
17	11.80	3.18	25	10.62	0.28	25	2.85	2.22	22	5.24	1.34	25
18	11.15	2.39	20	10.58	0.20	19	3.22	1.66	19	4.94	1.13	20
19	11.70	1.51	20	10.65	0.21	20				4.91	1.61	20
20	11.88	3.50	26	10.68	0.39	26	3.57	3.61	22	5.03	1.51	26
21	12.96	4.08	26	10.76	0.31	26	3.54	2.96	23	4.96	1.38	26
22	10.85	2.18	20	10.56	0.20	19	2.98	2.34	19	5.09	1.36	20
23	10.83	2.28	18	10.51	0.17	18	3.29	2.41	17	4.94	1.17	18
24	9.19	2.61	26	9.59	0.46	26	2.72	1.83	22	5.03	1.60	26
25	8.88	3.22	26	9.26	0.55	26	2.65	1.52	22	5.03	1.66	26
26	10.55	2.19	20	10.50	0.22	20	2.56	1.99	19	5.36	1.43	19
27	11.00	2.38	19	10.44	0.23	19	3.20	1.96	17	5.03	0.92	19
28	11.14	2.46	21	10.47	0.28	21	2.63	2.00	20	5.15	1.34	20
29	14.21	2.48	24	9.34	0.20	24	3.82	2.60	22	5.18	0.99	25
30	14.88	3.96	25	9.37	0.28	25	3.75	2.08	22	5.24	1.17	25
31	12.90	2.96	21	10.38	0.38	21	3.27	2.54	20	4.80	1.46	21
32	10.95	1.99	21	10.44	0.27	21	3.30	2.73	19	4.99	1.25	20
33	13.57	4.14	21	10.18	0.54	21	3.49	2.79	19	4.01	1.14	21
34	13.04	1.91	27	9.32	0.19	27				5.32	1.33	27
35	14.56	3.42	25	9.32	0.21	25	2.90	1.59	22	5.27	1.25	25
36	10.72	1.43	21	10.47	0.22	21				5.04	1.15	21
37	15.35	3.11	26	9.43	0.25	26	5.14	3.03	20	4.76	0.71	26
38	14.27	3.92	26	9.26	0.26	26	4.72	3.27	22	5.21	0.96	26
39	13.85	2.17	26	9.26	0.29	26	4.30	2.69	21	5.43	1.05	26
40	22.46	7.50	26	10.02	0.50	26	7.99	7.11	22	2.95	0.77	26
41	25.58	7.96	26	10.28	0.55	26	9.52	7.79	22	2.51	0.82	26
42	11.68	2.36	25	10.29	0.31	25	3.00	1.82	22	5.19	1.30	25
43	13.32	2.44	25	9.15	0.28	25	4.04	3.15	22	5.17	1.00	25
44	11.88	3.22	25	10.26	0.32	25	3.06	2.35	22	4.91	1.18	25
45	14.24	3.43	25	9.05	0.34	25	4.12	3.12	21	4.94	1.35	25
46	12.46	3.48	24	10.27	0.34	24	3.78	2.28	20	4.49	0.99	24
47	33.08	10.16	26	7.75	0.75	26	8.21	6.96	23	1.87	0.73	26
48	35.54	8.31	26	7.51	0.97	26	8.29	6.46	23	1.70	0.51	26
49	14.16	3.17	25	10.30	0.37	25	3.14	2.05	22	4.36	0.92	24
50	35.88	13.09	26	7.75	0.77	26	11.75	19.20	23	1.98	1.02	26
51	38.38	14.95	26	8.04	0.84	26	9.78	4.21	23	1.61	0.50	26
52	35.58	11.69	26	7.92	0.79	26	9.95	4.31	21	1.90	0.55	26

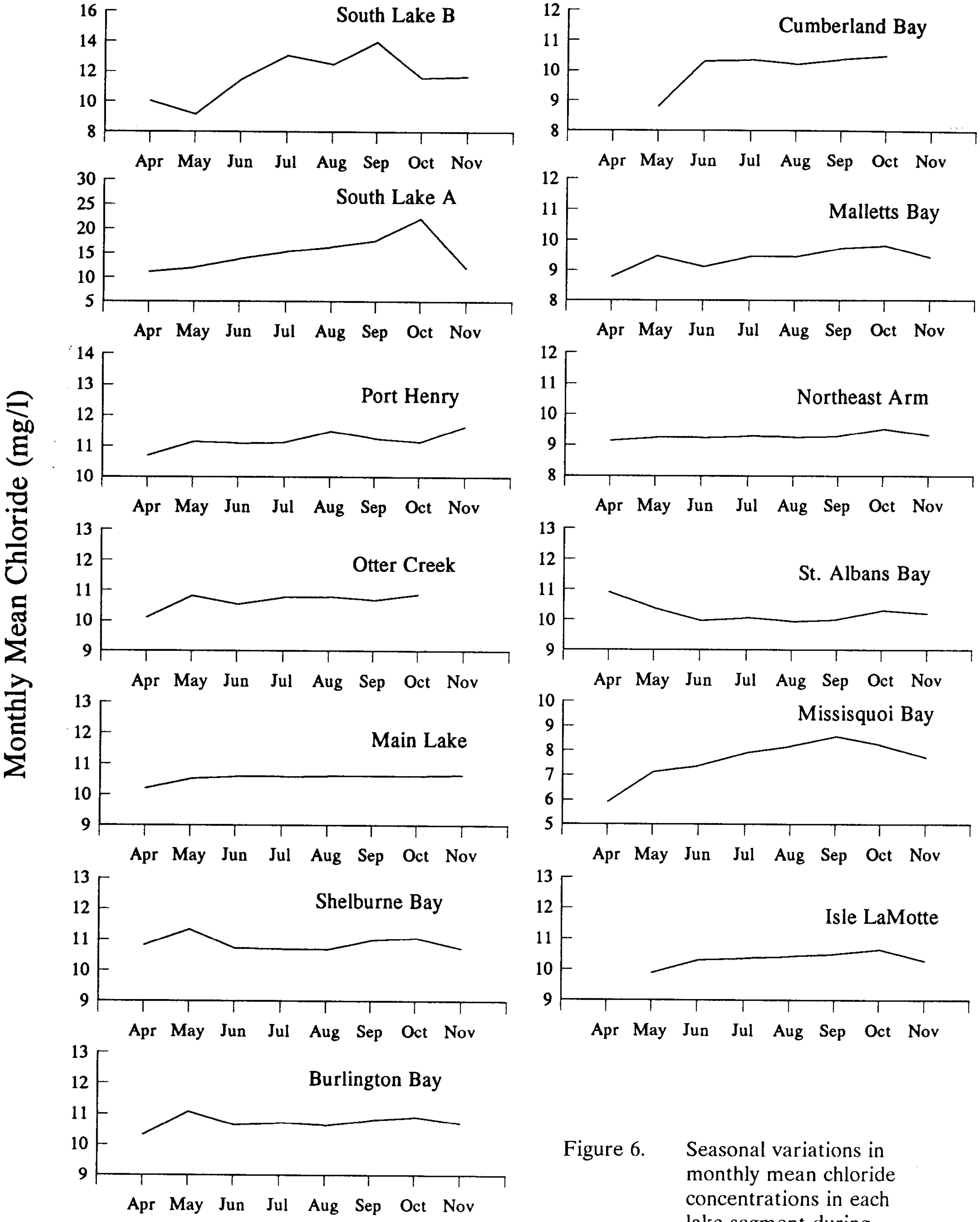


Figure 6. Seasonal variations in monthly mean chloride concentrations in each lake segment during 1990-1991.

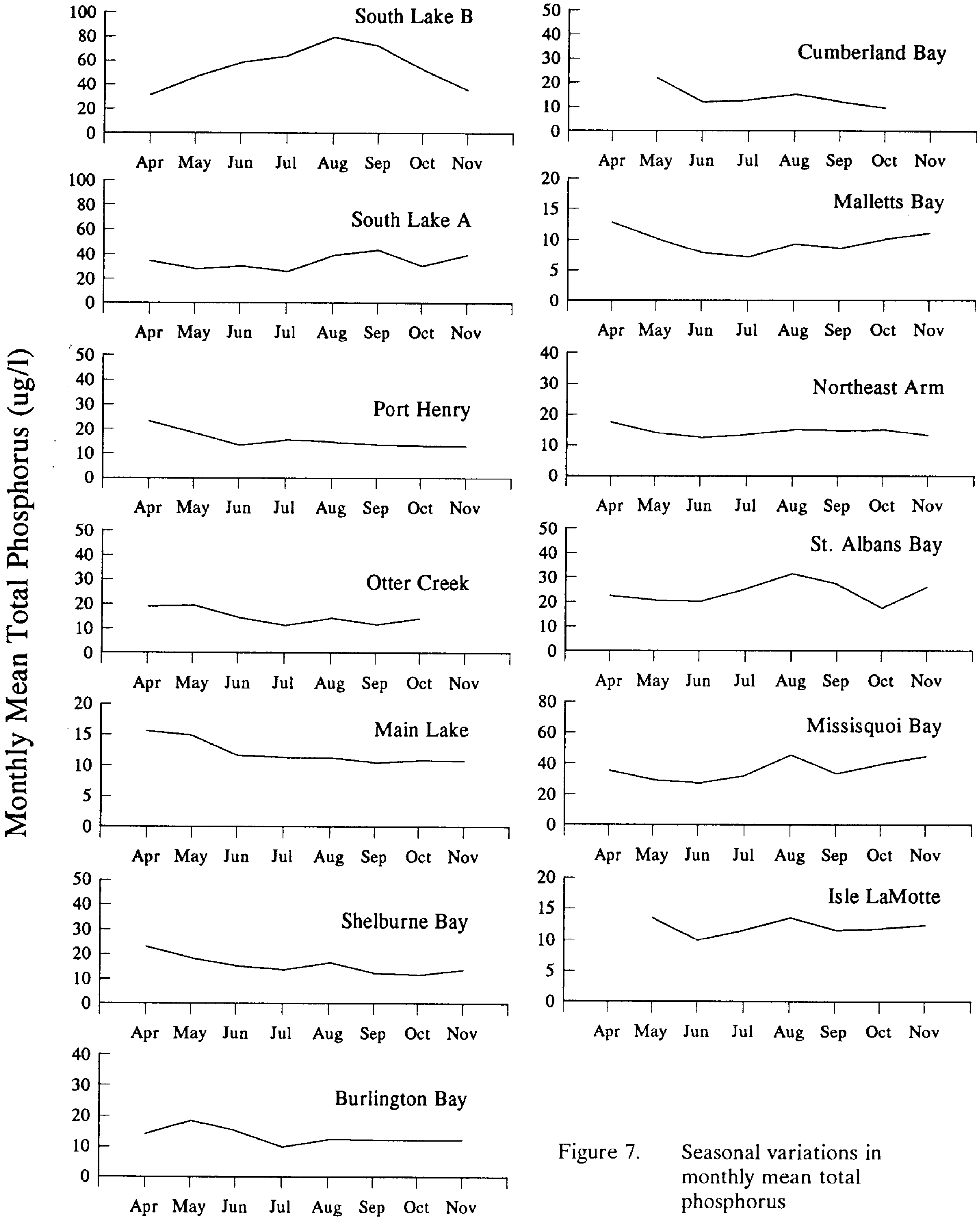


Figure 7. Seasonal variations in monthly mean total phosphorus concentrations in each lake segment during 1990-1991.

Chloride concentrations in most lake segments shown in Figure 6 were fairly stable during the open water season. The major exceptions were Missisquoi Bay and the two South Lake segments, where chloride levels peaked during the summer. The summer increase in the South Lake was apparently the result of less tributary inflow from the south available to dilute the high-chloride International Paper Co. discharge. The increase in Missisquoi Bay was probably related to low summer flows and less dilution from the low-chloride Missisquoi River.

Figure 7 shows that total phosphorus concentrations in most lake segments were moderately elevated during the spring runoff period and then declined to more stable levels through the summer and fall. Different trends were observed in some of the more shallow and eutrophic areas such as the South Lake, St. Albans Bay, and Missisquoi Bay where peak phosphorus concentrations occurred during the summer. In general, however, the spatial phosphorus variations between lake segments were more pronounced than the seasonal variations within individual segments.

Comparison with Long-Term Conditions

Total phosphorus data from the Vermont Lay Monitoring Program (Picotte and Lohner, 1993) were examined to indicate whether the 1990-1991 study period was unusual in comparison with the long-term phosphorus record for the lake. Figure 8 shows summer (June-August) mean total phosphorus levels recorded since 1979 at nine Lay Monitoring Program stations having ten or more years of data and five or more sample dates per year.

While no formal statistical trend analyses were applied, a visual examination of Figure 8 indicates that no sustained and general upward or downward trends in summer phosphorus levels have occurred in Lake Champlain since 1979. Furthermore, data obtained during the 1990-1991 study period fell within the normal range for most Lay Monitoring stations. Two exceptions were Lay Monitoring stations #7 and #8, where the 1991 mean values were influenced by a few questionably high sample results obtained on some dates. The other main exception was St. Albans Bay, where the lower than normal phosphorus values seen in 1990 and 1991 may indicate that water quality in the bay is beginning to respond to the upgrade of the St. Albans Treatment Plant, where phosphorus removal facilities were installed in 1986. In general, it appears that the 1990-1991 study period was a suitably representative time period on which to base a steady-state phosphorus modeling analysis for Lake Champlain.

Vertical Profile Data

Depth-time isopleth diagrams were made for each water quality parameter sampled at the five vertical profile stations. As described in the Methods section, these plots were based on a sampling frequency of approximately twice per month from late April to mid-November at most stations, at ten equal depth intervals in the water column.

The temperature data in Figure 9 show a summer thermal stratification pattern with typical epilimnion depths of about 10 meters, and a thermocline zone often extending down to a depth of 30 meters. Complete homoiothermal conditions were observed only in very early spring and late fall.

More detailed temperature studies (Myer, 1977; Aquatec, 1987) have revealed the presence of an internal seiche in Lake Champlain with a period of 4-5 days and an amplitude of up to 40 meters at times. Some of the variations within the summer season in the vertical location of the temperature isotherms shown in Figure 9 are probably related to the oscillations of this internal seiche, particularly at the most northern and southern sections of the Main Lake (e.g. stations 7 and 36)

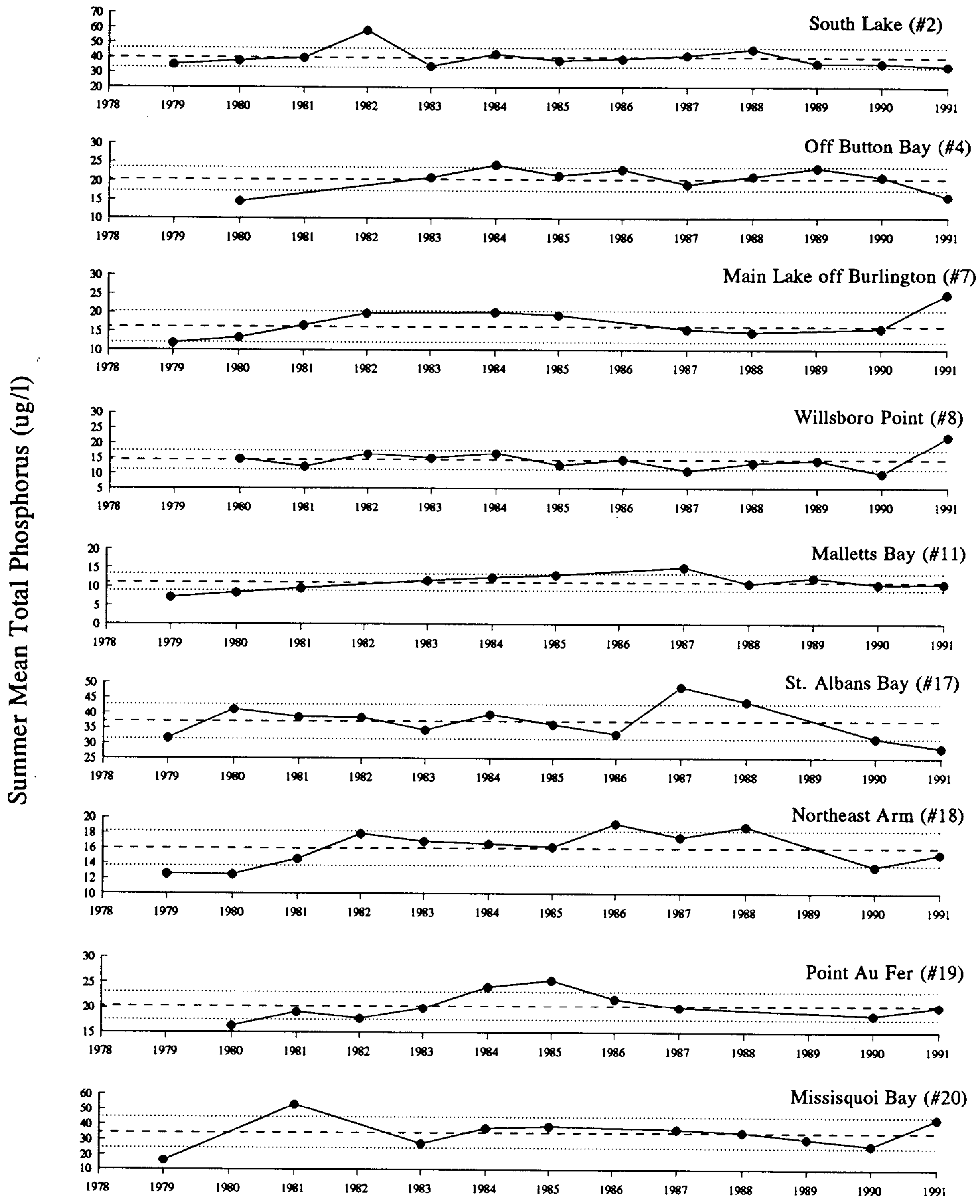


Figure 8. Long-term trends in mean summer total phosphorus concentrations at nine Vermont Lay Monitoring Program stations in Lake Champlain. The 1979-1991 long-term mean (dashed line) plus or minus the standard deviation (dotted line) are indicated for each station.

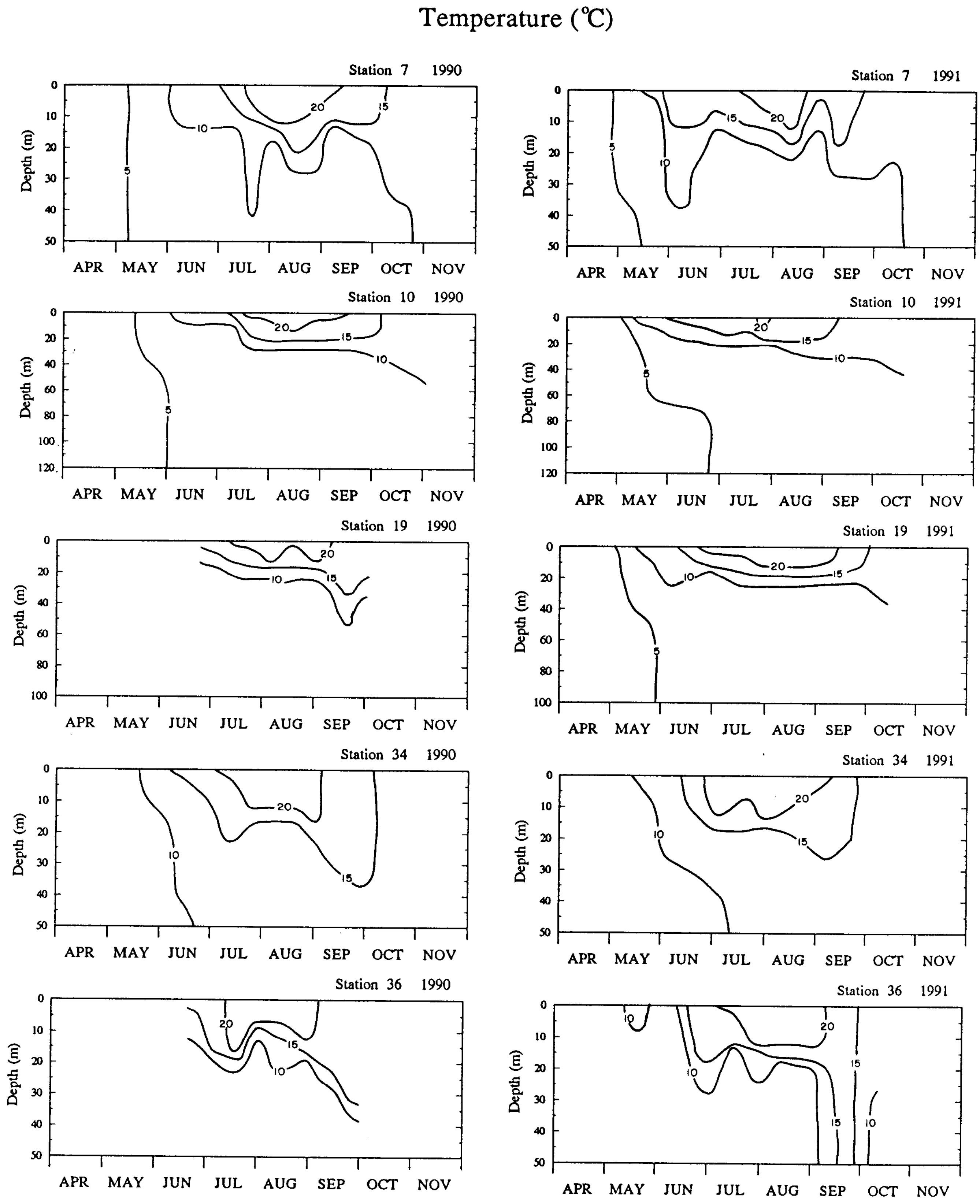


Figure 9. Depth-time plots for temperature at five vertical profile stations.

where the seiche amplitude is greatest. However, the temperature data for this study were not obtained at sufficient frequency to describe these seiche related thermal patterns in more detail.

Dissolved oxygen profiles are shown in Figures 10 and 11 based two different analytical methods. Dissolved oxygen concentration at stations 7, 10, and 34 were analyzed by Winkler titration. Orthograde dissolved oxygen profiles were observed at stations 7 and 10 during the summer stratification season, with higher dissolved oxygen concentrations occurring in the deeper water. In contrast, there was a substantial hypolimnetic dissolved oxygen deficit observed at station 34 located in the Northeast Arm, with minimum dissolved oxygen levels below 4.0 mg/l. A dissolved oxygen deficit of similar magnitude in the Northeast Arm of Lake Champlain was previously reported by Potash *et al.* (1969).

Dissolved oxygen was measured *in situ* by the membrane electrode method at stations 19 and 36. The dissolved oxygen profiles at these stations (Figure 11) varied from orthograde to clinograde at various times during the summer. Hypolimnetic dissolved oxygen concentrations were about 2 mg/l lower at stations 19 and 36 than at the other Main Lake stations (7 and 10) where titrimetric analyses were conducted (Figure 10).

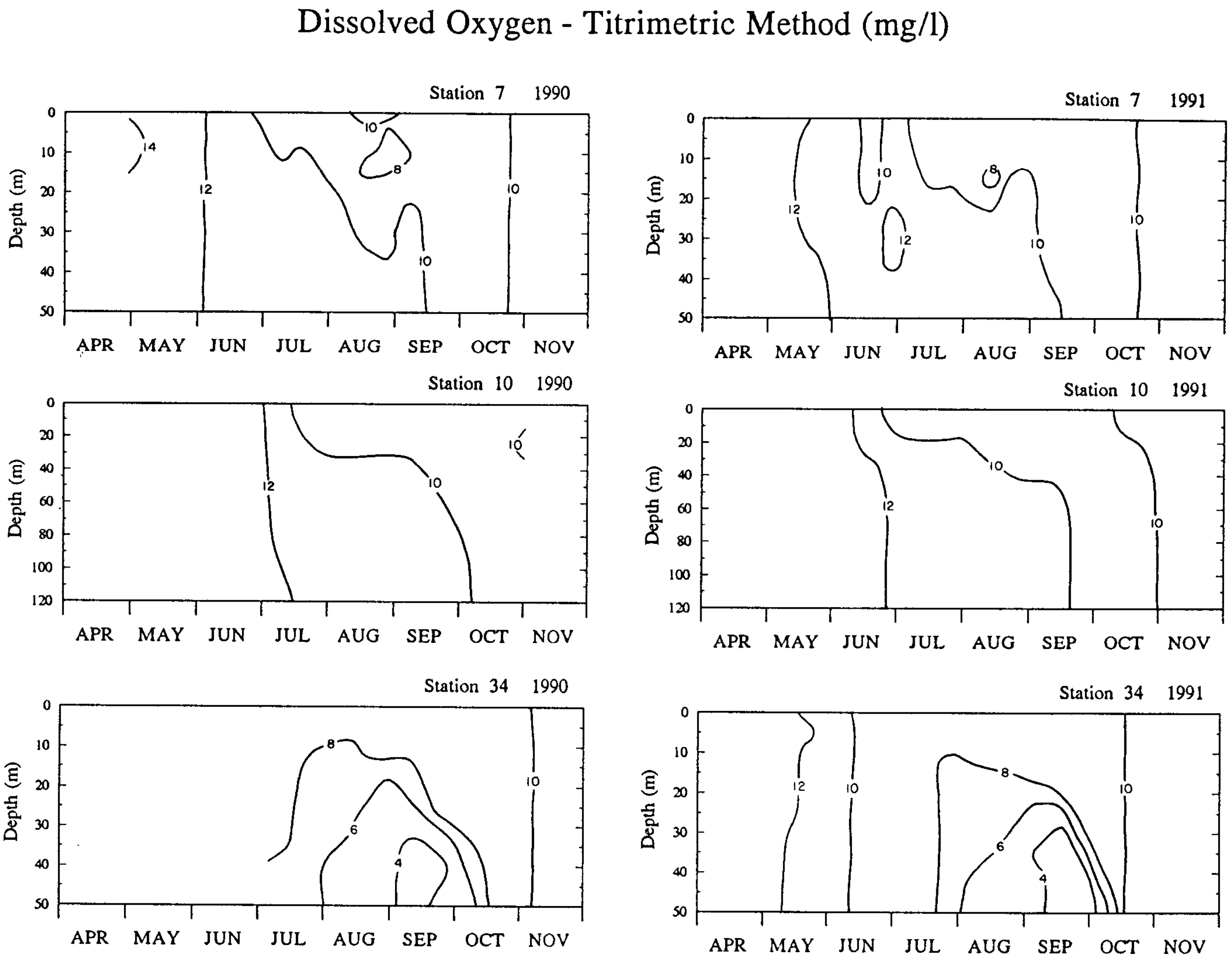
Comparison of dissolved oxygen profiles obtained simultaneously at the same lake station by the two methods (Figure 12) indicated that a discrepancy of about 2 mg/l existed between the two methods for samples obtained deep in the water column, while surface samples compared well. The differences seen in Figure 12 were also consistently noted at other stations when profiles were obtained by the two procedures on nearly the same dates. The reason for this discrepancy is not known. Standard calibration and analytical methods were followed for both procedures. Until the reason for these methodological differences can be resolved, care should be exercised in comparing these results with Lake Champlain data presented by other investigators.

The chloride data shown in Figure 13 show minimal vertical concentration differences at the vertical profile stations. It appears that chloride behaves conservatively in this respect, as would be expected.

Total phosphorus, dissolved phosphorus, and ortho-phosphorus results for the vertical profile stations are shown in Figures 14, 15, and 16. Vertical phosphorus gradients were very minor at these stations. While the dissolved fractions showed some hypolimnetic accumulation, total phosphorus levels usually remained within the range of 10-20 Fg/l throughout the water column, with occasional epilimnetic values below 10 Fg/l. Vertical mixing resulting from internal seiche activity may account for the lack of strong water column phosphorus gradients in Lake Champlain. Spatial differences in phosphorus levels between the various segments of Lake Champlain are much more pronounced than the vertical concentration gradients observed at these deep water stations.

Chlorophyll-a concentrations varied both seasonally and with depth at the vertical profile stations, as shown in Figure 17. The highest levels (rarely above 10 Fg/l) were generally observed in the spring and fall. During the summer stratification period, epilimnetic chlorophyll-a concentrations were typically in the 2-5 Fg/l range, declining to values less than 1 Fg/l in the hypolimnion. No metalimnetic chlorophyll maxima were observed at these stations during the study period.

Total Kjeldahl nitrogen (TKN) results are shown in Figure 18. TKN levels varied little throughout the water column at these stations. Total nitrogen to total phosphorus (TN/TP) mass ratios (where TN = TKN + NO_3/NO_2-N) averaged 34 to 53 among the five vertical profile stations, indicating that these areas of Lake Champlain are strongly phosphorus-limited for phytoplankton



Dissolved Oxygen - Titrimetric Method (mg/l)

Figure 10. Depth-time plots for dissolved oxygen (titrimetric method) at three vertical profile stations.

Dissolved Oxygen - Membrane Electrode Method (mg/l)

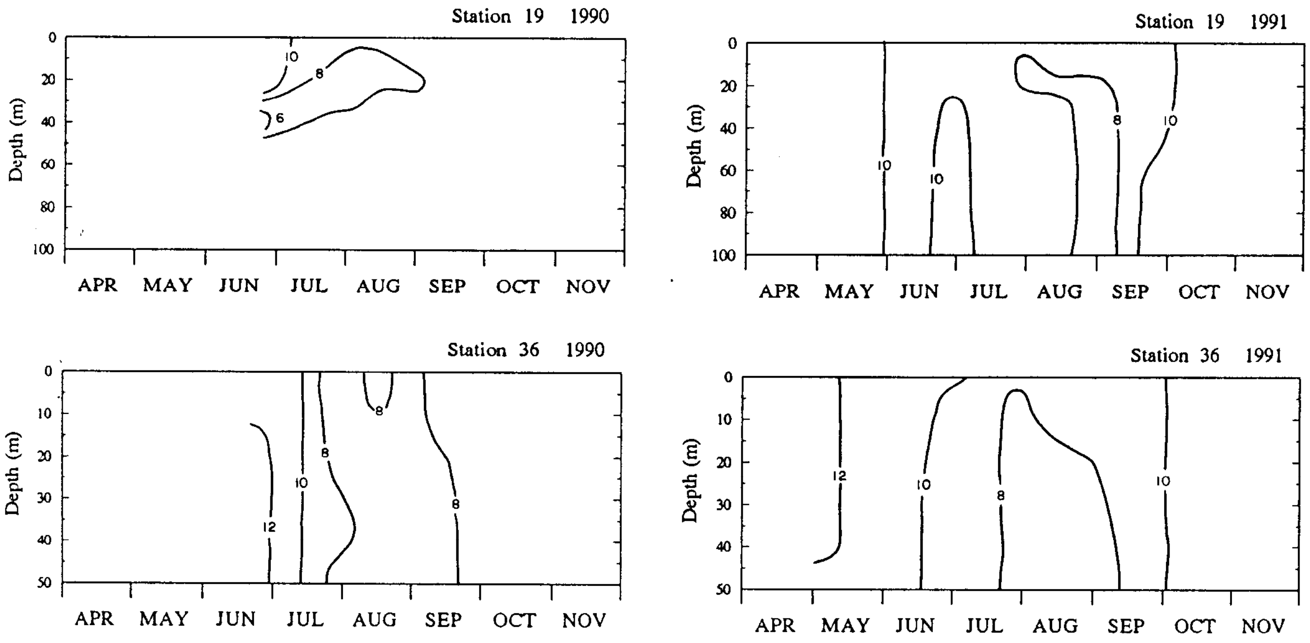


Figure 11. Depth-time plots for dissolved oxygen (membrane electrode method) at two vertical profile stations.

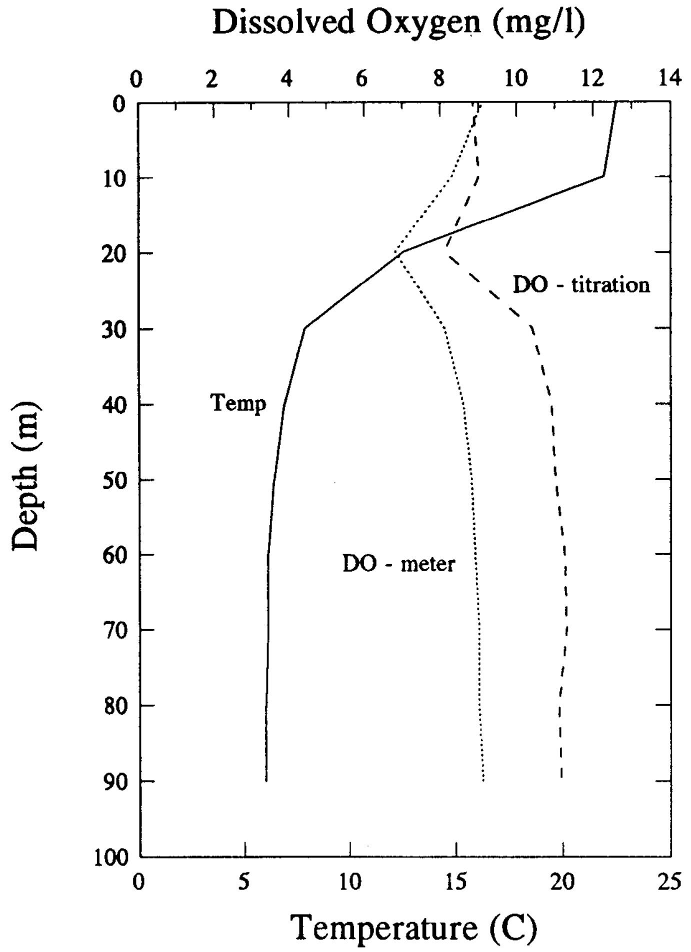


Figure 12. Comparison of dissolved oxygen profiles measured simultaneously at lake station 19 on August 13, 1991 by the titrimetric and the membrane electrode (meter) methods.

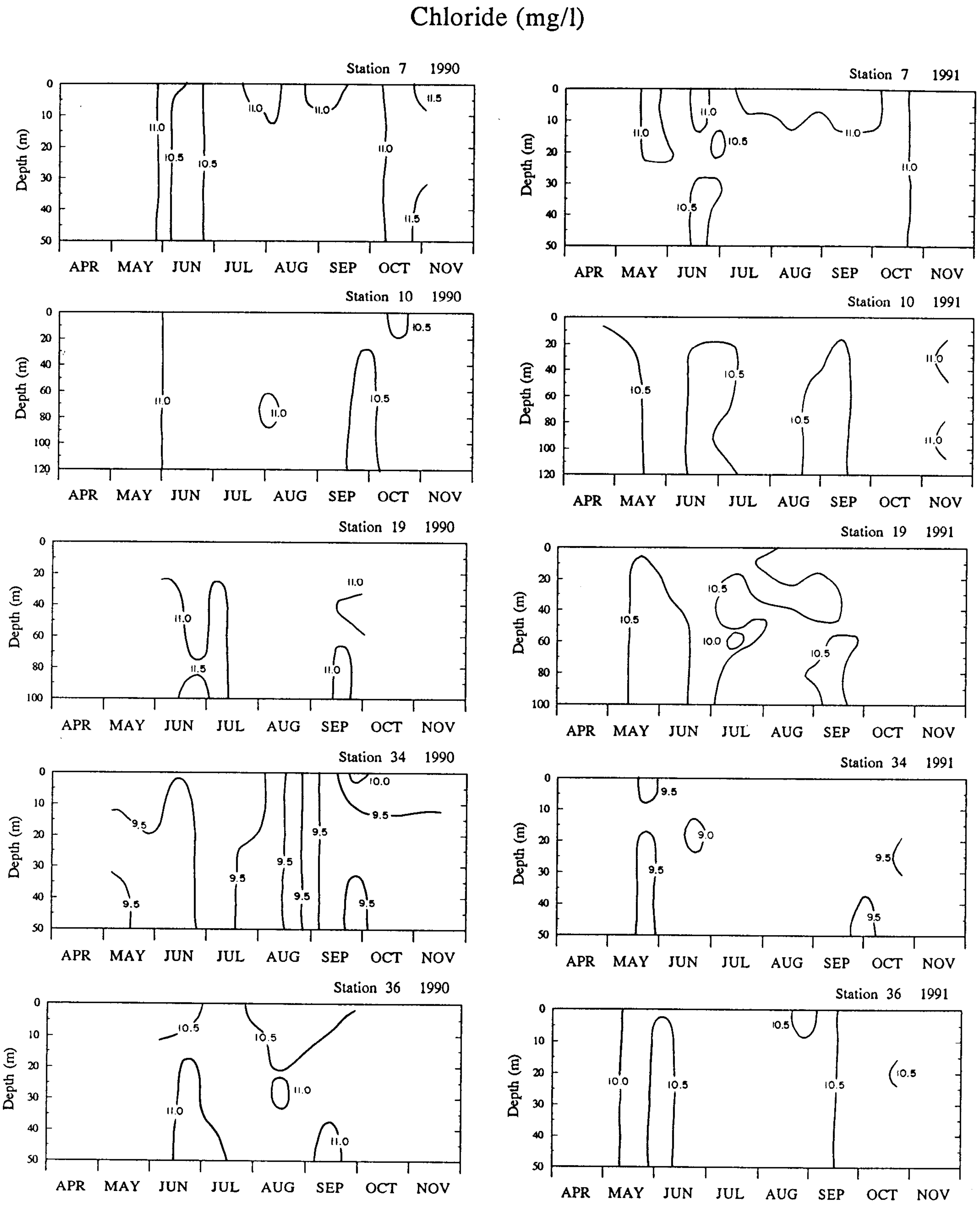


Figure 13. Depth-time plots for chloride at five vertical profile stations.

Total Phosphorus (ug/l)

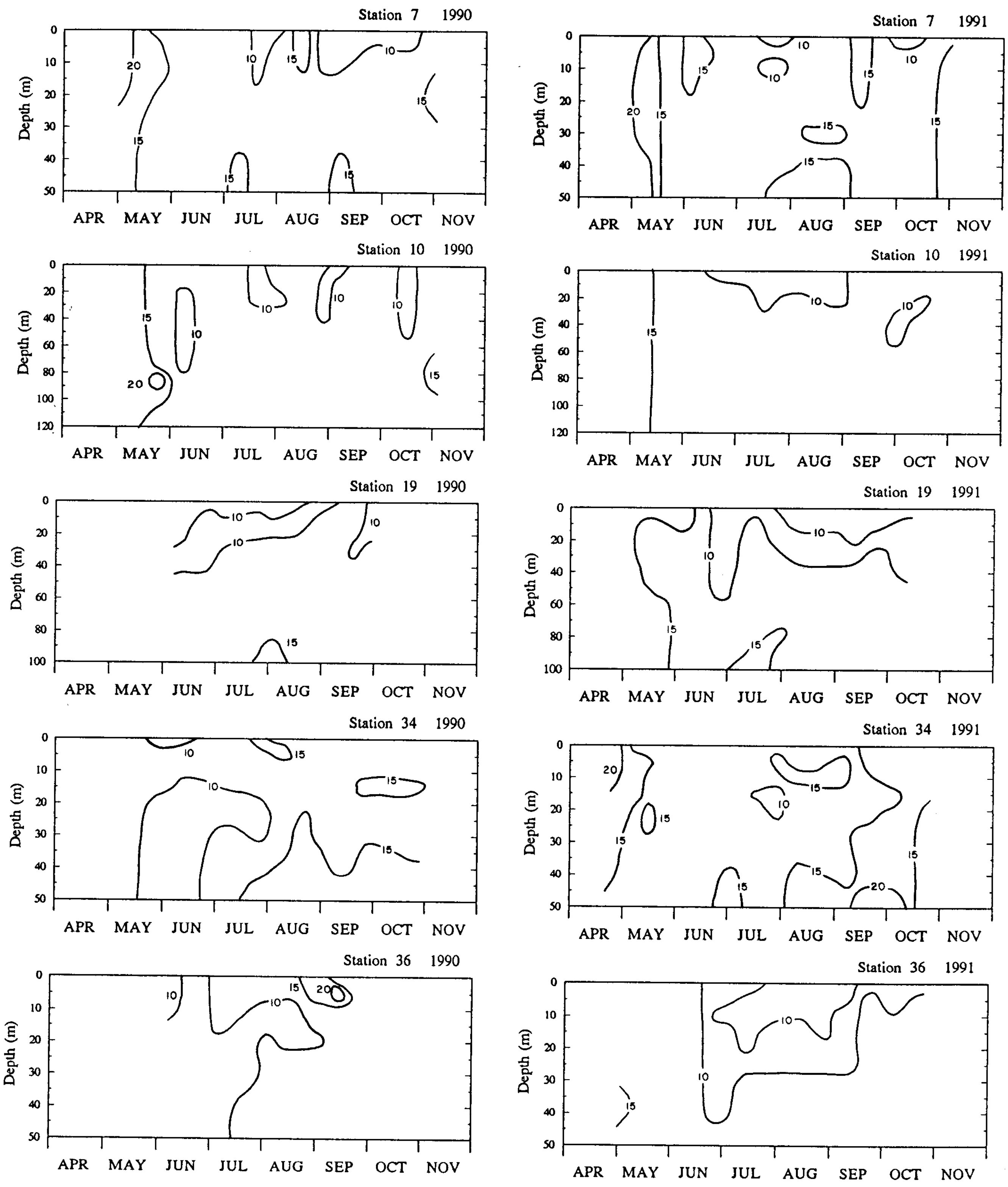


Figure 14. Depth-time plots for total phosphorus at five vertical profile stations.

Dissolved Phosphorus (ug/l)

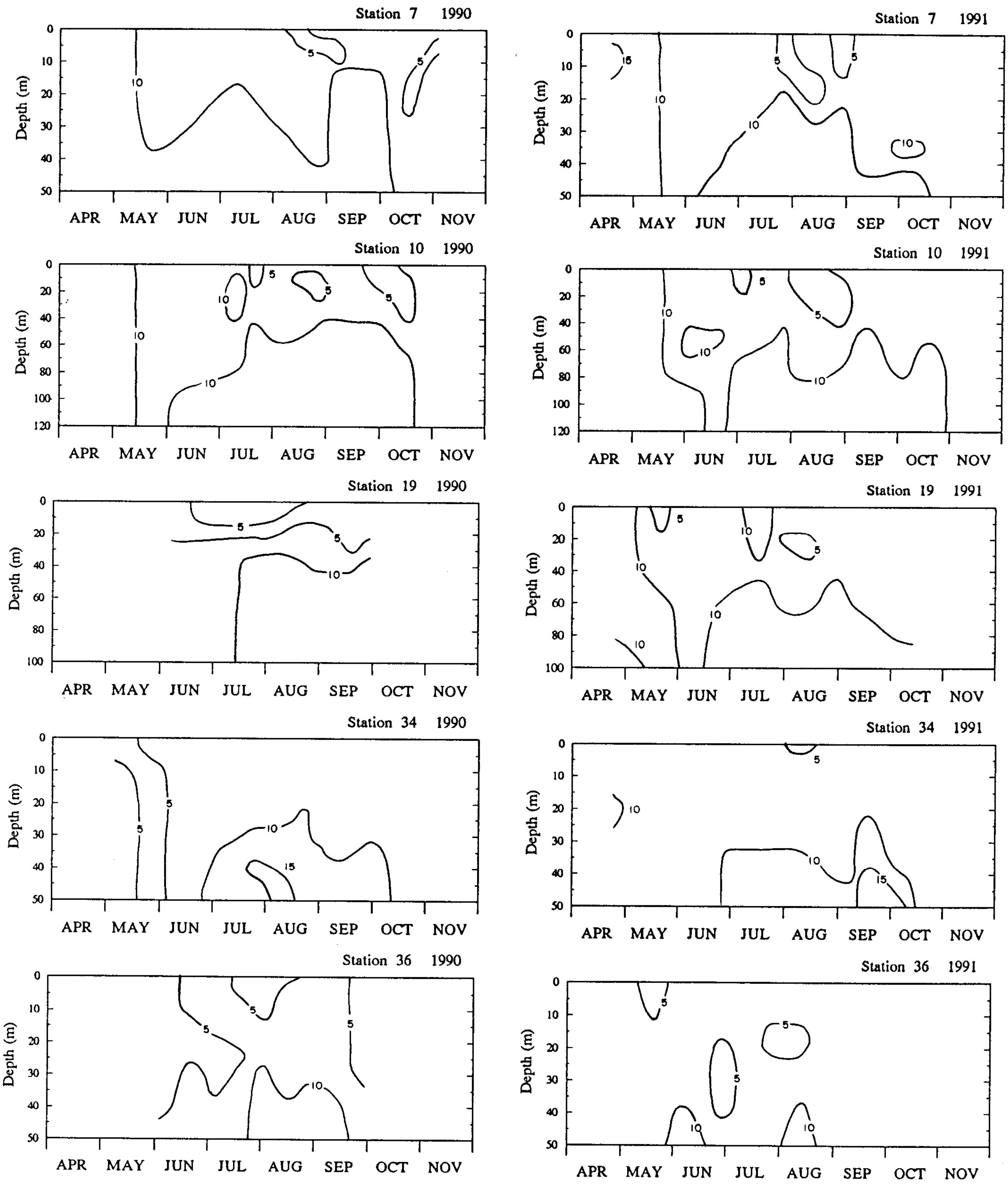


Figure 15. Depth-time plots for dissolved phosphorus at five vertical profile stations.

Ortho-Phosphorus (ug/l)

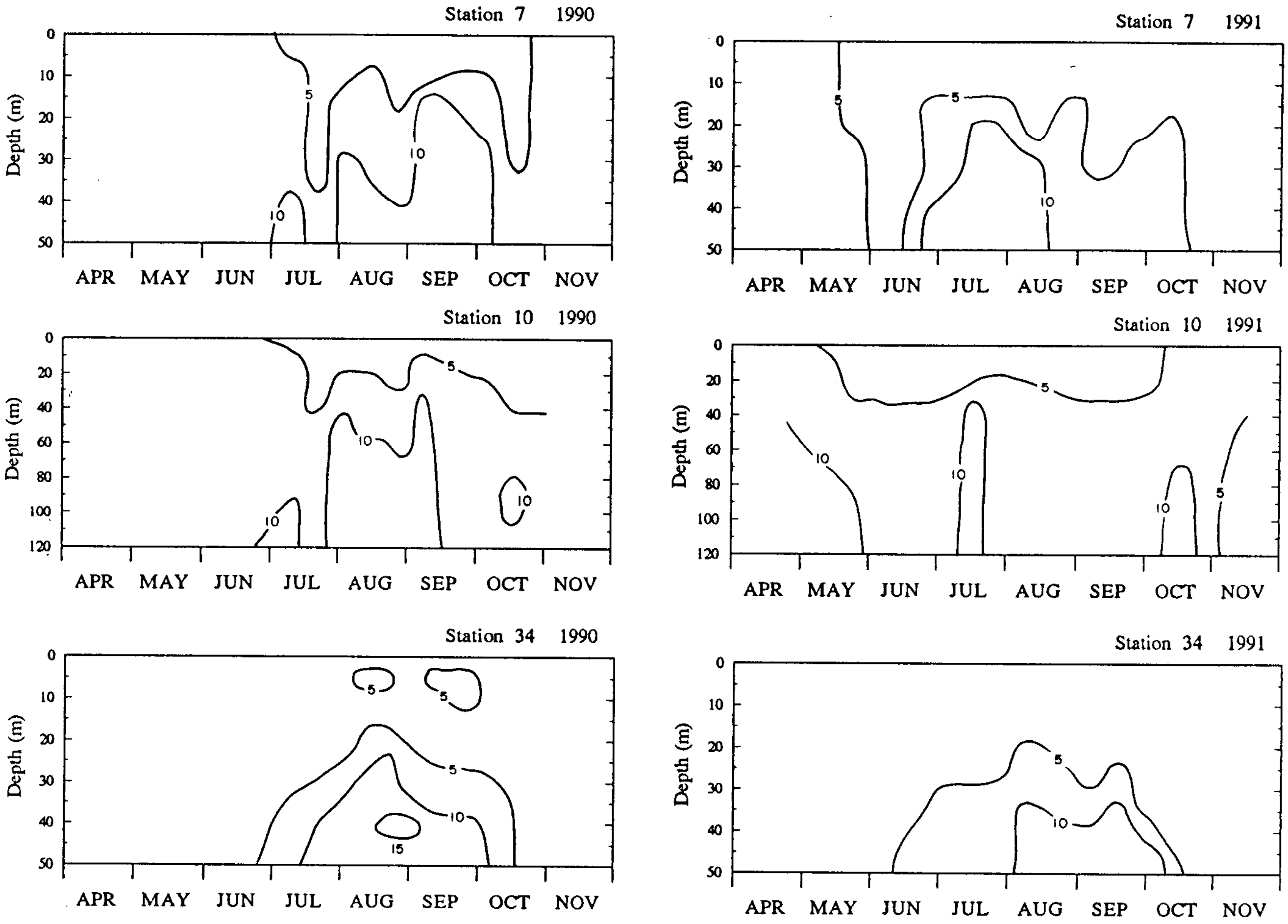


Figure 16. Depth-time plots for ortho-phosphorus at three vertical profile stations.

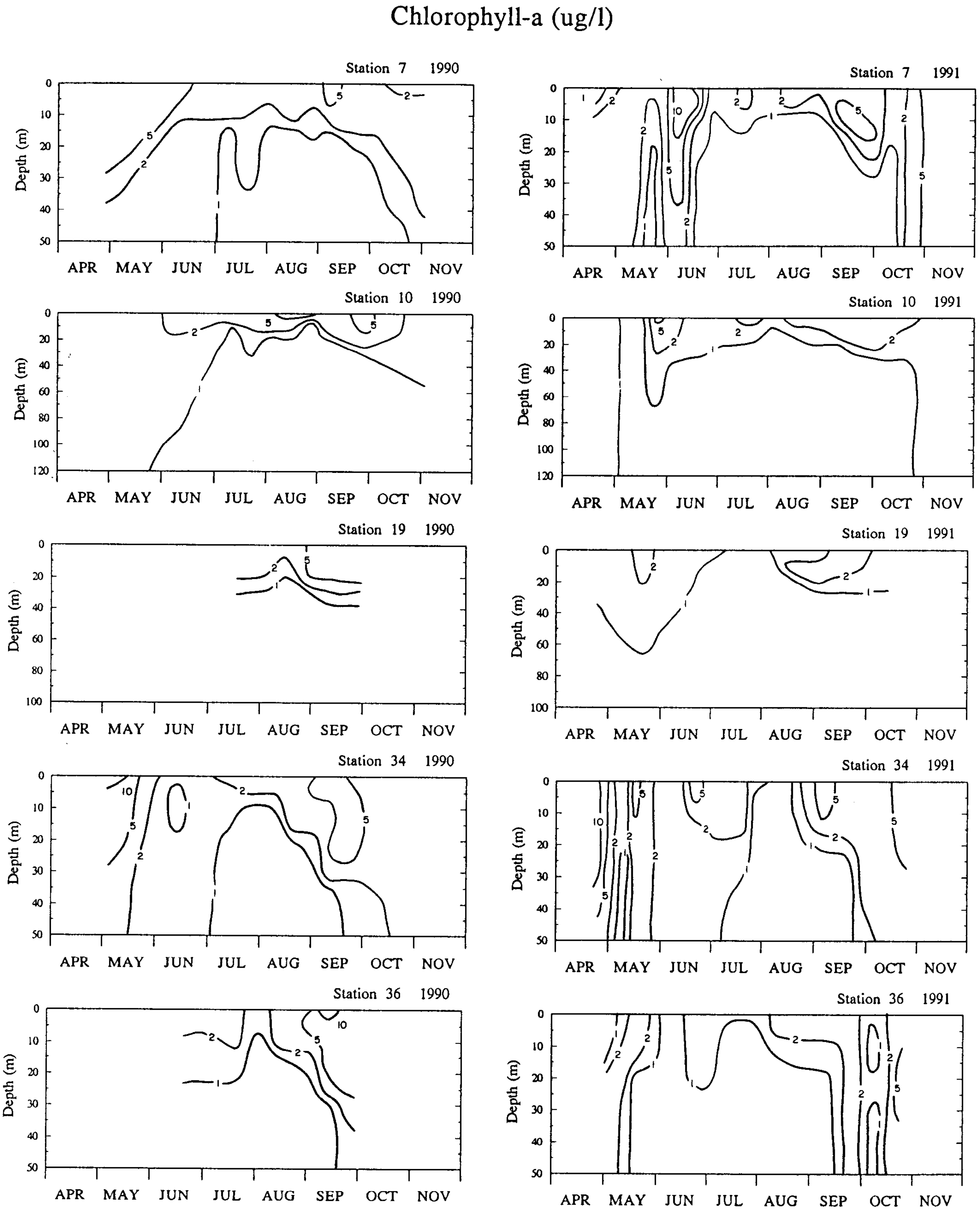


Figure 17. Depth-time plots for chlorophyll-a at five vertical profile stations.

Total Kjeldahl Nitrogen (mg/l)

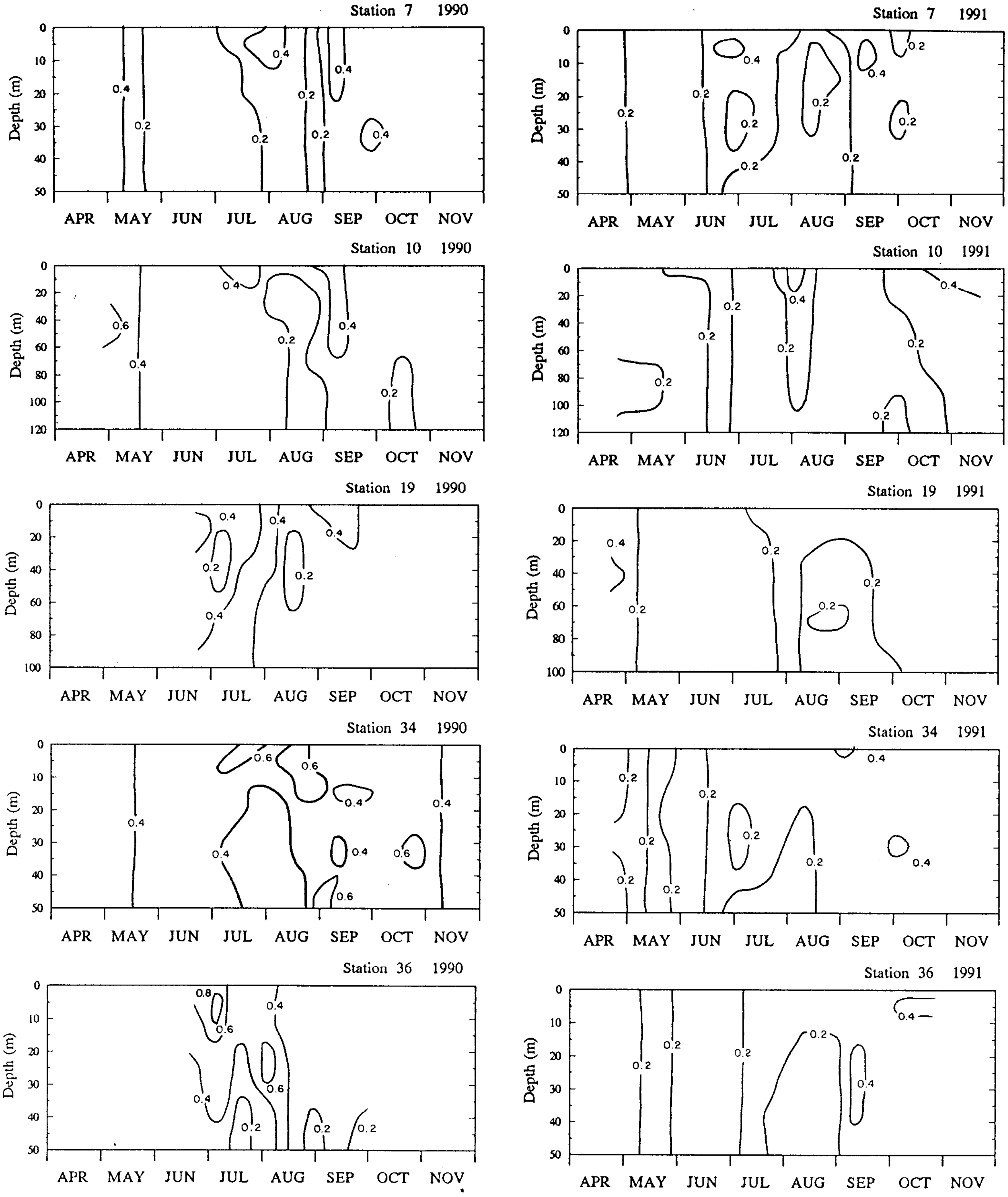


Figure 18. Depth-time plots for total Kjeldahl nitrogen at five vertical profile stations.

growth. At no depth or time was a TN/TP ratio seen as low as the critical value of 7, considered to be the point where nitrogen becomes the limiting nutrient in fresh water (Smith, 1990).

Nitrate/nitrite concentrations (Figure 19) were slightly elevated in the hypolimnion during the summer at all five vertical profile stations. Nitrate/nitrate levels were significantly lower at station 34 in the Northeast Arm than at the other vertical profile stations. Lower nitrate concentrations in the Northeast Arm were previously noted by Gruendling and Malanchuk (1974). No reason for the nitrate depletion is apparent, since the total nitrogen concentration and the TN/TP ratios at station 34 are similar to levels found at the other stations.

Figure 20 shows dissolved and total silica results for 1991. As indicated in the Methods section, "total" silica was operationally defined for this study as the analytically reactive fraction measured following alkaline digestion. The difference between total silica and dissolved silica is presumed to represent primarily silica bound in diatom frustules.

There were strong seasonal and vertical patterns in both dissolved and total silica concentrations observed at all profile stations. Maximum dissolved and total silica levels were seen in the spring and fall, with silica depletion to levels below 0.1 mg/l (as SiO_2) occurring during the summer. Silica accumulated in the hypolimnion to levels of 1-2 mg/l. Silica concentrations were similar among all five of the vertical profile stations. The silica concentrations and seasonal patterns were also similar to those measured in near-surface waters of Lake Champlain during 1970 by Gruendling and Malanchuk (1974).

As shown in Figure 20, nearly all of the total silica was present in dissolved form at most locations and times. However, an examination of the tabulated data indicated that epilimnetic "particulate" silica concentrations (total minus dissolved) were highest during spring and fall, and near zero during the summer in most cases.

The phytoplankton community in Lake Champlain was found in earlier studies to be dominated by diatoms (Myer and Gruendling, 1979). However, recent studies have found other algal taxa to be more dominant (Brown *et al.*, 1993). The observed seasonal silica patterns are probably controlled by the dynamics of the diatom populations. The summer silica depletion in the euphotic zone is consistent with patterns of seasonal diatom succession observed in Lake Champlain by Brown *et al.* (1993) and in many other lakes (Wetzel, 1975).

The vertical distribution of pH is shown in Figure 21 for the three stations where pH was measured during the study. The observed pH values were nearly always above neutral, even in the hypolimnion. Elevated values in excess of 8.5 were sometimes recorded in the near surface waters during the summer, probably caused by photosynthetic uptake of carbon dioxide.

Vertical patterns of alkalinity (as $CaCO_3$) shown in Figure 22 indicate that the deep water areas of Lake Champlain are characterized by moderate alkalinity levels, generally in the range of 40-55 mg/l. Seasonal and vertical variations in alkalinity during 1990 were extremely minor at these stations. There were, however, some general differences between stations, with slightly higher levels at the more southern stations (7 and 10), and with the lowest alkalinity values found at station 34 in the Northeast Arm.

The north-south alkalinity gradient in Lake Champlain was reported previously by Potash and Henson (1977). However, the alkalinity values measured during 1990 were generally 5-10 mg/l higher than the mean levels reported by Potash and Henson for the period of 1965 to 1974 in

Nitrate/Nitrite-N (mg/l)

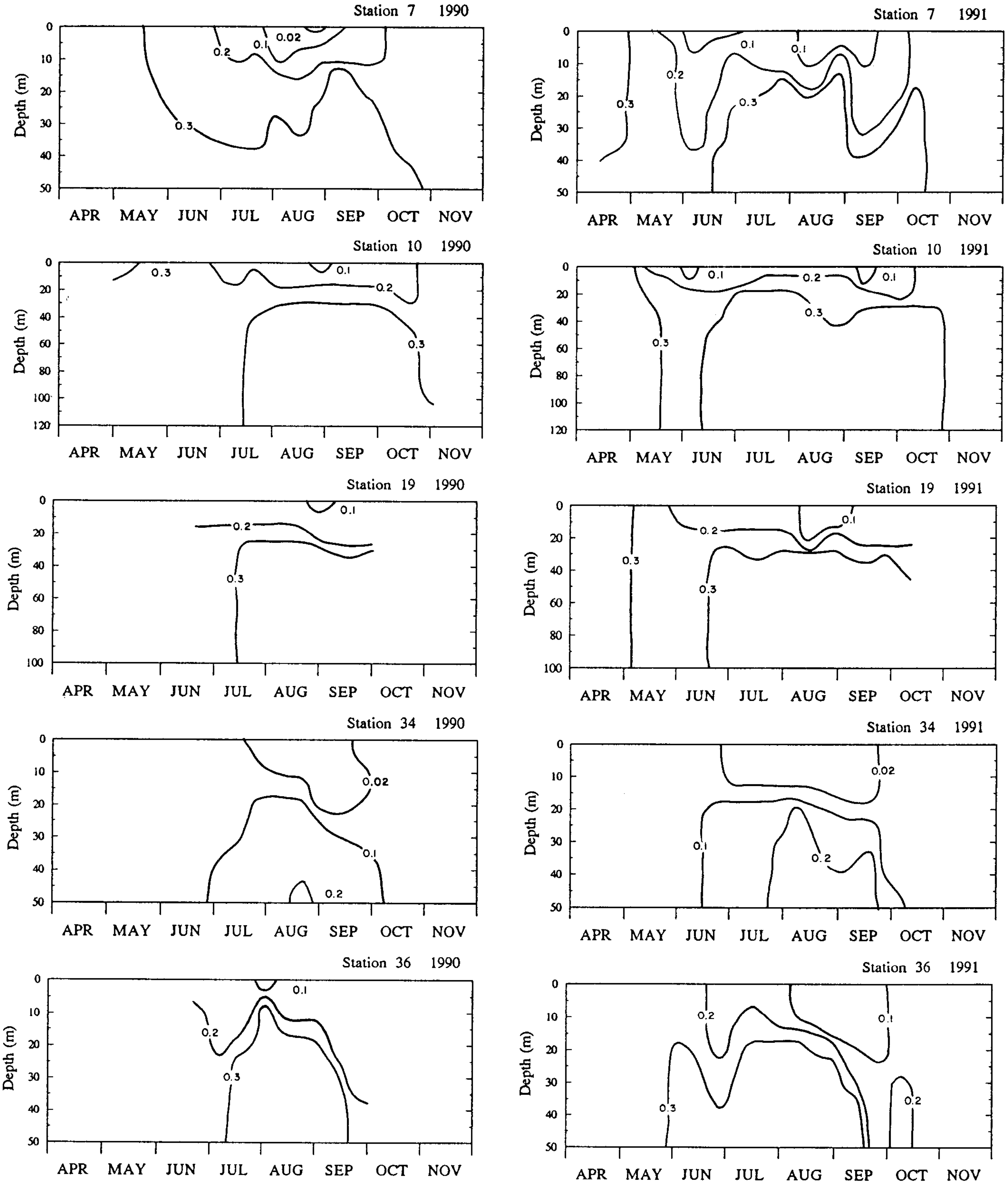


Figure 19. Depth-time plots for nitrite/nitrate-N at five vertical profile stations.

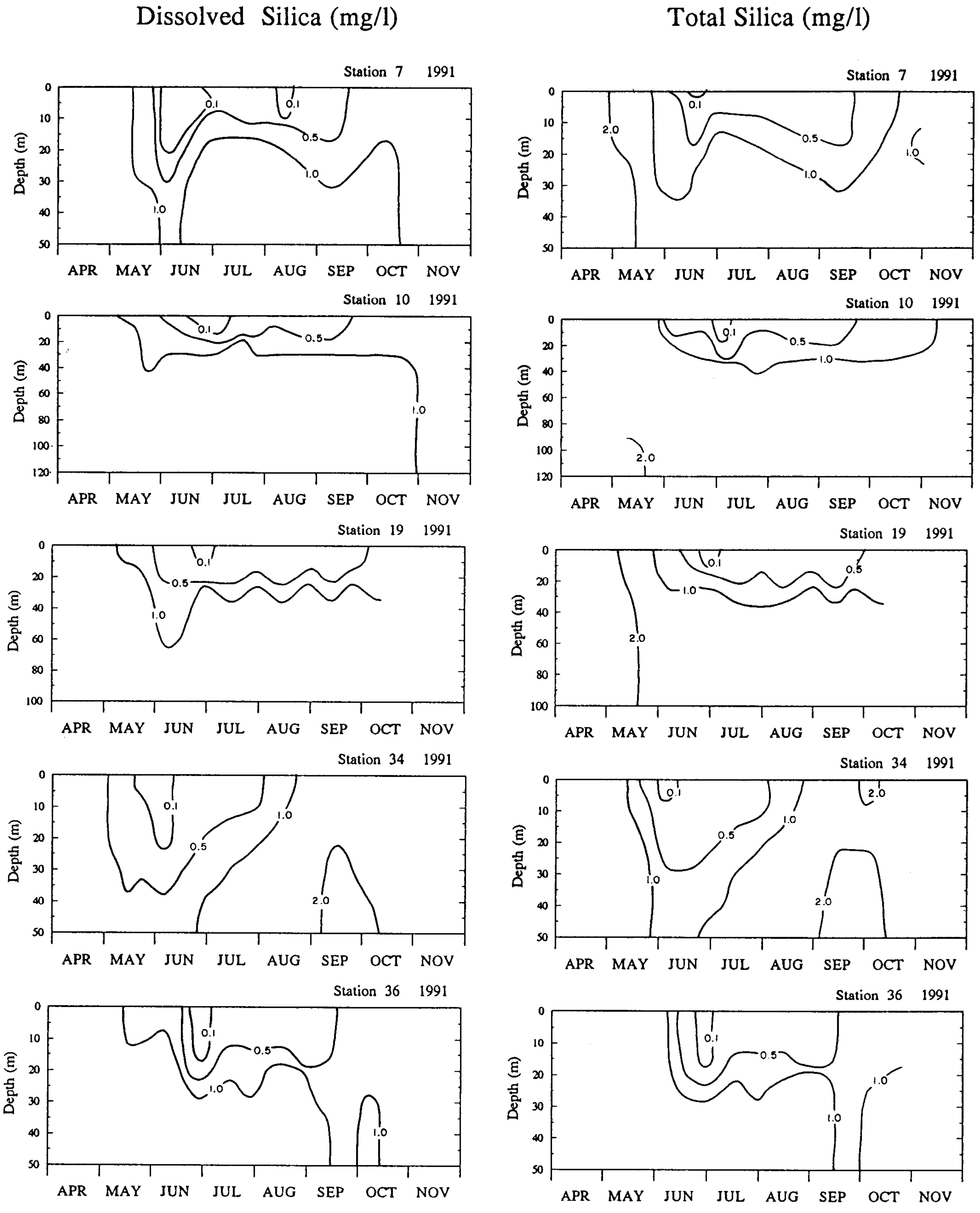


Figure 20. Depth-time plots for dissolved and total silica at five vertical profile stations.

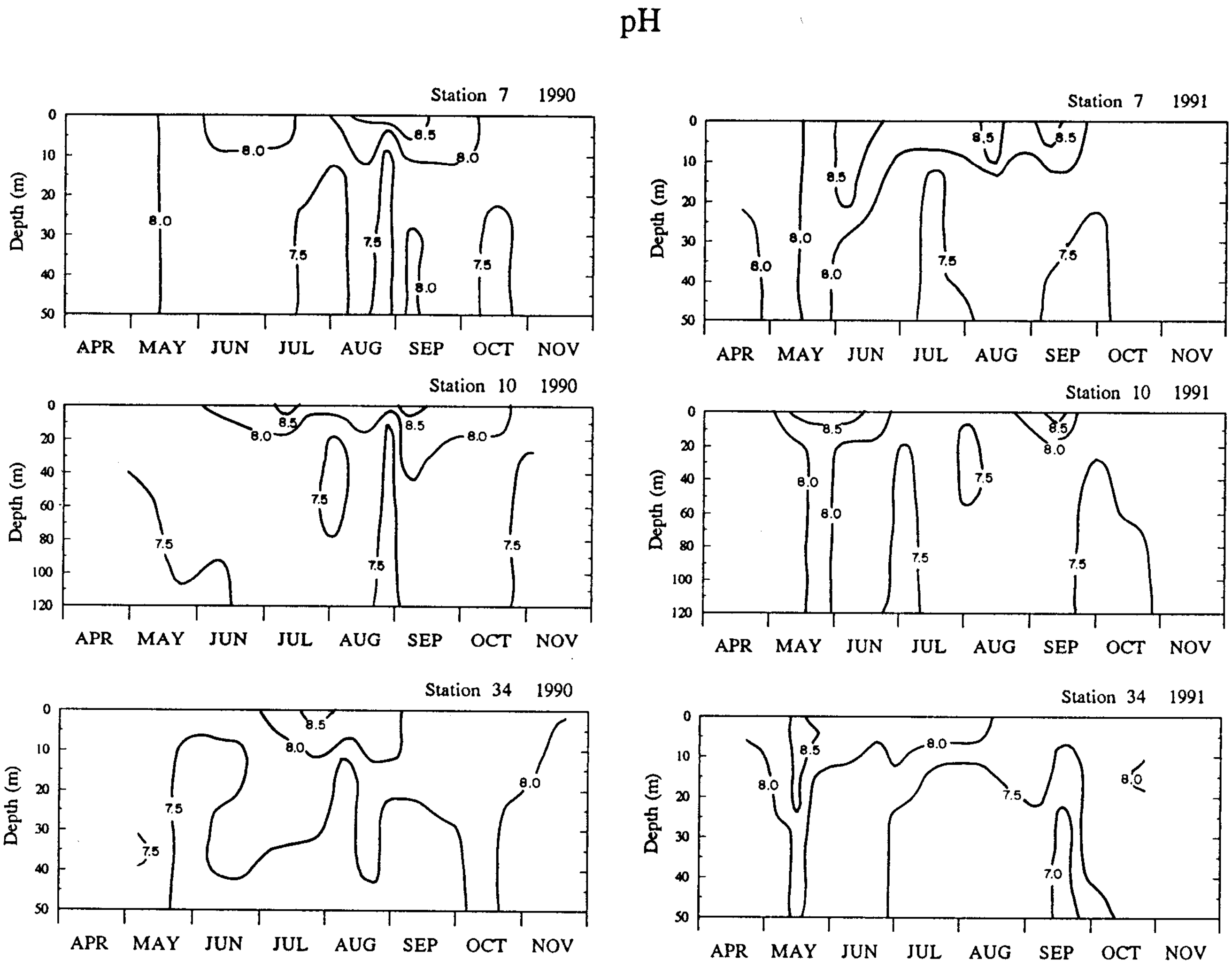


Figure 21. Depth-time plots for pH at three vertical profile stations.

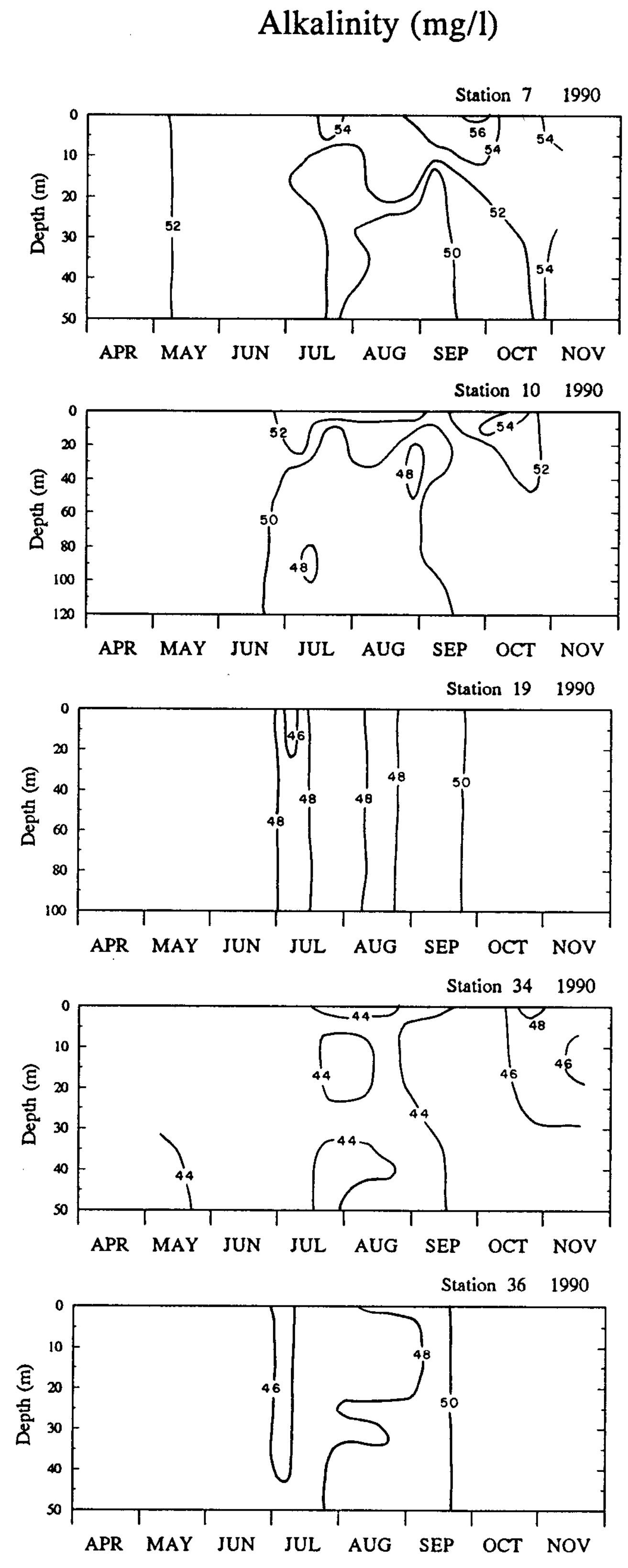


Figure 22. Depth-time plots for alkalinity at five vertical profile stations.

corresponding lake areas. This apparent alkalinity increase should be investigated further because it may indicate a significant water quality change in Lake Champlain during recent decades.

Total ammonia and total suspended solids were measured at the vertical profile stations during 1990, but the results are not presented here graphically because the levels were nearly always below the laboratory analytical detection limits of 0.02 and 1.0 mg/l, respectively. If these parameters are included in future monitoring programs on Lake Champlain, then field and laboratory procedures resulting in lower detection limits should be employed.

Tributary Flows and Loadings

Distribution of Tributary Sampling Effort

Table 10 summarizes the distribution of tributary total phosphorus sampling effort that was achieved with respect to high flow conditions during the 26 month sampling period of March 1990 through April 1992. The desired sampling bias in favor of high flow conditions was attained to a varying degree among the tributaries. Table 10 shows that the total number of samples per stream ranged from 84-107 for the Vermont/Quebec tributaries, and from 36-115 for the New York sites. The percentage of samples taken during high flow conditions (exceeding the upper 10th percentile of all daily flows) ranged from 23-36% at the Vermont/Quebec stations, and from 13-29% in New York. The percentage of all high flow days that were sampled ranged from 25-44% in Vermont and Quebec, and from 12-32% in New York.

Loading Estimates

All tributary phosphorus and chloride loading estimates for this study were developed using the FLUX program (Walker, 1987, 1990). This program was designed for situations where a continuous daily flow record is available, combined with discrete water quality samples obtained throughout the range of flow conditions. The program provides a choice of several alternative loading estimation methods, from which the optimum method for the study or the individual site can be selected. Stratification with respect to flow interval or season can be used in the FLUX procedures in order to reduce the variance of the loading estimates.

The FLUX program also produces statistical error estimates for all loading terms, and provides several diagnostic data review procedures to assist in the selection of an appropriate load estimation method and stratification scheme. Errors are expressed as the coefficient of variation (C.V.), which is the standard error of the estimate expressed as a fraction of the predicted value. Errors in the loading estimates are assumed to be log-normally distributed using a natural log scale. The loading and error estimation procedures used in the FLUX program are fully documented in Walker (1987, 1990).

The average daily flow records obtained for each gage station from the U.S. Geological Survey database and other sources and the tributary sampling data were used in the FLUX program to develop mean loading estimates for chloride, total phosphorus, and dissolved phosphorus for each tributary. Flow and sample data from the entire 26 month sampling period of March 1990 to April 1992 were used to calibrate the concentration *vs.* flow relationships employed by the FLUX program. However, estimates of annual mean flows and loadings developed for this study were restricted to specific 12 or 24 month intervals within the sampling period so that the estimates would be based on complete annual hydrologic cycles.

Table 10. Distribution of tributary sampling effort with respect to flow, March 1990 to April 1992. "High flow" is defined as an average daily flow in the upper 10 percent flow interval for the stream during the sampling period.

Tributary	Number of Samples	Percent of Samples Taken at High Flow	Percent of High Flow Days That Were Sampled
Vermont/Quebec			
Winooski	104	31	41
Missisquoi	98	27	33
Lamoille	98	35	43
Otter	107	30	41
Pike	87	23	25
Poultney	88	24	27
Lewis	97	31	38
Rock	87	28	30
Little Otter	98	26	31
Mill	89	36	41
LaPlatte	97	34	42
Malletts	98	28	30
East	84	27	29
Stone Bridge	88	36	39
Stevens	107	36	44
Indian	107	29	38
New York			
Saranac	69	20	17
Ausable	68	28	24
Mettawee	115	24	32
Mettawee/Barge Canal	89	26	29
Bouquet	66	27	23
Great Chazy	62	24	19
Putnam	39	26	13
Little Ausable	67	13	15
Little Chazy	67	22	19
Salmon	66	27	23
Mill (Port Henry)	37	27	13
Mt. Hope	36	28	12
Mill (Putnam Sta.)	36	28	13
Highlands Forge	66	21	16
Hoisington	69	29	22

Examination of concentration *vs.* flow plots indicated that most of the Lake Champlain tributaries had significant (either positive or negative) relationships between chloride or total phosphorus concentration and daily flow. Therefore, a regression-based load estimation procedure was selected, using Walker's (1990) method 6. In this method, a log-log regression relationship is developed between concentration and flow within each flow stratum, and applied with a correction for bias to each daily flow value to produce an estimate of the mean loading rate for the period. Error estimates for the mean loading values were obtained by the FLUX program using on a jackknifing procedure.

The same regression method was applied for all three water quality parameters, for all tributaries. This approach sacrifices optimum (lowest error) estimates for each stream and each parameter, in favor of consistency of method across all streams and parameters. In practice, however, it was found that the mean loading values and their standard errors were generally similar across all alternative load estimation methods provided by the FLUX program.

The concentration *vs.* flow relationships were stratified with respect to flow, and in a few cases by season, to improve the precision of the loading estimates. The boundaries between flow strata were selected by visual examination of concentration *vs.* flow log-log plots to choose the flow values for each stream and each parameter at which the slope of the relationship most obviously changed. The FLUX program developed estimates of the mean loading rates for each tributary based on individual regression relationships for each flow interval. From one to three flow intervals were used in the load estimation procedure for the Lake Champlain tributaries.

The concentration *vs.* flow relationship for total phosphorus used in the FLUX load estimation procedure for the Winooski River is shown in Figure 23 as an example. Two flow intervals were used to estimate total phosphorus loading for the Winooski River, with a flow interval boundary at 1500 hm^3/yr (average daily flow at the gage station). A regression relationship was derived for each flow interval.

Regression residuals were examined for each tributary and each sample variable to detect any systematic dependence on flow or time. In a small number of cases, it was necessary to stratify the regression relationships by season in order to eliminate a residual dependence on time. The load estimation procedures described above produced a generally random pattern of regression residuals for all tributaries and sample parameters, indicating that appropriate stratification schemes were employed. The stratum boundaries used in the FLUX program load estimation procedures for Lake Champlain tributaries are listed in Table 11.

Estimates of mean flows and loadings of chloride, total phosphorus, and dissolved phosphorus in each tributary are given in Table 12. The estimates in Table 12 apply to the two year period of March 1990 to February 1992. The loading estimates given in Table 12 were calculated in the FLUX program using the flow values from the appropriate gage station, and then multiplied by a total/gaged drainage area ratio factor (from Table 4), in order to represent loads delivered to the lake at the tributary mouths.

Loading rates given in Table 12 for the Mettawee River were based on data obtained at the Mettawee/Barge Canal station (METT01, Table 5). Comparison of samples obtained on the same day at this station and at the upstream station on the Mettawee River (METT02) indicated that significantly higher mean concentrations of chloride, total phosphorus, and dissolved phosphorus existed at the barge canal site (paired t-test, $p < .001$). Therefore, results from the barge canal station were presumed to more accurately represent loadings to the lake from the entire Mettawee River and Lake Champlain Barge Canal drainage.

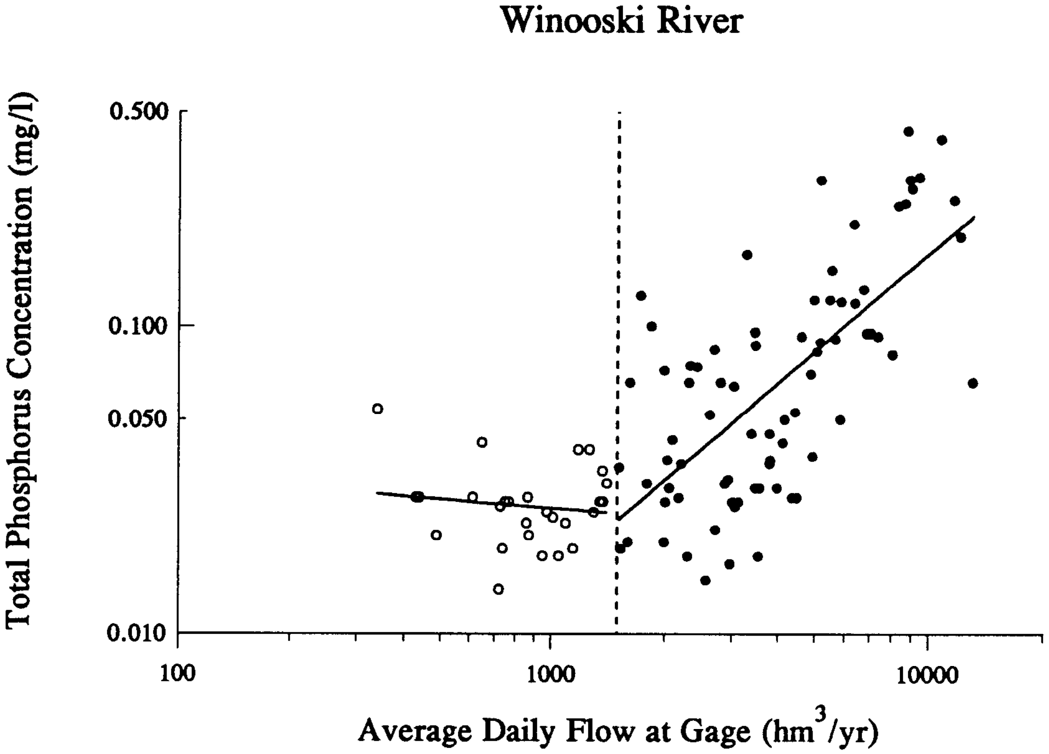


Figure 23. Phosphorus concentration *vs.* flow relationship for the Winooski River, March 1990 to April 1992. Regression lines are shown for the two flow intervals used in the FLUX program load estimation procedure for the Winooski River.

Table 11. Stratum boundaries for FLUX program load estimation procedures applied to Lake Champlain tributaries for the sampling period of March 1990 to April 1992.

Tributary	Sample Variable	Number of Samples[1]	Number of Strata	Upper Boundaries of Flow Strata (hm^3/yr @ gage)	Seasonal Strata Boundaries
Winooski	TP	104	2	1500, 15631	
	DP	88	2	1500, 15631	
	CL	104	1	15631	
Missisquoi	TP	97	2	2000, 18757	
	DP	81	2	2000, 18757	
	CL	98	2	2000, 18757	
Lamoille	TP	97	2	700, 10897	
	DP	82	2	700, 10897	
	CL	97	2	700, 10897	
Otter[2]	TP	107	2	4037	Apr-Jun, Jul-Mar
	DP	89	2	4037	Apr-Jun, Jul-Mar
	CL	106	1	4037	
Pike	TP	85	2	90, 3659	
	DP	69	2	90, 3659	
	CL	87	2	500, 3659	
Poultney	TP	88	2	150, 2671	
	DP	71	2	150, 2671	
	CL	88	1	2671	
Lewis	TP	97	2	100, 1340	
	DP	77	2	200, 1340	
	CL	97	1	1340	
Rock	TP	87	2	300, 1518	
	DP	68	1	1518	
	CL	87	2	400, 1518	
Little Otter	TP	98	2	80, 787	
	DP	79	2	80, 787	
	CL	98	1	787	
Mill	TP	88	2	5, 1679	
	DP	73	2	5, 1679	
	CL	88	3	5, 300, 1679	
LaPlatte	TP	97	1	929	
	DP	78	1	929	
	CL	97	1	929	
Malletts	TP	94	2	20, 234	
	DP	80	2	20, 234	
	CL	92	1	234	
East	TP	83	2	194	Jun-Aug, Sep-May
	DP	65	2	194	Jun-Aug, Sep-May
	CL	84	1	194	
Stone Bridge	TP	88	1	228	
	DP	73	1	228	
	CL	86	1	228	
Stevens	TP	93	2	20, 179	
	DP	69	3	3, 30, 179	
	CL	90	2	3, 179	
Indian	TP	105	2	12, 143	
	DP	86	1	143	
	CL	105	1	143	
Saranac	TP	69	2	2000, 5618	
	DP	59	1	5618	
	CL	69	1	5618	
Ausable	TP	68	2	300, 5672	
	DP	59	1	5672	
	CL	68	1	5672	

Tributary	Sample Variable	Number of Samples[1]	Number of Strata	Upper Boundaries of Flow Strata (hm^3/yr @ gage)	Seasonal Strata Boundaries
Mettawee/Barge Canal	TP	89	2	400, 1795	
	DP	73	2	600, 1795	
	CL	89	1	1795	
Bouquet	TP	66	2	200, 4100	
	DP	57	2	200, 4100	
	CL	66	1	200, 4100	
Great Chazy	TP	62	2	200, 3555	
	DP	52	2	200, 3555	
	CL	62	1	3555	
Putnam	TP	38	1	1313	
	DP	33	1	1313	
	CL	38	1	1313	
Little Ausable	TP	57	1	1152	
	DP	49	1	1152	
	CL	57	2	40, 1152	
Little Chazy	TP	67	2	80, 612	
	DP	57	2	80, 612	
	CL	67	3	8, 80, 612	
Salmon	TP	66	1	851	
	DP	56	1	851	
	CL	66	2	200, 851	
Mill (Port Henry)	TP	37	1	672	
	DP	31	1	672	
	CL	38	1	672	
Mt. Hope	TP	36	2	10, 249	
	DP	31	1	249	
	CL	36	1	249	
Mill (Putnam Sta.)	TP	36	2	8, 255	
	DP	29	1	255	
	CL	36	1	255	
Highlands Forge	TP	66	2	10, 125	
	DP	57	2	20, 125	
	CL	66	2	2, 125	
Hoisington	TP	66	2	4, 193	
	DP	57	1	193	
	CL	65	1	193	

[1] After averaging of samples obtained on the same date and exclusion of some regression outliers.

[2] Flows are the sum of the Otter Creek (at Middlebury) and the New Haven River gage flows.

Table 12. **Tributary flow and loading estimates for the period of March 1990 to February 1992. Flow and loading values apply to the tributary mouths. Flow units are cubic hectometers per year (1 $hm^3/yr = 10^6 m^3/yr$). Loading units are metric tons per year (1 mt/yr = 1000 kg/yr). Concentration values are flow-weighted means. C.V. is the coefficient of variation for the mean loading and concentration estimates.**

		Chloride			Total Phosphorus			Dissolved Phosphorus		
Tributary	**Mean Flow (hm^3/yr)**	**Mean Load (mt/yr)**	**Mean Conc. (mg/l)**	**C.V.**	**Mean Load (mt/yr)**	**Mean Conc. (mg/l)**	**C.V.**	**Mean Load (mt/yr)**	**Mean Conc. (mg/l)**	**C.V.**
Winooski	2,003	24,615	12.3	.029	153.7	.077	.083	24.8	.012	.060
Missisquoi	1,534	9,899	6.5	.024	113.6	.074	.072	27.9	.018	.066
Otter	1,427	13,786	9.7	.016	141.7	.099	.062	86.6	.061	.071
Lamoille	1,423	12,515	8.8	.032	47.2	.033	.079	15.1	.011	.073
Poultney	371	3,890	10.5	.019	32.3	.087	.111	8.3	.022	.117
Pike	347	4,128	11.9	.022	60.1	.173	.120	23.7	.068	.112
Lewis	116	857	7.4	.018	10.4	.089	.108	3.0	.026	.171
Rock	85	838	9.8	.036	30.0	.353	.186	8.6	.100	.111
Little Otter	80	879	11.0	.031	9.8	.122	.065	6.0	.075	.075
LaPlatte	58	1,551	27.0	.040	14.5	.253	.064	10.9	.190	.089
Malletts	40	600	14.9	.024	2.5	.063	.086	.90	.023	.066
Mill	39	1,002	25.9	.032	5.2	.135	.103	2.4	.062	.114
East	35	319	9.2	.045	1.8	.052	.057	1.0	.030	.050
Stevens	16	995	61.4	.042	3.9	.238	.139	1.6	.101	.144
Indian	15	643	43.2	.111	1.2	.081	.087	.24	.017	.038
Stone Bridge	13	236	17.6	.096	1.1	.086	.098	.45	.034	.131
Saranac	877	4,722	5.4	.029	19.2	.022	.062	6.9	.008	.056
Ausable	804	4,684	5.8	.030	24.2	.030	.140	5.9	.007	.092
Mettawee[1]	621	5,087	8.2	.019	35.8	.057	.109	14.5	.023	.137
Mettawee/Barge Canal	621	7,174	11.6	.021	49.2	.079	.076	17.1	.028	.081
Great Chazy	366	3,692	10.1	.032	18.9	.052	.154	7.1	.019	.104
LaChute	365	3,435	8.4	.010	1.5	.004	.061	.81	.002	.086
Bouquet	361	3,349	9.3	.028	21.7	.060	.151	3.9	.011	.189
Putnam	95	524	5.5	.039	2.3	.024	.227	.55	.006	.083
Little Ausable	84	797	9.5	.061	4.6	.055	.126	2.0	.024	.078
Salmon	62	363	5.8	.048	2.0	.032	.121	.75	.012	.163
Little Chazy	56	797	14.3	.036	4.2	.074	.162	3.0	.054	.249
Mill (Port Henry)	37	365	9.7	.031	1.3	.035	.105	.25	.007	.101
Mt. Hope	19	16	.80	.051	.15	.008	.163	.07	.004	.069
Mill (Putnam Sta.)	14	103	7.3	.076	.75	.053	.204	.33	.023	.193
Hoisington	13	123	9.5	.043	.87	.067	.182	.26	.020	.169
Highlands Forge	12	91	7.7	.027	.14	.012	.086	.05	.005	.068
TOTAL	11,387	106,980			780			271		

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[1] Excluded from total.

No sampling or flow data were obtained during this study on LaChute Creek, which is the largest ungaged tributary to Lake Champlain (see Table 4). The mean concentration estimates for LaChute Creek given in Table 12 were based on 1990-1991 lake monitoring data from Lake George (New York State Department of Environmental Conservation data), the outlet of which incorporates about 86% of the total drainage area of this stream. The mean flows for LaChute Creek given in Table 12 were estimated from an inter-gage log-log regression ($R^2 = .81$) based on 36 years (1943-1978) of historical annual mean flows recorded at a discontinued U.S. Geological Survey gage station on the LaChute River. Annual mean flows measured concurrently at the gage station on the Poultney River were used in the regression analysis to estimate the mean flow in LaChute Creek during the March 1990 to February 1992 study period. LaChute Creek loading values given in Table 12 were calculated as the product of mean flow and mean concentration.

The annual mean flow rates given in Table 12 for continuously gaged streams were assumed, for purposes of the subsequent lake model error analysis, to have standard errors of 5% of the means (Winter, 1981). Standard errors for the partial record sites and for sites such as LaChute Creek and the ungaged drainages (see below), where inter-gage correlations were used to estimate annual flows, were assumed to be 20% of the mean flow (Winter, 1981).

Flows and Loadings from Ungaged Areas

The gaged tributaries listed in Table 4 drain about 94% of the total watershed area of Lake Champlain. Flows and loadings from the remaining 6% of the lake's drainage area were estimated on an areal proportional basis, using data from appropriate reference stations given in Table 4.

Henson and Gruendling (1977) presented an inventory of all mapped tributaries to Lake Champlain, regardless of size. This inventory was used to provide drainage areas for all tributaries not listed in Table 4. The ungaged tributary areas were summed for each of the lake segments shown in Figure 1, and for both east and west shores where applicable.

The sampled tributaries listed in Table 5 were grouped according to segment and shore and the total flows and loadings of chloride, total phosphorus, and dissolved phosphorus were summed for each segment and shore. Areal water runoff (m/yr) and mass export rates (mt/km^2-yr) for chloride, total phosphorus, and dissolved phosphorus were calculated for each segment and shore, using data from adjacent segments in cases where no sampled tributary discharges existed for a segment. The measured areal runoff and mass export rates were applied to the ungaged drainage areas to produce estimates of flows and mean concentrations for the ungaged areas. This procedure may somewhat over-estimate the loadings from ungaged areas because the data for the monitored tributaries included the effects of upstream point source discharges, which were not present in the ungaged areas.

The results of these calculations are summarized in Table 13. The standard errors of the concentration values given in Table 13 were assumed to be similar to the estimated flow errors (i.e. 20% of the means), for purposes of the subsequent lake model error analysis.

Wastewater Treatment Facilities

Direct Lake Discharges

The sampling results for the 17 wastewater treatment facilities discharging directly to the lake or immediately upstream which were sampled regularly for this study are statistically summarized in Table 14. The 1990-1991 monthly effluent flow rates used in calculating the values given in Table 14

Table 13. **Flow and mean concentration estimates for ungaged drainage areas for the period of March 1990 to February 1992, listed by segment and shore. Lake segments are as shown in Figure 1, and numbered as follows:**

1. **South Lake B**
2. **South Lake A**
3. **Port Henry**
4. **Otter Creek**
5. **Main Lake**
6. **Shelburne Bay**
7. **Burlington Bay**
8. **Cumberland Bay**
9. **Malletts Bay**
10. **Northeast Arm**
11. **St. Albans Bay**
12. **Missisquoi Bay**
13. **Isle LaMotte**

			Monitored Tributaries				Ungaged Areas			
Segment	Shore	Ungaged Area (km^2)	Water Runoff Rate (m/yr)	Chloride Export Rate (mt/km^2-yr)	Tot. Phos. Export Rate (mt/km^2-yr)	Diss. Phos. Export Rate (mt/km^2-yr)	Flow (hm^3/yr)	Chloride Conc. (mg/l)	Tot. Phos. Conc. (mg/l)	Diss. Phos. Conc. (mg/l)
1	E	29	.55	6.16	.044	.014	16.1	11.1	.080	.024
1	W	101	.62	.51	.005	.002	62.3	0.8	.008	.004
2	E	77	.43	3.93	.024	.015	33.3	9.1	.055	.034
2	W	82	.58	3.26	.014	.004	47.9	5.6	.023	.008
3	E	24	.43	3.93	.024	.015	10.4	9.1	.055	.034
3	W	133	.50	4.78	.021	.005	65.9	9.6	.043	.009
4	E	21	.57	5.41	.056	.033	12.0	9.5	.098	.058
4	W	5	.46	4.43	.029	.008	2.3	9.5	.062	.018
5	E	26	.71	8.71	.053	.008	18.4	12.3	.074	.012
5	W	101	.54	3.77	.018	.005	55.0	6.9	.034	.009
6	E	39	.42	11.34	.107	.080	16.4	27.0	.254	.190
7	E	3	.42	11.34	.107	.080	1.3	27.0	.254	.190
8	W	104	.56	2.91	.012	.004	57.9	5.2	.021	.008
9	E	46	.73	6.84	.026	.008	33.7	9.3	.035	.011
10	E	95	.42	7.31	.035	.014	39.7	17.5	.083	.034
11	E	6	.47	16.67	.073	.034	2.8	35.2	.155	.071
12	E	63	.68	5.16	.071	.021	42.8	7.6	.105	.031
13	W	73	.46	4.83	.023	.011	33.6	10.5	.050	.024

Table 14. Wastewater treatment facility sampling results for the direct lake discharges during 1990-1991.

Facility	Average Flow (hm^3/yr)	Number of Samples	Chloride Volume Weighted Mean (mg/l)	Chloride C.V.	Total Phosphorus Volume Weighted Mean (mg/l)	Total Phosphorus C.V.
Burlington Main	5.35	24	104	.078	2.12	.12
Burlington North	1.81	23	71	.054	1.91	.10
International Paper Co.	24.07	19	415	.046	0.34	.17
Plattsburgh/Champlain Park	0.18	19	150	.157	1.62	.20
Port Henry	0.81	17	56	.137	1.92	.11
Plattsburgh City	11.54	20	76	.061	1.74	.15
Northwest State Correctional	0.02	25	56	.050	0.17	.28
St. Albans City	2.92	25	78	.084	0.27	.14
So. Burlington Bart. Bay	1.00	22	118	.076	0.58	.18
Shelburne F.D. #1	0.42	22	101	.042	0.67	.16
Shelburne F.D. #2	0.38	21	119	.051	0.70	.22
Swanton	1.02	22	122	.082	2.38	.34
Ticonderoga	1.25	10	96	.244	1.12	.17
Vergennes	0.65	19	70	.068	0.70	.20
Westport	0.12	20	78	.068	1.81	.13
Whitehall	0.74	8	55	.084	0.96	.17
Wyeth-Ayerst, Chazy	0.054	2	466	--	83.80	.28

were obtained from plant operation records on file at the Vermont Department of Environmental Conservation and the New York State Department of Environmental Conservation. The mean flow values, which were generally measured at well-calibrated flume or weir structures, were assumed to have standard errors of 5% of the means (Winter, 1981). The concentration statistics given in Table 14 were calculated using procedures for ratio estimates (W. W. Walker, pers. comm., 6/18/92). The chloride concentration at the Wyeth-Ayerst, Chazy facility, which was based on a single sample, was assumed to have a standard error of 50%, for lake model error analysis purposes.

There were operational changes during the study period at some of the facilities listed in Table 14 that affected the results. The 1990-1991 average effluent total phosphorus value for the Swanton facility includes results obtained both before and after phosphorus removal treatment became operational in late 1990. The 1991 average effluent total phosphorus concentration in the Swanton discharge dropped to 0.50 mg/l. A reduction in effluent total phosphorus levels also occurred at the Plattsburgh City facility as a result of the cessation of pulping operations at the Georgia-Pacific Mill in 1991 and other operational changes. Phosphorus loading from the Plattsburgh City facility averaged 8,400 kg/yr during 1993 (J. Ruff, pers. comm., 12/6/93).

The average flow rate for the St. Albans City facility given in Table 14 incorporates a correction factor of 0.67 by which the reported flows were multiplied to adjust for the effects of an improperly located flow meter at the plant that monitored some recirculated flows. The correction factor was 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.

Basin-Wide Point Source Inventory

An inventory of all permitted municipal and industrial wastewater phosphorus discharges to surface waters in the Lake Champlain basin is given in Table 15. The data in Table 15 were obtained from plant operation records on file at the Vermont Department of Environmental Conservation, the New York State Department of Environmental Conservation, and the Quebec Ministry of the Environment. This inventory was compiled for the purpose of estimating the point source component of the phosphorus loadings to Lake Champlain measured during this study. This information was used to support the development of a phosphorus load reduction strategy between point and nonpoint sources, and between individual sub-basins in Vermont, New York, and Quebec.

The information provided in Table 15 reflects to the greatest extent possible the operational conditions existing at these facilities during the 1990-1991 study period. Average flows recorded during 1990-1991 are listed, along with the permitted flows for each facility. Some of the facilities listed in Table 15 had regular (e.g. monthly) effluent total phosphorus sampling requirements included as conditions of their discharge permits. Where effluent phosphorus samples were obtained on a monthly or greater frequency, only recent data (1990-1991) were included in Table 15. For those 17 direct lake discharges sampled for this study, the results from Table 14 were used in this inventory. Other facilities had more limited phosphorus data available and all effluent total phosphorus samples obtained since 1984 were summarized in Table 15 in those cases. Best estimates of the effluent mean total phosphorus concentrations were used for those few facilities where no data existed.

The Colchester, Vermont plant was consolidated with the South Burlington Airport Parkway facility in late 1990, and is therefore not listed as a discharge in Table 15. The Weed Fish Culture Station in Grand Isle, Vermont began operation in late 1991, and loadings from this facility are not included in Tables 14 and 15. The Agrimark Inc. discharge has since been diverted to a new municipal facility serving Troy/Jay, Vermont, which became operational in 1992.

Table 15. Wastewater treatment facility phosphorus discharge inventory for the Lake Champlain basin, 1990-1991.

Facility	State	Type[1]	Basin[2]	1990-1991 Mean Flow (mgd)	Permitted Flow (mgd)	Mean TP (mg/l)	Number of Samples	1990-1991 Phosphorus Load (kg/yr)
Alburg	VT	MUN	LAKE	0.069	0.130	0.03	27	3
Barre City	VT	MUN	WINO	2.938	3.800	1.36	18	5518
Benson	VT	MUN	POUL	0.009	0.018	4.34	2	54
Brandon	VT	MUN	OTTE	0.327	0.700	1.40	4	632
Burlington East	VT	MUN	WINO	0.805	1.00	0.68	58	756
Burlington Main	VT	MUN	LAKE	3.841	5.300	2.12	24	11245
Burlington North	VT	MUN	LAKE	1.303	2.000	1.91	23	3437
Castleton	VT	MUN	POUL	0.263	0.360	2.15	4	781
Enosburg Falls	VT	MUN	MISS	0.364	0.450	2.65	1	1332
Essex Jct.	VT	MUN	WINO	1.487	2.750	0.78	69	1602
Fair Haven	VT	MUN	POUL	0.289	0.500	3.02	5	1205
Fairfax	VT	MUN	LAMO	0.040	0.078	3.95	1	218
Hardwick	VT	MUN	LAMO	0.216	0.371	2.75	1	820
Hinesburg	VT	MUN	LAPL	0.177	0.250	17.09	27	4177
Jeffersonville	VT	MUN	LAMO	0.032	0.077	2.00[3]	0	88
Johnson	VT	MUN	LAMO	0.154	0.200	1.82	7	387
Marshfield	VT	MUN	WINO	0.018	0.045	3.94	1	98
Middlebury	VT	MUN	OTTE	1.103	2.200	28.69	11	43702
Milton	VT	MUN	LAMO	0.163	0.225	0.56	1	126
Montpelier	VT	MUN	WINO	2.044	3.970	2.52	12	7113
Morrisville	VT	MUN	LAMO	0.335	0.425	2.70	2	1249
Newport Center	VT	MUN	MISS	0.018	0.042	0.10[3]	0	2
North Troy	VT	MUN	MISS	0.082	0.110	1.90	5	215
Northfield	VT	MUN	WINO	1.096	1.630	2.35	5	3557
Orwell	VT	MUN	EAST	0.027	0.033	2.00[3]	0	75
Pittsford	VT	MUN	OTTE	0.081	0.070	2.63	4	294
Plainfield	VT	MUN	WINO	0.062	0.100	4.29	2	367
Poultney	VT	MUN	POUL	0.318	0.350	2.14	5	940
Proctor	VT	MUN	OTTE	0.248	0.325	2.18	2	747
Richford	VT	MUN	MISS	0.347	0.380	1.04	2	498
Richmond	VT	MUN	WINO	0.083	0.222	6.12	8	701
Rutland City	VT	MUN	OTTE	5.407	6.800	2.03	13	15158
Shelburne F.D.#1	VT	MUN	LAKE	0.306	0.310	0.67	22	283
Shelburne F.D.#2	VT	MUN	LAKE	0.278	0.450	0.70	21	269
Sheldon Springs	VT	MUN	MISS	0.025	0.054	2.04	24	70
South Burlington Airport Park.	VT	MUN	WINO	1.313	2.300	0.76	67	1378
South Burlington Bart. Bay	VT	MUN	LAKE	0.727	0.800	0.58	22	582
St. Albans City	VT	MUN	LAKE	2.114	4.000	0.27	25	788
Stowe	VT	MUN	WINO	0.147	0.250	0.22	26	45
Swanton	VT	MUN	LAKE	0.742	0.900	2.38	22	2439
Troy/Jay	VT	MUN	MISS		0.200			0
Vergennes	VT	MUN	LAKE	0.467	0.660	0.70	19	451
Wallingford	VT	MUN	OTTE	0.110	0.120	2.98	4	453
Waterbury	VT	MUN	WINO	0.239	0.510	4.95	7	1634
West Pawlet	VT	MUN	METT	0.020	0.040	6.30	1	174
West Rutland	VT	MUN	OTTE	0.236	0.325	2.00[3]	0	652
Williamstown	VT	MUN	WINO	0.075	0.150	2.11	5	219
Winooski	VT	MUN	WINO	0.837	1.200	0.52	62	601
Agrimark	VT	IND	MISS	0.044		15.38	14	935

Facility	State	Type[1]	Basin[2]	1990-1991 Mean Flow (mgd)	Permitted Flow (mgd)	Mean TP (mg/l)	Number of Samples	1990-1991 Phosphorus Load (kg/yr)
Green Mt. Trout Farm	VT	IND	LEWI		0.144	0.08	1	16[4]
IBM	VT	IND	WINO	1.957	5.000	0.26	48	703
Northwest State Correctional	VT	IND	LAKE	0.018	0.040	0.17	25	4
Pittsford Fish Hatchery	VT	IND	OTTE		5.000	0.10[3]	0	691[4]
Rock Tenn	VT	IND	MISS	2.515	3.500	0.40	71	1389
Vermont Whey	VT	IND	LAMO	0.333	0.425	0.36	124	166
Weed Fish Culture Station	VT	IND	LAKE		11.500			0
Abercorn	PQ	MUN	MISS	0.040		4.00[3]	0	221
Bedford	PQ	MUN	PIKE	0.729		4.00[3]	0	4027
Eastman	PQ	MUN	MISS	0.064		4.00[3]	0	354
Notre-Dame-de-Stanbridge	PQ	MUN	PIKE	0.111		4.00[3]	0	613
Potton	PQ	MUN	MISS	0.079		4.00[3]	0	436
Stanbridge Station	PQ	MUN	PIKE	0.066		4.00[3]	0	365
Stukely-Sud	PQ	MUN	MISS	0.016		4.00[3]	0	88
Sutton	PQ	MUN	MISS	0.268		4.00[3]	0	1480
Synder et Fils	PQ	IND	PIKE	0.170		4.00[3]	0	939
Ausable Forks	NY	MUN	AUSA	0.043	0.147	3.70	1	220
Champlain	NY	MUN	GCHA	0.323	0.400	2.05	3	914
Dannemora	NY	MUN	SARA	0.699	1.500	3.08	3	2973
Fort Ann	NY	MUN	BACA	0.067	0.110	2.70	1	250
Granville	NY	MUN	METT	0.793	0.650	1.50	1	1643
Keeseville	NY	MUN	AUSA	0.264	0.300	3.25	3	1185
Lake Placid	NY	MUN	AUSA	1.341	2.500	2.27	3	4204
Peru	NY	MUN	LAUS	0.254	0.500	4.00	1	1403
Peru/Valcour	NY	MUN	LAKE	0.005	0.048	2.27	4	16
Plattsburgh City	NY	MUN	LAKE	8.402	16.000	1.74	20	20190
Plattsburgh/Champlain Park	NY	MUN	LAKE	0.131	0.162	1.62	19	293
Port Henry	NY	MUN	LAKE	0.588	0.440	1.92	17	1559
Rouses Point	NY	MUN	LAKE	0.814	2.000	1.62	4	1821
Saranac Lake	NY	MUN	SARA	1.884	2.800	2.06	3	5360
St. Armand	NY	MUN	SARA	0.032	0.060	3.89	3	172
Ticonderoga	NY	MUN	LAKE	0.902	1.000	1.12	10	1395
Wadhams	NY	MUN	BOUQ	0.006	0.015	5.10	1	42
Westport	NY	MUN	LAKE	0.082	0.120	1.81	20	205
Whitehall	NY	MUN	LAKE	0.539	0.600	0.96	8	715
Adirondak Fish Culture Station	NY	IND	SARA	2.623	3.600	0.05	12	181
Altona Correctional	NY	IND	GCHA	0.066	0.080	2.00[3]	0	182
Great Meadows Correctional	NY	IND	BACA	0.371	0.356	2.00[3]	0	1025
International Paper Co.	NY	IND	LAKE	17.464		0.34	19	8200
Washington Correctional	NY	IND	BACA	0.110	0.105	2.00[3]	0	304
Wyeth-Ayerst, Chazy	NY	IND	LAKE	0.039	0.026	83.80	2	4513

[1] **Type: MUN = Municipal; IND = Industrial**

[2] **Basin: LAKE = Outfall in lake or immediately upstream; BACA = Barge Canal; See Table 5 for other abbreviations**

[3] **Estimated value; no data available.**

[4] **Load calculated using permitted flow.**

Only partial information was available for the Quebec discharges, as many of these are in various stages of planning for the construction and upgrade of sewer and treatment facilities. The operational data for the Quebec facilities were supplied by the Quebec Ministry of the Environment (M. Simoneau, 3/18/92, and D. Brouillette, 11/23/93, pers. comm.).

About half of the facilities listed in Table 15 had phosphorus loading estimates that were based on three or fewer effluent samples. However, these tended to be the smaller discharges, representing only 21% (40 mt/yr) of the total 189 mt/yr point source loading estimate for 1990-1991.

Table 15 indicates that there were 88 municipal and industrial discharges in the Lake Champlain basin operating during 1990-1991, including 54 in Vermont, 25 in New York, and 9 in Quebec. Phosphorus loadings from all wastewater treatment facilities in the basin were calculated from the actual flow and effluent phosphorus data given in Table 15. The point source phosphorus loading estimates listed in Table 15 were distributed among the three jurisdictions as follows. The loading units are metric tons per year (mt/yr = 1000 kg/yr).

Vermont	**121 mt/yr**
New York	**59**
Quebec	**9**
Total	**189 mt/yr**

Previous basin-wide point source phosphorus loading estimates summarized by Bogdan (1978) ranged from 255-327 mt/yr, in comparison with the 1990-1991 rate of 189 mt/yr. The reduction in point source phosphorus loadings to Lake Champlain since the 1970's can be attributed to the implementation of basin-wide phosphorus detergent bans and advanced wastewater treatment for phosphorus removal at some facilities.

Further point source phosphorus reductions below the 1990-1991 loading rates listed in Table 15 have either already occurred since 1991 or are planned for implementation under new phosphorus reduction policies in Vermont, Quebec, and New York. Phosphorus management policies and future loading rates will be presented and discussed in detail in the Phosphorus Load Reduction Strategy section of this report.

Precipitation

Precipitation Sampling

The number of individual precipitation events sampled for chloride and total phosphorus concentration ranged from 12 to 138 among the four precipitation sampling stations. Potentially contaminated samples were eliminated from the data set according to screening procedures described in the Methods section. A further problem with the results was that 44% of all chloride samples and 8% of all total phosphorus samples had concentrations that were below the laboratory analytical detection limits of 0.2 mg/l for chloride and 0.003 mg/l for phosphorus. The "less than" signs associated with these results below detection were ignored in calculating the average concentration and loading estimates given below, thereby resulting in slight overestimates of these terms, particularly for chloride.

The precipitation sampling results are summarized in Table 16. The volume-weighted mean concentrations across all sampled events, and the coefficient of variation (C.V.) of the mean

concentrations, were calculated according statistical procedures for ratio estimates (W.W. Walker, pers. comm., 6/18/92).

Precipitation Volumes

Precipitation data collected daily at several National Oceanic and Atmospheric Administration (N.O.A.A.) weather stations located near Lake Champlain were used to estimate the volume of precipitation falling directly on the lake surface. The N.O.A.A. weather stations used for this purpose are listed below.

Vermont	New York
St. Albans	Chazy, Miner Inst.
South Hero	Plattsburgh
Essex Jct.	Ticonderoga Mill
Burlington Airport	Whitehall

The mean precipitation rate measured at these eight stations during the two year monitoring period of March 1990 to February 1992 was 0.96 m/yr, with a standard error of 0.04 m/yr. The range among stations was 0.80-1.13 m/yr.

Precipitation Loading Estimates

Precipitation loading estimates for chloride and total phosphorus based on the eight-station average precipitation rate of 0.96 m/yr are given in Table 16. Deposition rates ranged from 259-298 mg/m^2-yr for chloride and 8.4-27.1 mg/m^2-yr for phosphorus. The lowest phosphorus deposition rate was recorded at the "wet only" precipitation station at Underhill, VT. Higher phosphorus deposition rates were measured at the three bulk precipitation stations, with the highest value recorded at the South Hero, VT site, located in an agricultural setting.

The mean chloride concentrations in precipitation measured during this study (Table 16) were substantially lower than the volume-weighted mean chloride level of 1.77 mg/l reported by Henson and Vibber (1969) for two stations located near Lake Champlain sampled during 1967-1968. The reason for this difference is not known. The lower values reported for the present study are more consistent with recent data from National Atmospheric Deposition Program monitoring sites in the Northeast, which typically show average chloride concentrations in the range of 0.1-0.3 mg/l in wet deposition.

The range of total phosphorus deposition rates measured among the four sampling stations (Table 16) includes the value of 18 mg/m^2-yr previously applied to Lake Champlain by Henson and Gruendling (1977) based on a literature review. Phosphorus deposition rates of 18 mg/m^2-yr (bulk), measured at Lake Morey, Vermont (Morgan *et al*. 1984), and 12-27 mg/m^2-yr (wet) measured at various sites in Lake Michigan (Murphy and Doskey, 1975), are also consistent with the range of results for Lake Champlain given in Table 16.

The mean deposition rates among the four sampling stations given in Table 16 correspond to mass loading rates to the lake surface (1,130 km^2) of 314 mt/yr for chloride, and 18 mt/yr for total phosphorus. These results indicate that precipitation is a relatively minor source of phosphorus to Lake Champlain, in comparison with the total tributary loadings.

Table 16. Precipitation sampling results. Deposition rates were calculated based on an eight-station March 1990 - February 1992 average precipitation rate of 0.96 m/yr.

Parameter	Station	Number of Samples	Minimum (mg/l)	Median (mg/l)	Maximum (mg/l)	Volume Weighted Mean (mg/l)	C.V.	Deposition Rate (mg/m^2-yr)
Chloride	BLO	12	<.2	<.2	.8	.27	.17	259
	ORW	119	<.2	.2	1.6	.31	.08	298
	SHE	39	<.2	.2	.8	.28	.09	269
	UND[1]	138	<.2	<.2	.9	.29	.08	278
	Mean (C.V.)							276 (.03)
Total Phos.	BLO	12	.004	.012	.061	.0134	.29	12.9
	ORW	119	<.003	.010	.374	.0167	.21	16.0
	SHE	39	.004	.020	.320	.0282	.26	27.1
	UND[1]	138	<.003	.005	.386	.0087	.24	8.4
	Mean (C.V.)							16.1 (.25)

[1] Wet deposition only

Evaporation

Records of pan evaporation rates for April-October, 1990-1991 were obtained from the N.O.A.A. station in Essex Jct., Vermont. The April-October total pan evaporation values at this site were 0.69 meters during 1990 and 0.82 meters during 1991. These rates were adjusted by a pan-to-lake coefficient of 0.77 (Kohler *et al.*, 1959). Kohler *et al.* (1959) indicated that May-October evaporation rates are about 80% of annual values in the region of Lake Champlain. The April-October evaporation totals measured at the Essex Jct. site were assumed to represent about 90% of the annual total.

With these adjustments, a 1990-1991 average annual evaporation rate of 0.65 meters was calculated for the surface of Lake Champlain (1,130 km^2), corresponding to a water volume loss of 735 hm^3/yr. The 1990-1991 estimate derived from the Essex Jct. data was within the long-term average annual lake evaporation rate range of 0.61-0.66 meters given for the Lake Champlain region by Kohler *et al.* (1959). Annual lake evaporation rates estimated by this method probably have a standard error of about 20% (Winter, 1981).

Water Intakes

Water is withdrawn from Lake Champlain at many locations by municipal, industrial, and private water users. The 1990-1991 mean flow rates at the six largest water intakes in the lake are listed below. The data for the Vermont facilities were obtained from Vermont Department of Environmental Conservation files. Water withdrawal rates at the International Paper Co. were estimated from effluent flow rates (from Table 14), which are the most accurate available measure of the intake flows (S. LeBarron, International Paper Co., pers. comm., 7/13/93). Water withdrawal from other smaller facilities were considered to be insignificant for the purposes of this study.

International Paper Co., NY	**24.1 hm^3/yr**
Champlain Water District, VT	**11.5**
Burlington City, VT	**7.6**
Swanton Village, VT	**1.1**
St. Albans City, VT	**1.0**
Tri-Town Water District, VT	**0.8**

Lake Levels

Lake surface elevations are recorded daily in Lake Champlain at a U.S. Geological Survey gage in Burlington Harbor, Vermont. The lake levels that existed during the 1990-1991 study period are compared in Figure 24 with long term (1939-1991) median and extreme monthly average conditions. Lake levels were generally higher in 1990 than in 1991, particularly during the summer and fall, but neither year was extreme in relation to the long term record. There was a net drop in the lake level of 0.62 m from March 1990 to February 1992.

Quality Assurance Results

A field quality assurance program was conducted according to procedures described in the Methods section. The results of the field duplicate analyses are summarized in Table 17. The relative percent difference (RPD) in concentration between each duplicate sample pair was calculated as the absolute value of the concentration difference, expressed as a percentage of the mean of the two samples. The mean RPD values given in Table 17 were calculated across all duplicate pairs

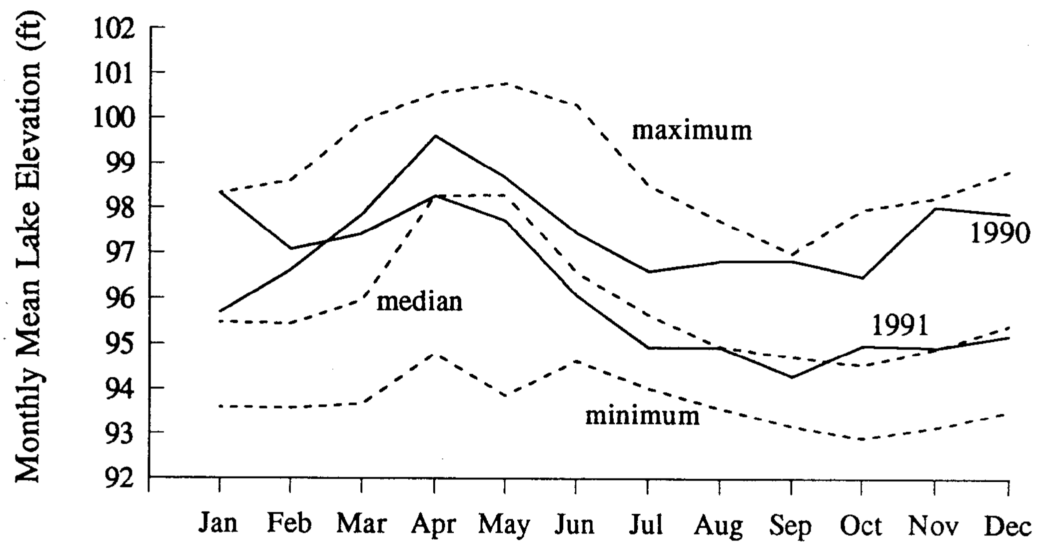

Figure 24. Monthly average lake levels (NGVD) in Burlington Harbor, Lake Champlain during 1990 and 1991, compared with the long-term (1939-1991) median and extreme values. Data are from the U.S. Geological Survey.

Table 17. **Quality assurance field duplicate results, expressed as the mean relative percent difference (RPD) between duplicate pairs.**

	Mean Relative Percent Difference		
Parameter	**Lake**	**Tributaries**	**Wastewater**
Chloride	**1**	**1**	**2**
Total Phosphorus	**13**	**9**	**3**
Dissolved Phosphorus	**15**	**10**	
Ortho-phosphorus	**16**		
Total Kjeldahl Nitrogen	**15**		
Nitrate/Nitrite	**3**		
Chlorophyll-a	**32**		
Total Silica	**9**		
Dissolved Silica	**3**		
Dissolved Oxygen	**2**		
Alkalinity	**1**		
pH	**0.3**		

separately for the lake, tributary, and wastewater samples. No field duplicates were collected on precipitation samples.

Table 17 shows that the mean RPD values were 1-2% for chloride, and 3-13% for total phosphorus, indicating a good level of combined field and laboratory analytical precision for these two most important parameters in this study. Mean RPD values for the other parameters were generally less than 15%, with the exception of chlorophyll-a, which had a relatively poor precision of 32%.

The field blank results for the lake and tributaries are summarized in Table 18. No field blanks were obtained on wastewater or precipitation samples. A "contamination rate" was defined for the purpose of this summary as the percent of all field blank samples with concentrations exceeding twice the practical quantitation limit (PQL) for the analytical method. Table 18 shows that overall contamination rates were low for most parameters.

The chief purpose of the field blank procedure was to detect and correct contamination problems as they occurred during the sampling program. While field blanks exceeded twice the PQL values at a rate of 0-18% (Table 18), instances of more severe contamination were very rare, and the sources of contamination were traced and quickly eliminated during the sampling program.

Lakewide Water, Chloride, and Total Phosphorus Budgets

The sampling results were used to calculate water, chloride, and total phosphorus budgets for Lake Champlain for the two year period of March 1990 to February 1992. Water and mass balance calculations indicate how well the sampling and stream gaging program succeeded in accounting for all major inputs and losses of water, chloride, and phosphorus from the lake. If well balanced water and materials budgets can be demonstrated, then greater confidence in the accuracy of the chloride and phosphorus loading estimates and mass balance modeling analysis can be achieved.

Sources of water, chloride, and phosphorus to Lake Champlain include tributary inflow, direct wastewater discharges, and direct precipitation. Loss routes include outflow via the Richelieu River, water withdrawals, and evaporation from the lake's surface. Change in storage volume as a result of changing lake levels must also be accounted for in the budget calculations. Groundwater inflow and outflow were assumed to be insignificant, pending the results of the water and mass balance calculations.

Total water and mass inputs from tributary streams, ungaged areas, and direct wastewater discharges were obtained from data given in Tables 12, 13, and 14. Inputs from precipitation direct to the lake surface were estimated from mean deposition rates given in Table 16, applied to the lake surface area of 1,130 km^2.

Losses via the lake's outlet were estimated using data from the Water Survey of Canada gage on the Richelieu River. An average flow rate of 13,329 hm^3/yr existed at the Richelieu River gage station during the period of March 1990 to February 1992. This total was multiplied by a factor of 0.961 to account for the additional drainage area between the lake outlet and the downstream gage station, giving a lake outflow rate estimate of 12,809 hm^3/yr. This flow rate was applied to the 1990-1991 mean lake chloride and total phosphorus concentrations measured at the lake station (#49) nearest the outlet (from Table 9) to calculate mass outflow of chloride and total phosphorus. Losses of water, chloride, and phosphorus by water withdrawal from the lake were estimated from the intake.

Table 18. **Quality assurance field blank results. "Contamination rate" values are the percent of all field blank samples having concentrations greater than two times the practical quantitation limit (PQL) for the method.**

		Contamination Rate (%)	
Parameter	PQL[1] (mg/l)	Lake Samples	Tributary Samples
Chloride	.2	7	6
Total Phosphorus	.003	0	2
Dissolved Phosphorus	.003	0	4
Ortho-phosphorus	.003	0	
Total Kjeldahl Nitrogen	.2	0	
Nitrate/Nitrite	.02	0	
Total Ammonia	.02	0	
Chlorophyll-a	.0001[2]	6	
Total Suspended Solids	1.0[2]	0	
Total Silica	.1	9	
Dissolved Silica	.1	18	

[1] Vermont Department of Environmental Conservation Laboratory (1992)

[2] Assumed value; PQL not yet determined

flow data given in the previous section for the six largest water users, using mean chloride and total phosphorus concentrations measured at the lake station nearest to each intake (from Table 9).

Lake level data recorded in Burlington Harbor indicated a net decline in lake level of 0.62 meters from March 1990 to February 1992. This corresponds to a net change in storage volume in the lake of -350 hm^3/yr. Corresponding mass storage changes for chloride and total phosphorus were calculated using the mean concentrations at lake station 49 (from Table 9).

Water, chloride, and total phosphorus budgets for Lake Champlain for the two year period of March 1990 to February 1992 are summarized in Table 19. The budget "error" terms given in Table 19 were calculated as follows:

[Error] = [Total Inputs] - [Total Outputs] - [Storage Change]

In the case of total phosphorus, the error term actually represents the retention of phosphorus within the lake through net sedimentation processes.

Table 19 shows that the water and chloride balance errors were only about 1% and 3% of the total inputs, respectively. The small magnitudes of the errors in the water and chloride budgets, which were calculated from independently estimated hydrologic and loading terms, indicate that all major inputs and losses of water and chloride to Lake Champlain have been accurately accounted for during this study.

The total phosphorus budget for Lake Champlain shown in Table 19 indicates that loadings of phosphorus to Lake Champlain are dominated by tributary inflows. About 81% of the phosphorus loading to the lake was retained within the lake.

Table 19. Water, chloride, and total phosphorus budgets for Lake Champlain for the period of March 1990 to February 1992. "ND" = "no data".

SOURCE	WATER (hm^3/yr)	CHLORIDE (mt/yr)	PHOSPHORUS (mt/yr)
INPUTS			
Gaged Tributaries	11,387	106,980	780
Ungaged Tributaries	552	4,838	29
Direct Wastewater Discharges	52	12,423	57
Direct Precipitation	1,085	312	18
Groundwater Inflow	ND	ND	ND
Total Inputs	13,076	124,553	884
OUTPUTS			
Outlet Flow	12,809	131,933	181
Water Withdrawals	46	549	1
Evaporation	735	0	0
Groundwater Outflow	ND	ND	ND
Total Outputs	13,590	132,482	182
CHANGE IN STORAGE	-350	-3610	-5
ERROR/RETENTION (% of Total Inputs)	-164 (1%)	-4319 (3%)	707 (80%)

MODEL DEVELOPMENT

Modeling Approach

The phosphorus mass balance modeling approach for Lake Champlain was selected for this study based on a number of considerations. Major considerations in model selection included the extent of the available data, the limnological characteristics of the lake, the intended management application of the model results, and the need for optimum simplicity and utility of the model framework.

The complex morphometry of Lake Champlain, the significant water quality differences between regions of the lake, and the segment-specific phosphorus management criteria (Table 2) all dictated that a spatially segmented model be used which could account for phosphorus concentration variations in two (horizontal) spatial dimensions. Vertical profile sampling (see Figure 14) confirmed that vertical phosphorus concentration gradients within the water column were generally much less pronounced than the spatial differences between lake segments. Therefore, each lake segment was modeled as a mixed reactor in which vertically averaged water column phosphorus concentrations were simulated.

The ultimate management application of the model was to determine the phosphorus load reductions necessary to attain the in-lake total phosphorus concentration criteria specified in Table 2. These criteria were intended to be applied as seasonal or annual mean concentrations in central, open water areas of each lake segment. Significant seasonal variations in total phosphorus concentrations do exist in some lake segments (see Figure 7). However, a time-dependent modeling approach would add considerable model complexity, only to provide output data needing later statistical reduction for compatibility with the seasonal or annual mean criteria values. For this reason, a simpler steady-state modeling approach was used as the primary basis for predicting lake total phosphorus concentrations in relation to the numeric water quality criteria.

The modeling approach used for this study was based on the general steady-state mass balance equation for a lake segment given in equation 1, modified from Chapra and Reckhow (1983). A similar model was applied to the entire Great Lakes system by Chapra and Sonzogni (1979) to predict the response of each segment of the Great Lakes to phosphorus loading changes. The resulting series of mass balance equations, one for each lake segment, was solved for various terms during the modeling procedure, using methods described below. A 13 segment model for Lake Champlain based on equation 1 is illustrated schematically in Figure 25.

$$V_i\, dc_i/dt = 0 = W_i + \Sigma_j \{-Q_{ij}c_i + Q_{ji}c_j + E_{ij}(c_j - c_i)\} - S_i \qquad (1)$$

Where

V_i **= volume of segment i (hm^3)**
c_i **= concentration in segment i (mg/l)**
c_j **= concentration in adjacent segment j (mg/l)**
W_i **= direct external mass loading to segment i (mt/yr)**
Q_{ij} **= advective outflow from segment i to adjacent downstream segment j (hm^3/yr)**
Q_{ji} **= advective inflow to segment i from adjacent upstream segment j (hm^3/yr)**
E_{ij} **= diffusive exchange flow between adjacent segments i and j (hm^3/yr)**
S_i **= net internal sedimentation rate in segment i (mt/yr)**

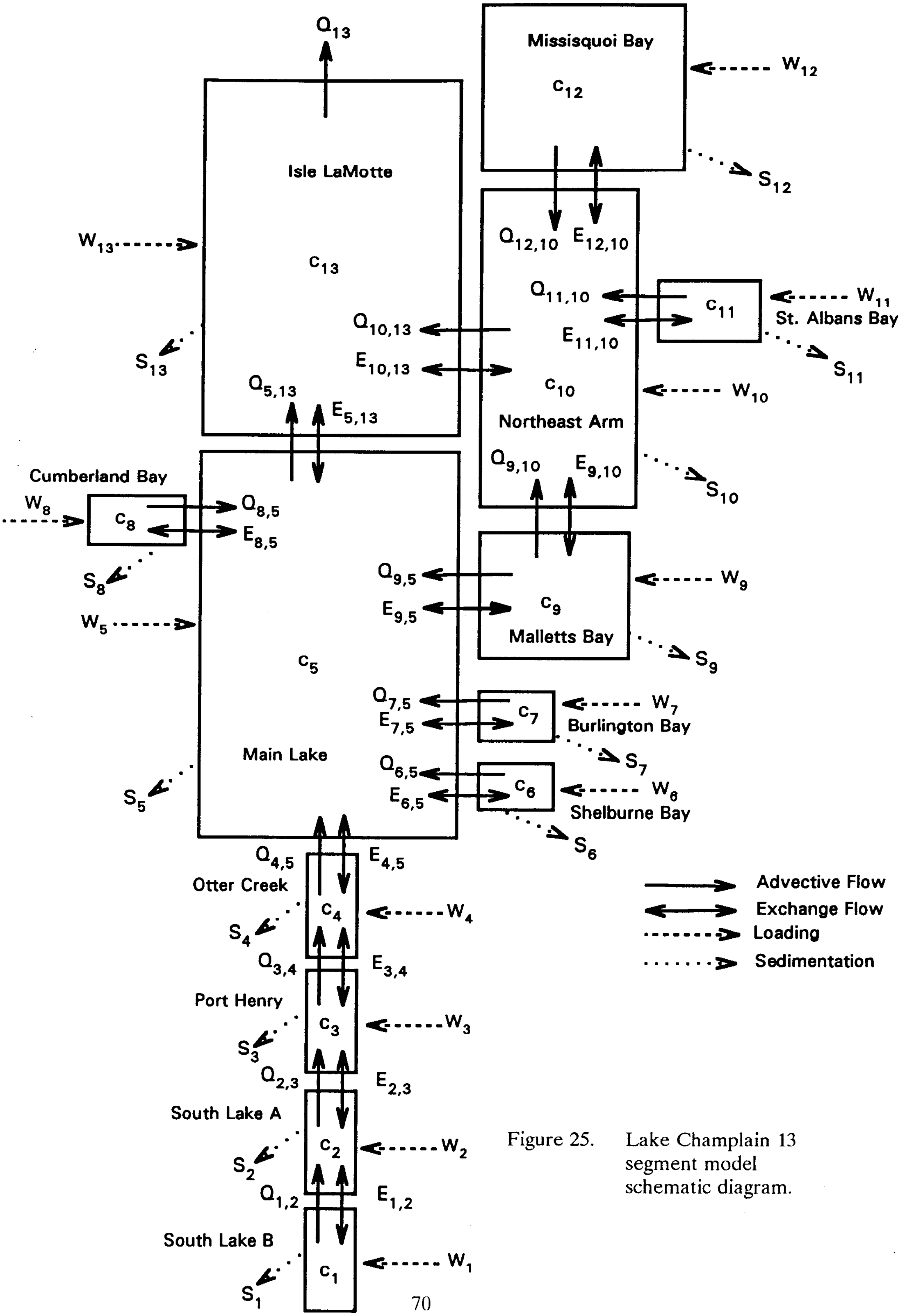

Figure 25. Lake Champlain 13 segment model schematic diagram.

Modeling analyses for this study were done using the computer program BATHTUB (Walker, 1987). BATHTUB was originally developed for modeling water quality in spatially segmented reservoirs and was later modified for use on Lake Champlain by Walker (1992). The BATHTUB program conducts steady-state mass balance calculations using the model terms given in equation 1, with several enhancements.

External loadings (W_i) are calculated in the BATHTUB program for each lake segment as the sum of all tributary inputs, wastewater discharges, and atmospheric loadings. The water balance terms (Q_{ij}, Q_{ji}), are calculated for each segment as the net sum of contributing tributary inflows, wastewater discharges, direct precipitation, and evaporation. Flows are routed through the segments to the lake outlet according to user-specified advective flow pathways (see Figure 25).

The modified BATHTUB program (Walker, 1992) evaluates the exchange flow terms (E_{ij}, equation 1) through a model calibration process using loading and in-lake concentration data for a conservative substance such as chloride ($S_i = 0$). The program solves for the exchange flows at each user-specified segment interface so that predicted chloride concentrations equal the observed mean values in each segment.

The exchange rate calibration is limited by the mathematical constraint that there be no more than one exchange boundary for each model segment. However the updated BATHTUB program (Walker, 1992) provides for additional "channels" between lake segments in which advective and diffusive exchange flows can be directly specified in order to achieve a more fully two-dimensional model segment structure. The segmentation requirements for Lake Champlain (Figure 25) were such that an additional channel was necessary only for the Malletts Bay segment. All other segments were represented as a linear branching network with only one exchange boundary per segment.

The BATHTUB program provides several alternative empirically derived sub-models for estimating the phosphorus sedimentation terms (S_i), as documented in Walker (1987). Phosphorus sedimentation can be represented in the program as either a first or second order settling process, as a function of either lake water column total phosphorus or inflow "available" phosphorus concentrations. The same sedimentation coefficients can be applied to all lake segments, or phosphorus sedimentation rates can be calibrated to individual lake segments through BATHTUB program procedures in order to provide a better fit between predicted and observed lake total phosphorus concentrations.

Error analysis is provided by the BATHTUB program through specification of error terms for all input data values and for any sub-models employed, such as phosphorus sedimentation sub-models. Error terms for input data such as flows and loadings are specified as the coefficient of variation (C.V.) of the mean (i.e. the standard error divided by the mean), with the assumption that the errors are lognormally distributed using a natual log scale. The BATHTUB program computes error statistics for all model predicted variables using a first-order error analysis. Approximate 95% confidence limits are calculated according to equation 2 (Walker, 1987).

$$Y_m \exp(-2CV) < Y < Y_m \exp(2CV) \qquad (2)$$

where

Y_m = predicted mean value (arithmetic scale)
CV = error mean coefficient of variation
Y = 95% confidence range for the mean value

Lake Segmentation and Flow Routing

The lake segmentation scheme used in developing the model employed the same 13 lake segments for which in-lake total phosphorus criteria have been established (see Figure 1). Precise definitions of segment boundaries including geographic landmarks were given in the Vermont Water Quality Standards (Vermont Water Resources Board, 1991) and extended to include New York and Quebec waters through the 1993 Lake Champlain Water Quality Agreement (Lake Champlain Phosphorus Management Task Force, 1993).

The surface area, volume, and mean depth of each lake segment are given in Table 20. Bathymetric data contained in National Oceanic and Atmospheric Administration (N.O.A.A.) 1:40,000 scale navigation charts for Lake Champlain were completely digitized, and the segment morphometric values given in Table 20 were computed by the New York State Department of Environmental Conservation Geographic Information System. The calculations were based on the lake surface datum for the charts (28.35 meters, N.G.V.D.), corresponding to "low lake level". Table 20 also lists the lake sampling stations contained in each lake segment.

Flows were routed through the lake segments in the model according to the advective flow pathways specified in Figure 25. The flow paths shown in Figure 25 were based on the general circulation pattern for Lake Champlain described in Henson and Gruendling (1977). Exchange flow interfaces were established by the BATHTUB program at each segment boundary where advective flows were indicated.

Model Input Data

The BATHTUB program application to Lake Champlain required input data on segment morphometry, in-lake segment mean concentrations of chloride and total phosphorus, and mean flows and loadings of chloride, total phosphorus, and dissolved phosphorus, grouped by lake segment. Segment morphometric data are given in Table 20. Other model input data used in the calibration procedures were derived from the sampling results, as described below.

Lake Chloride and Phosphorus Concentrations

Segment mean concentrations of chloride and total phosphorus for the two year study period are given in Table 21, based on the distribution of sampling stations used to represent each lake segment indicated in Table 20. Chloride and total phosphorus concentrations were averaged for all lake stations within each segment for each 14-day sampling interval during 1990-1991, and the segment means and coefficients of variation across all sampling intervals were computed. Statistical data reduction for this purpose was assisted by the computer program CRUNCH (W. W. Walker, pers. comm., 12/4/92).

Lake station #4, located approximately 3 km north of the International Paper Co. outfall, was strongly influenced by the high chloride concentration of this discharge (see Figure 4) and may not have been representative of segment average conditions. Chloride and phosphorus data from station #4 were therefore excluded from the segment concentration statistics given in Table 21 in order to avoid a possible bias in the estimate of the whole-segment mean chloride and total phosphorus concentration for the South Lake A segment.

Table 20. Lake model segment morphometric data, and listing of lake sampling stations within each segment.

Segment	Area (km^2)	Volume (km^3)	Mean Depth (m)	Length (km)	Lake Sampling Stations
1. South Lake B	5.79	.0078	1.35	20.1	1 2
2. South Lake A	43.27	.125	2.89	33.5	3 5
3. Port Henry	75.55	1.463	19.36	20.1	6 7
4. Otter Creek	28.49	.955	33.52	10.1	8 9
5. Main Lake	414.14	16.787	40.53	47.0	10-15, 17-20, 22 23 26 27 28 31
6. Shelburne Bay	9.62	.140	14.55	3.4	16
7. Burlington Bay	5.51	.063	11.43	2.0	21
8. Cumberland Bay	10.75	.063	5.86	3.4	33
9. Malletts Bay	55.06	.722	13.11	6.7	24 25
10. Northeast Arm	248.25	3.380	13.62	33.5	29 30 34 35 37 38 39 43 45
11. St. Albans Bay	7.21	.023	3.19	3.4	40 41
12. Missisquoi Bay	89.94	.205	2.28	16.8	47 48 50 51 52
13. Isle LaMotte	185.59	1.892	10.19	40.3	32 36 42 44 46 49
TOTAL	1179.17	25.826	21.90		

Table 21. **Mean chloride and total phosphorus concentrations in each lake segment, 1990-1991. C.V. = coefficient of variation of the mean.**

	Chloride (mg/l)		Total Phosphorus (Fg/l)	
Segment	**Mean**	**C.V.**	**Mean**	**C.V.**
South Lake B	11.62	.042	57.55	.067
South Lake A	13.47	.045	33.88	.062
Port Henry	11.18	.007	14.97	.048
Otter Creek	10.72	.005	14.58	.050
Main Lake	10.61	.004	11.79	.025
Shelburne Bay	10.89	.008	15.09	.058
Burlington Bay	10.78	.006	13.34	.068
Cumberland Bay	10.18	.012	13.57	.067
Malletts Bay	9.43	.012	9.35	.059
Northeast Arm	9.29	.004	14.23	.024
St. Albans Bay	10.20	.010	23.71	.052
Missisquoi Bay	7.78	.019	35.24	.056
Isle LaMotte	10.33	.006	12.10	.027

Two Year (1990-1992) Flow and Loading Data Set

Mean flows and loadings from tributaries, ungaged drainage areas, and direct wastewater treatment facility discharges for the two year period of March 1990 to February 1992 were obtained from Tables 12, 13, and 14 and grouped by lake segment as indicated in Table 22. Dissolved phosphorus concentrations in the wastewater treatment plant effluents were not measured during this study, but were assumed in Table 22 to be equal to the effluent total phosphorus concentrations for the purposes of the BATHTUB program calculations. Other BATHTUB program lakewide input data such as atmospheric deposition rates, precipitation amounts, evaporation rates, water withdrawal rates, and lake level changes were obtained from the results presented in Table 16 and elsewhere in the Sampling Results section of this report, and are summarized in Table 23.

Hydrologic Base Year Flow and Loading Data Set

Tributary flows and loadings change from year to year as a result of natural variability. When using a model to develop phosphorus load reduction strategies, it is desirable to normalize the flows and loadings measured during a particular monitoring period to better represent long-term average conditions, as was done for phosphorus management purposes in the Great Lakes (Thomas *et al.*, 1980). For this reason, a second set of model input data were developed in order to represent a hydrologic "base year".

The selection of a hydrologic base year for Lake Champlain was based on an examination of long term stream flow gage records for ten tributaries in the basin. Tributary flows measured during the March 1990 to February 1992 monitoring period are compared with the distribution of long term annual mean values in Figure 26. As shown in Figure 26, the 1990-1992 mean flows were consistently higher than the long term median annual flows. This was the result of unusually high flows existing during 1990. However, the mean flows for (calendar year) 1991 were generally similar to the long term median annual values, and fell within the inter-quartile ranges of the long term annual means in most cases. Calendar year 1991 was therefore selected as a good hydrologic base year for use in establishing "current average" loading conditions.

Tributary flows and loadings calculated for the 1991 hydrologic base year are given in Table 24. Chloride and total phosphorus loads given in Table 24 were recalculated for 1991 using the FLUX program methods previously described. Concentration *vs.* flow regressions for each stream based on the entire March 1990 to April 1992 sampling period were applied to the average daily flow records for 1991 to produce estimates of the mean flows and loadings of chloride, total phosphorus, and dissolved phosphorus during 1991. Flows and loadings from ungaged areas and from LaChute Creek given in Table 24 were estimated from the 1991 measurements in the same manner that estimates were made for the two year 1990-1992 sampling period (see Table 13). Flows and loadings from wastewater treatment facilities, and water withdrawal rates from intakes in the lake, were assumed to remain constant at their 1990-1991 mean values. Lakewide data input values for 1991 such as atmospheric deposition rates, evaporation rates, and lake level changes were derived from 1991 monitoring data, and are summarized in Table 23.

The full two year, 1990-1992 data set (Table 22) was used for most model calibration procedures, as discussed below. The 1991 hydrologic base year data set (Table 24) was used to best describe "current average" phosphorus loading conditions, and to assess the water quality response of the lake to long-term average tributary phosphorus loadings for load reduction strategy purposes, as discussed in a later section of this report.

Table 22. Model input data for gaged tributaries, ungaged areas, and direct wastewater treatment facility (WWTF) discharges, for the period of March 1990 to February 1992.

	Flow (hm^3/yr)		Chloride (mg/l)		Tot. P (Fg/l)		Diss. P (Fg/l)	
Segment/Source	Mean	C.V.	Mean	C.V.	Mean	C.V.	Mean	C.V.
SOUTH LAKE B								
Mt. Hope	19	.05	0.8	.051	8	.163	4	.069
Mettawee/Barge Canal	621	.05	11.6	.021	79	.076	28	.081
Poultney	371	.05	10.5	.019	87	.111	22	.117
Ungaged E	16	.20	11.1	.200	80	.200	24	.200
Ungaged W	62	.20	0.8	.200	8	.200	4	.200
Whitehall WWTF	.74	.05	55.0	.084	960	.170	960	.170
SOUTH LAKE A								
Mill (Putnam Sta.)	14	.05	7.3	.076	53	.204	23	.193
East	35	.20	9.2	.045	52	.057	30	.050
LaChute	365	.20	8.4	.010	4	.061	2	.086
Putnam	95	.05	5.5	.039	24	.227	6	.083
Ungaged E	33	.20	9.1	.200	55	.200	34	.200
Ungaged W	48	.20	5.6	.200	23	.200	8	.200
Ticonderoga WWTF	1.3	.05	96.0	.244	1120	.170	1120	.170
International Paper Co. WWTF	24	.05	415.0	.046	340	.170	340	.170
PORT HENRY								
Mill (Port Henry)	37	.05	9.7	.031	35	.105	7	.101
Hoisington	13	.05	9.5	.043	67	.182	20	.169
Ungaged E	10	.20	9.1	.200	55	.200	34	.200
Ungaged W	66	.20	9.6	.200	43	.200	9	.200
Port Henry WWTF	.81	.05	56.0	.137	1920	.110	1920	.110
Westport WWTF	.12	.05	78.0	.068	1810	.130	1810	.130
OTTER CREEK								
Otter	1427	.05	9.7	.016	99	.062	61	.071
Little Otter	80	.05	11.0	.031	122	.065	75	.075
Lewis	116	.05	7.4	.018	89	.108	26	.171
Ungaged E	12	.20	9.5	.200	98	.200	58	.200
Ungaged W	2.3	.20	9.5	.200	62	.200	18	.200
Vergennes WWTF	.65	.05	70.0	.068	700	.200	700	.200
MAIN LAKE								
Bouquet	361	.05	9.3	.028	60	.151	11	.189
Highlands Forge	12	.05	7.7	.027	12	.086	5	.068
Winooski	2003	.05	12.3	.029	77	.083	12	.060
Ausable	804	.05	5.8	.030	30	.140	7	.092
Little Ausable	84	.05	9.5	.061	55	.126	24	.078
Salmon	62	.05	5.8	.048	32	.121	12	.163
Ungaged E	18	.20	12.3	.200	74	.200	12	.200
Ungaged W	55	.20	6.9	.200	34	.200	9	.200
Burlington North WWTF	1.8	.05	71.0	.054	1910	.100	1910	.100
SHELBURNE BAY								
LaPlatte	58	.05	27.0	.040	253	.064	190	.089
Ungaged E	16	.20	27.0	.200	254	.200	190	.200
Shelburne F.D. #2 WWTF	.38	.05	119.0	.051	700	.200	700	.220
Shelburne F.D. #1 WWTF	.42	.05	101.0	.042	670	.160	670	.160
So. Burlington Bart. Bay WWTF	1.0	.05	118.0	.076	580	.180	580	.180

	Flow (hm^3/yr)		Chloride (mg/l)		Tot. P (Fg/l)		Diss. P (Fg/l)	
Segment/Source	Mean	C.V.	Mean	C.V.	Mean	C.V.	Mean	C.V.
BURLINGTON BAY								
Ungaged E	1.3	.20	27.0	.200	254	.200	190	.200
Burlington Main WWTF	5.4	.05	104.0	.078	2120	.120	2120	.120
CUMBERLAND BAY								
Saranac	877	.05	5.4	.029	22	.062	8	.056
Ungaged W	58	.20	5.2	.200	21	.200	8	.200
Plattsburgh City WWTF	11.5	.05	76.0	.061	1740	.150	1740	.150
Plattsburgh/Champlain Park WWTF	.18	.05	150.0	.157	1620	.200	1620	.200
MALLETTS BAY								
Indian	15	.20	43.2	.111	81	.087	17	.038
Malletts	40	.20	14.9	.024	63	.086	23	.066
Lamoille	1423	.05	8.8	.032	33	.079	11	.073
Ungaged E	34	.20	9.3	.200	35	.200	11	.200
NORTHEAST ARM								
Stone Bridge	13	.05	17.6	.096	86	.098	34	.131
Ungaged E	40	.20	17.5	.200	83	.200	34	.200
ST. ALBANS BAY								
Mill	39	.20	25.9	.032	135	.103	62	.114
Stevens	16	.20	61.4	.042	238	.139	101	.144
Ungaged E	2.8	.20	35.2	.200	155	.200	71	.200
St. Albans City WWTF	2.9	.05	78.0	.084	270	.140	270	.140
Northwest Correctional WWTF	.02	.05	56.0	.050	170	.280	170	.280
MISSISQUOI BAY								
Missisquoi	1534	.05	6.5	.024	72	.072	18	.066
Rock	85	.20	9.8	.036	353	.186	100	.111
Pike	347	.05	11.9	.022	173	.120	68	.112
Ungaged E	43	.20	7.6	.200	105	.200	31	.200
Swanton WWTF	1.0	.05	122.0	.082	2380	.340	2380	.340
ISLE LAMOTTE								
Little Chazy	56	.05	14.3	.036	74	.162	54	.249
Great Chazy	366	.05	10.1	.032	52	.154	19	.104
Ungaged W	34	.20	10.5	.200	50	.200	24	.200
Wyeth-Ayerst, Chazy WWTF	.054	.05	466.0	.500	83800	.280	83800	.280

Table 23. Lakewide input data values for BATHTUB model. Coefficients of variation are given in parentheses.

Model Term	Two Year Sampling Period (Mar 1990 - Feb 1992)	Hydrologic Base Year (Jan 1991 - Dec 1991)
Precipitation Rate (m/yr)	0.96 (.04)	0.81 (.04)
Chloride Atmospheric Deposition Rate (mg/m^2-yr)	276 (.03)	233 (.03)
Phosphorus Atmospheric Deposition Rate (mg/m^2-yr)	16.1 (.25)	13.6 (.25)
Evaporation Rate (m/yr)	0.65 (.20)	0.70 (.20)
Lake Level Change (m/yr)	-0.31 (.00)	-1.15 (.00)

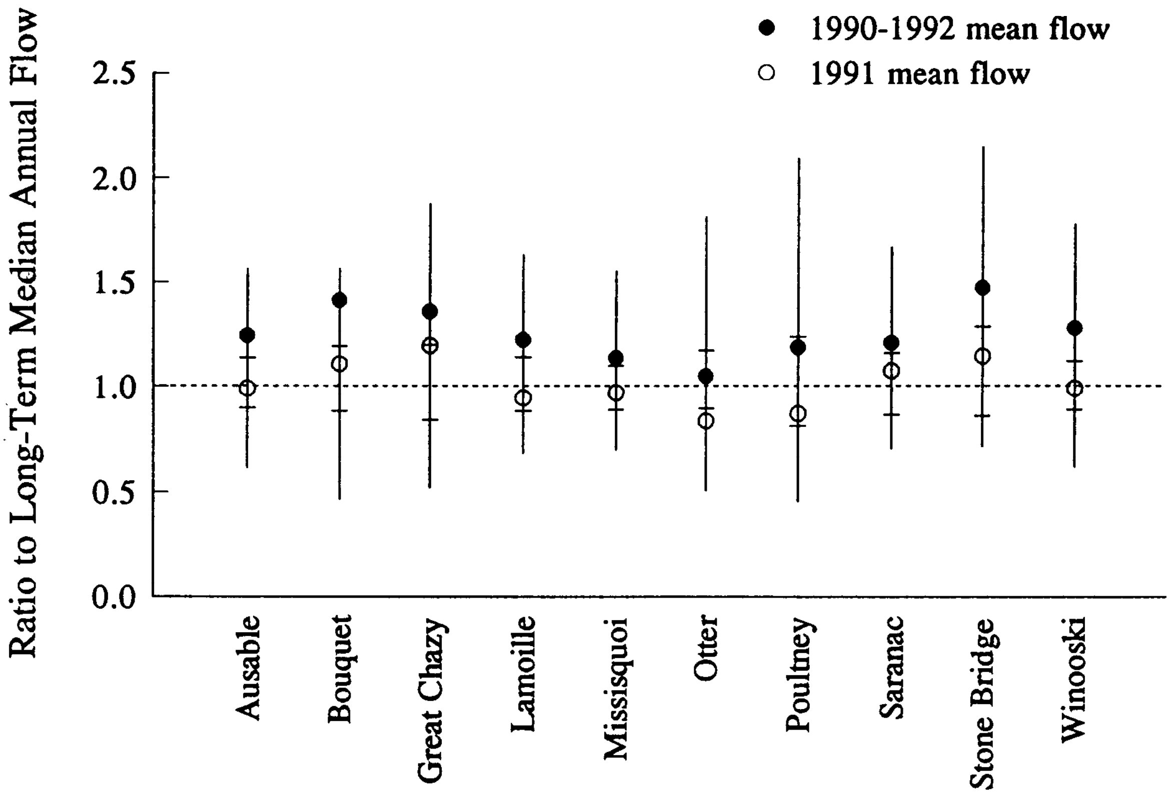

Figure 26. Comparison of annual mean stream flows measured during the project period with long-term annual mean flows recorded at ten gage stations in the Lake Champlain drainage basin. Mean flows for March 1990 to February 1992, and for calendar year 1991 only, are plotted as a ratio to the median of the long-term mean flows at each station. Error bars indicate the inter-quartile range and the extreme range of the mean annual flows for the long-term period of record at each gage station. The gage stations are as indicated in Table 4, except the Missisquoi River data were obtained from the gage in East Berkshire, Vermont.

Table 24. Model input data for gaged tributaries, ungaged areas, and direct wastewater treatment facility (WWTF) discharges, for the hydrologic base year of January 1991 through December 1991.

	Flow (hm^3/yr)		Chloride (mg/l)		Tot. P (Fg/l)		Diss. P (Fg/l)	
Segment/Source	Mean	C.V.	Mean	C.V.	Mean	C.V.	Mean	C.V.
SOUTH LAKE B								
Mt. Hope	13	.05	.8	.046	7	.141	4	.065
Mettawee/Barge Canal	487	.05	12.1	.020	76	.072	24	.075
Poultney	273	.05	11.3	.019	62	.112	20	.107
Ungaged E	12	.20	11.7	.200	71	.200	23	.200
Ungaged W	44	.20	.8	.200	8	.200	4	.200
Whitehall WWTF	.74	.05	55	.084	960	.170	960	.170
SOUTH LAKE A								
Mill (Putnam Sta.)	9.0	.05	7.7	.071	47	.197	24	.169
East	24	.20	9.3	.040	53	.053	30	.045
LaChute	273	.20	8.4	.010	4	.061	2	.086
Putnam	67	.05	6.3	.034	19	.108	6	.072
Ungaged E	23	.20	9.3	.200	50	.200	33	.200
Ungaged W	33	.20	6.5	.200	20	.200	8	.200
Ticonderoga WWTF	1.3	.05	96.0	.244	1120	.170	1120	.170
International Paper Co. WWTF	24	.05	415.0	.046	340	.170	340	.170
PORT HENRY								
Mill (Port Henry)	25	.05	10.9	.031	25	.135	7	.148
Hoisington	9.6	.05	10.2	.048	50	.162	19	.162
Ungaged E	7.1	.20	9.3	.200	54	.200	33	.200
Ungaged W	46	.20	10.7	.200	32	.200	12	.200
Port Henry WWTF	.81	.05	56.0	.137	1920	.110	1920	.110
Westport WWTF	.12	.05	78.0	.068	1810	.130	1810	.130
OTTER CREEK								
Otter	1119	.05	10.1	.016	98	.057	62	.065
Little Otter	55	.05	11.4	.030	99	.058	64	.066
Lewis	90	.05	7.5	.016	58	.070	20	.073
Ungaged E	9.3	.20	10.0	.200	95	.200	59	.200
Ungaged W	1.7	.20	10.2	.200	52	.200	21	.200
Vergennes WWTF	.65	.05	70.0	.068	700	.200	700	.200
MAIN LAKE								
Bouquet	281	.05	9.9	.025	48	.150	10	.178
Highlands Forge	8.8	.05	7.8	.026	12	.088	4	.055
Winooski	1543	.05	13.2	.032	54	.069	12	.057
Ausable	639	.05	6.3	.029	26	.124	8	.083
Little Ausable	89	.05	9.2	.072	58	.145	23	.087
Salmon	55	.05	5.8	.046	32	.124	12	.160
Ungaged E	14	.20	13.2	.200	54	.200	12	.200
Ungaged W	45	.20	7.5	.200	35	.200	10	.200
Burlington North WWTF	1.8	.05	71.0	.054	1910	.100	1910	.100
SHELBURNE BAY								
LaPlatte	44	.05	30.4	.042	270	.062	209	.067
Ungaged E	13	.20	30.4	.200	270	.200	209	.200
Shelburne F.D. #2 WWTF	.38	.05	119.0	.051	700	.220	700	.220
Shelburne F.D. #1 WWTF	.42	.05	101.0	.042	670	.160	670	.160

	Flow (hm^3/yr)		Chloride (mg/l)		Tot. P (Fg/l)		Diss. P (Fg/l)	
Segment/Source	**Mean**	**C.V.**	**Mean**	**C.V.**	**Mean**	**C.V.**	**Mean**	**C.V.**
So. Burlington Bart. Bay WWTF	1.0	.05	118.0	.076	580	.180	580	.180
BURLINGTON BAY								
Ungaged E	1.0	.20	30.4	.200	270	.200	209	.200
Burlington Main WWTF	5.4	.05	104.0	.078	2120	.120	2120	.120
CUMBERLAND BAY								
Saranac	776	.05	5.5	.029	21	.053	8	.055
Ungaged W	51	.20	5.5	.200	21	.200	8	.200
Plattsburgh City WWTF	11.5	.05	76.0	.061	1740	.150	1740	.150
Plattsburgh/Champlain Park WWTF	.18	.05	150.0	.157	1620	.200	1620	.200
MALLETTS BAY								
Indian	13	.20	44.5	.110	68	.070	17	.037
Malletts	31	.20	15.5	.024	54	.066	22	.056
Lamoille	1100	.05	9.4	.032	27	.061	10	.064
Ungaged E	26	.20	10.0	.200	28	.200	10	.200
NORTHEAST ARM								
Stone Bridge	10	.05	19.0	.106	76	.094	33	.139
Ungaged E	31	.20	19.0	.200	77	.200	29	.200
ST. ALBANS BAY								
Mill	26	.20	26.4	.064	131	.093	60	.114
Stevens	14	.20	61.5	.040	257	.142	102	.145
Ungaged E	2.1	.20	38.9	.200	176	.200	714	.200
St. Albans City WWTF	2.9	.05	78.0	.084	270	.140	270	.140
Northwest Correctional WWTF	.02	.05	56.0	.050	170	.280	170	.280
MISSISQUOI BAY								
Missisquoi	1307	.05	6.6	.024	63	.061	18	.064
Rock	69	.20	9.5	.036	401	.211	100	.112
Pike	296	.05	12.0	.021	169	.116	69	.111
Ungaged E	36	.20	7.7	.200	96	.200	30	.200
Swanton WWTF	1.0	.05	122.0	.082	2380	.340	2380	.340
ISLE LAMOTTE								
Little Chazy	44	.05	14.4	.038	72	.157	54	.236
Great Chazy	320	.05	10.0	.032	54	.147	20	.106
Ungaged W	29	.20	10.5	.200	56	.200	24	.200
Wyeth-Ayerst, Chazy WWTF	.054	.05	466.0	.500	83800	.280	83800	.280

Exchange Rate Calibration

The exchange rate calibration was conducted with two general goals in mind. The first goal was to derive exchange rates that resulted in an exact fit between the observed (from Table 21) and predicted chloride concentrations in each lake segment. A second goal was to obtain exchange flow values that were physically realistic in relation to the morphometry of the lake and the expected hydrodynamic characteristics of each area.

These goals placed a conservation of mass constraint on the calibration procedure necessitating that the lakewide flow-weighted mean inflow chloride concentration be equal to the concentration in the lake outflow segment (Isle LaMotte segment, Figure 1). Data summarized in Table 19 indicated that the lakewide mean inflow chloride concentration was 10.09 mg/l (adjusted for evaporation), slightly lower than the measured value of 10.33 for the Isle LaMotte segment (Table 21). Although the overall chloride input-output budget for the lake (Table 19) appeared to be accurate to within 3%, and the difference could have been accounted for simply by data variability, the slight inflow *vs.* outflow concentration difference was enough to disrupt the calibration procedure. An additional chloride input of about 3,500 mt/yr (3% of the total measured chloride loading) was necessary to balance the inflow and outflow chloride concentrations and attain an exact exchange rate calibration by the BATHTUB program.

One chloride source that was not measured during this study was runoff of road de-icing salts from urban areas not incorporated into the tributary monitoring network. The Cities of Burlington, Vermont and Plattsburgh, New York applied an average of 1,203 and 1,125 metric tons of chloride, respectively, from road salt each winter to their streets during 1990-1992, based on information provided by the Public Works Departments in the two cities. Most of the road salt applied to city streets would have run off into the lake unmonitored by the sampling program. These values do not include salt applied to other urban roadways adjacent to the Cities of Burlington and Plattsburgh.

When the total unmonitored chloride loading potential from road salt is considered, this source could well have accounted for the additional 3,500 mt/yr input required to balance the inflow and outflow concentrations. Most of the transient winter and spring chloride load from urban road salt would have been rapidly mixed into the Main Lake region and reflected in the data obtained there. An additional chloride load of 3,500 mt/yr was therefore added in the model directly to the Main Lake segment for model calibration purposes.

The model segmentation scheme employed for this study (see Figure 25) included two advective and exchange flow boundaries for the Malletts Bay segment. As described earlier, the BATHTUB program provided for the direct specification of additional flow "channels" to accommodate this situation. An advective and exchange flow channel was established in the BATHTUB program at the northern boundary of Malletts Bay in order to model transport of chloride and phosphorus between Malletts Bay and the Northeast Arm. Advective and exchange flows at this segment boundary were directly specified in the program based on available data, as described below. Exchange flows at the western boundary of Malletts Bay with the Main Lake were evaluated through the model calibration process, as was done for all other lake segments.

Myer (1977) measured flow rates through the openings between Malletts Bay and the adjacent lake segments and found that the direction of flow through these openings was variable in response to wind direction, indicating that significant exchange flows occurred at these boundaries. Data given in Myer and Gruendling (1979) indicated that flow rates (in either direction) observed at the northern outlet of Malletts Bay to the Northeast Arm averaged about 19% of the flows measured at the western

openings to the Main Lake, based on 114 hours of observations during the course of three summers. Accordingly, advective flows leaving Malletts Bay through the northern channel were specified in the BATHTUB program to be 242 hm^3/yr out of the total 1512 hm^3/yr tributary inflow to Malletts Bay (from Table 22). The remaining 1,270 hm^3/yr of tributary inflow was routed through the western opening of Malletts Bay to the Main Lake. Exchange flows specified at the northern boundary were adjusted during the exchange rate calibration procedure so that the rates at the northern opening were 19% of the final calibrated exchange value for the western boundary.

The BATHTUB program was used to solve for the exchange flow terms at each segment boundary using the chloride model input data given in Tables 22 and 23 for the two year period of March 1990 to February 1992. The exchange rate calibration results are shown in Table 25.

Theoretical considerations and empirical findings reviewed by Chapra and Reckhow (1983) indicate that the lake model bulk exchange flow terms (E) specified in equation 1 should be a function of a scale-dependent diffusion coefficient and the segment morphometry, as indicated by equation 3.

$$E = D\,A_c / L_m \tag{3}$$

Where

E = exchange flow between adjacent segments (m^3/yr)
D = turbulent diffusion coefficient at the interface between segments (m^2/yr)
A_c = cross-sectional area of the exchange interface between segments (m^2)
L_m = effective mixing length of the interface between segments (m)

Values for the diffusion coefficients and the effective mixing lengths between segments would be difficult to evaluate for Lake Champlain with its non-uniform segment morphometry and its artificially narrow openings between segments in some cases. However, the cross-sectional area of the exchange interfaces (A_c) were readily estimated from lake morphometric information provided on 1:40,000 scale N.O.A.A. navigation charts for Lake Champlain, and these interface areas are given in Table 25. The calibrated exchange rates are plotted against the A_c values in Figure 27.

Figure 27 shows that an internally consistent, positive relationship existed between the calibrated exchange values and the segment interface areas, as would be predicted from equation 2 and expected from common intuition. The fact that the exchange flow rates calibrated from a chloride mass balance model were strongly related to the interface areas suggests that the exchange values are physically realistic estimates of actual diffusive mixing between segments in Lake Champlain. The calibrated exchange rates listed in Table 25 were used in all subsequent phosphorus modeling analyses for this study.

Phosphorus Sedimentation Calibration

The phosphorus net sedimentation terms in equation 1 (S_i, mt/yr) were evaluated through a model calibration procedure using methods provided by the BATHTUB program (Walker, 1987, 1992). The general goal of the phosphorus sedimentation rate calibration was to produce a reasonably accurate fit between predicted and observed phosphorus concentrations in each lake segment, with a minimum departure from existing broadly based empirical phosphorus sedimentation models established in the literature.

Table 25. Exchange rate calibration results. Exchange flows and interface areas are given for the "downstream" boundary of each lake segment, based on the flow routing scheme shown in Figure 25.

Lake Segment	Calibrated Exchange Rate (hm^3/yr)	Observed Chloride (mg/l)	Predicted Chloride (mg/l)	Area of Exchange Interface (m^2)
South Lake B	712	11.62	11.62	855
South Lake A	1259	13.47	13.47	2573
Port Henry	13998	11.18	11.18	67312
Otter Creek	49427	10.72	10.72	98709
Main Lake	8861	10.61	10.61	98207
Shelburne Bay	4816	10.89	10.89	23418
Burlington Bay	2986	10.78	10.78	89645
Cumberland Bay	8672	10.18	10.18	44582
Malletts Bay	272 (52[1])	9.43	9.43	420 (27[1])
Northeast Arm	1968	9.29	9.29	570
St. Albans Bay	1844	10.20	10.20	4923
Missisquoi Bay	297	7.78	7.78	682
Isle LaMotte	--	10.33	10.33	--

[1] Value for the northern interface of Malletts Bay with the Northeast Arm.

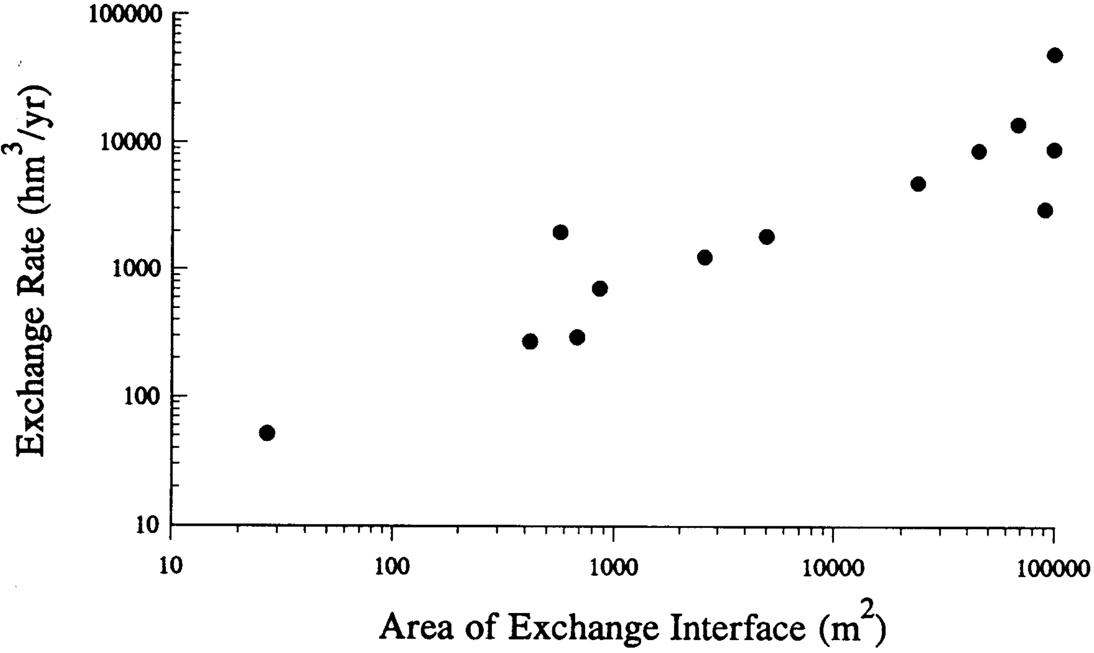

Figure 27. Calibrated exchange flow rates plotted against the cross-sectional area of the exchange interfaces.

The BATHTUB program supports a choice from among several alternative phosphorus sedimentation models. The five models chosen as being most appropriate for evaluation for Lake Champlain are listed in Table 26. All five models represent phosphorus sedimentation as a function of the water column total phosphorus concentration in the lake segment. Models 1 and 2 use first order settling velocity and "decay" coefficients, respectively. Model 3 represents net phosphorus sedimentation as a simple second order decay process.

The last two models modify the second order decay process so that phosphorus sedimentation rates are a function of the partitioning of the inflow phosphorus concentration between dissolved and particulate forms. These two models allow for the physically realistic possibility that lake segments receiving a higher proportion of "available" phosphorus loading would have lower net sedimentation rates. Model 4 involves the calibration of a second order sedimentation coefficient to modified loading and lake mass balance data in which the inflow concentrations are modified to be a weighted sum of dissolved and particulate phosphorus fractions, with the dissolved fractions weighted more heavily. Model 5 uses total phosphorus inflow concentrations, but adjusts the sedimentation rate as a function of the ratio of dissolved to total phosphorus loading. The inflow phosphorus partitioning functions used in these models were empirically derived by Walker (1987) from a U.S. Army Corps of Engineers reservoir data set using inflow ortho-phosphorus concentrations, but were applied to Lake Champlain for model evaluation purposes using dissolved phosphorus loading data.

The five alternative phosphorus sedimentation models were initially evaluated by running a calibration procedure within the BATHTUB program using the two year (1990-1992) phosphorus model input data given in Tables 22 and 23 with the calibrated exchange terms from Table 25. For each of the five sedimentation models, a "global" calibration was conducted in which an optimum least-squares fit of a single sedimentation coefficient (i.e. k values, Table 26) across all Lake Champlain segments was obtained. The relative performance of each sedimentation model was evaluated based on the R^2 statistic for the regression of log predicted *vs.* log observed segment phosphorus concentrations following the global calibration, and on the number of individual segments having significant differences between their predicted and observed phosphorus concentrations.

The predicted and observed segment phosphorus concentrations were compared for this purpose using a t-test procedure within the BATHTUB program which accounted for all phosphorus model input data variability (i.e. C.V. values, Tables 22, 23), but which did not consider uncertainty in the calibrated exchange rate values or the global sedimentation coefficients. For this reason, and because of the multiple comparison nature of the t-test procedure, the number of statistically significant differences indicated by the comparison of predicted *vs.* observed concentrations may be somewhat inflated.

The results of the phosphorus sedimentation model comparison using the global calibration procedure are shown in Table 27. The first order models (1 and 2) produced the poorest performance, with the lowest R^2 values and greatest number of significant differences between predicted and observed segment phosphorus concentrations. The superior performance of second order phosphorus sedimentation models is consistent with Walker's (1987) findings for a U.S. Army Corps of Engineers reservoir data set.

The second order phosphorus inflow partitioning models (4 and 5) produced some marginal improvement over the simple second order decay formulation (model 3). However, use of the phosphorus partitioning models for predictive purposes on Lake Champlain would require quantitative knowledge of how inflow dissolved and particulate phosphorus fractions will change in response to future management actions. This level of understanding of the link between watershed phosphorus

Table 26. Alternative phosphorus sedimentation models tested for Lake Champlain.

Model 1: First order settling velocity (Walker, 1987, model 7)

$$S = A\ k_1\ P$$

Model 2: First order decay (Walker, 1987, model 6)

$$S = V\ k_2\ P$$

Model 3: Second order decay, simple (Walker, 1987, model 3)

$$S = V\ k_3\ P^2$$

Model 4: Second order decay, available phosphorus modified loading (Walker, 1987, model 1)

$$S = V\ k_4\ [\ Qs/(Qs+13.3)\ ]\ P^2$$

Model 5: Second order decay, available phosphorus concentration function (Walker, 1987, model 2)

$$S = V\ k_5\ (Pi/Pio)\ [\ Qs/(Qs+13.3)\ ]\ P^2$$

Definitions:

S = segment phosphorus net sedimentation rate (mt/yr)
A = segment area (km^2)
V = segment volume (hm^3)
P = segment total phosphorus concentration (mg/l)
Pi = segment inflow total phosphorus concentration (mg/l)
Pio = segment inflow ortho-phosphorus concentration (mg/l)
Qs = segment surface overflow rate (m/yr)
k_1 = phosphorus net settling velocity (m/yr)
k_2 = phosphorus first order decay rate (yr^{-1})
$k_{3,4,5}$ = phosphorus second order decay rate (m^3/g-yr)

Table 27. Phosphorus sedimentation rate global calibration results. See Table 26 for model definitions. An asterisk (*) indicates a statistically significant difference between predicted and observed values at p<.05.

		Predicted Phosphorus Concentrations (Fg/l)				
Lake Segment	Observed Phosphorus Conc. (Fg/l)	Model 1	Model 2	Model 3	Model 4	Model 5
South Lake B	57.6	50.1	60.5	56.7	56.9	57.1
South Lake A	33.8	24.5*	36.8	31.9	32.3	32.5
Port Henry	15.0	16.8*	14.4	14.2	14.5	14.7
Otter Creek	14.6	18.2*	12.8*	13.6	13.9	14.1
Main Lake	11.8	16.3*	10.0*	11.2	11.5	11.6
Shelburne Bay	15.1	18.8*	13.0*	14.1	14.4	14.5
Burlington Bay	13.3	18.9*	13.2	14.4	14.6	14.8
Cumberland Bay	13.6	18.2*	13.0	14.1	14.4	14.5
Malletts Bay	9.4	14.6*	15.7*	15.1*	13.6*	13.2*
Northeast Arm	14.2	9.1*	17.2*	13.2*	13.0*	12.8*
St. Albans Bay	23.7	12.5*	21.5	17.6*	17.5*	17.4*
Missisquoi Bay	35.2	36.6	75.5*	51.1*	47.4*	44.5*
Isle LaMotte	12.1	12.3	10.5*	11.0*	11.2*	11.3
R^2 statistic[1]		.61	.83	.86	.90	.91
Calibrated k value[2]		38 m/yr	2.41 yr^{-1}	173 m^3/g-yr	205 m^3/g-yr	224 m^3/g-yr

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[1] For regression of log predicted *vs.* log observed phosphorus concentrations.

[2] Phosphorus sedimentation coefficients for each model, as defined in Table 26.

management and phosphorus loading to the lake is not likely to be available in the near future. For this reason, and because the simpler second order sedimentation function (model 3) produced nearly the same results in the global calibration (Table 27), the phosphorus partitioning models were not considered further for use in this study.

The results shown in Table 27 for the global calibration procedures indicated that the simple second order decay function (model 3) was the most appropriate model for Lake Champlain. However, application of a single second order sedimentation coefficient throughout Lake Champlain resulted in significant differences between observed and predicted phosphorus concentrations in some lake segments.

The final phosphorus sedimentation calibration procedure for Lake Champlain used the simple second order settling model (model 3) and focused on those segments where the lakewide global coefficient produced inaccurate predictions. For all other lake segments, the second order sedimentation coefficient was returned to Walker's (1987) empirically derived value of 100 m^3/g-yr (C.V. = 0.45) so that there would be the least possible departure from the coefficients that were independently derived from the reservoir data set. The major differences between observed and predicted phosphorus concentrations listed in Table 27 for model 3 existed in St. Albans Bay, Malletts Bay, and Missisquoi Bay. Segment-specific sedimentation rate calibration adjustments were applied to these three segments only, as described below.

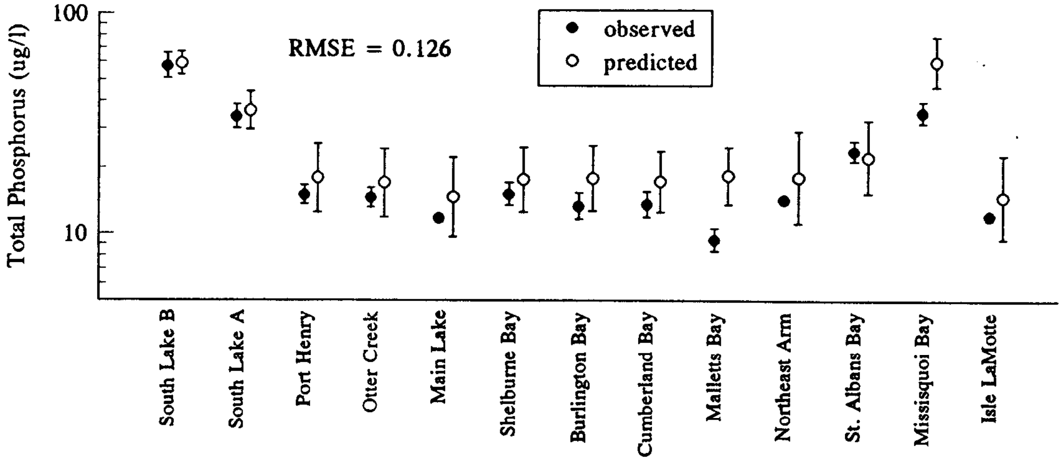
Phosphorus loadings to the St. Albans Bay segment were adjusted to account for the substantial internal phosphorus loading to the bay that is known to occur (Smeltzer, 1991). Mass balance calculations using the data set from the present study with the simple embayment model of Chapra (1979a) indicated that St. Albans Bay had a net internal phosphorus loading rate (i.e. a negative sedimentation rate) of 8.6 mt/yr, or 3.3 mg/m^2-day expressed on an areal basis, during the two year (1990-1992) monitoring period.

An internal phosphorus load of 3.3 mg/m^2-day and a second order phosphorus sedimentation coefficient of zero were specified for the St. Albans Bay segment in the BATHTUB program input data so that an accurate calibrated fit of predicted and observed phosphorus concentrations could be obtained for St. Albans Bay within the program constraint that sedimentation coefficients must be non-negative. This somewhat artificial adjustment was necessary to prevent a model under-prediction of phosphorus concentrations in St. Albans Bay that would have disrupted the predictions for adjacent lake segments. Prediction of the response of St. Albans Bay to future phosphorus loading changes will require a more detailed understanding of how the internal loading rates and bay phosphorus levels may be changing over time in response to previous external load reductions, incorporating the results of internal phosphorus loading studies conducted for St. Albans Bay by Martin *et al.* (1994).

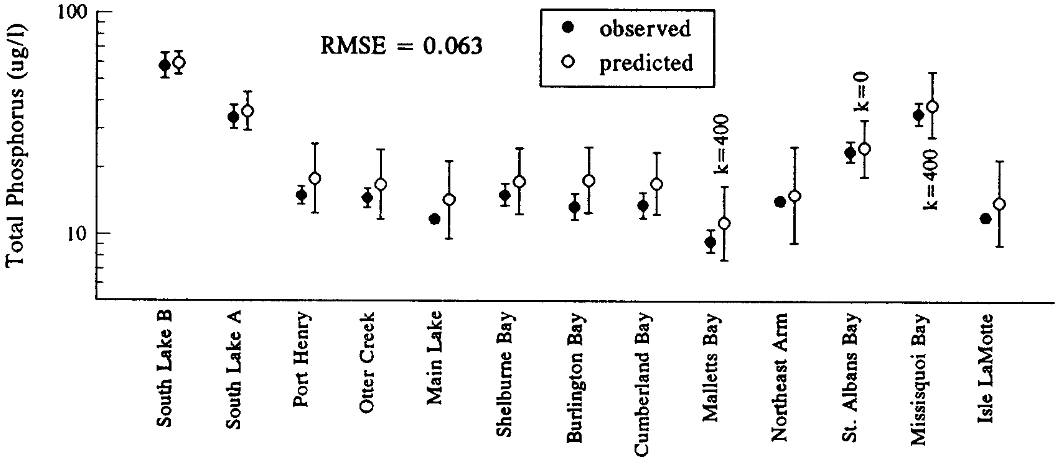
The globally calibrated models (Table 27) significantly over-predicted the observed phosphorus concentrations in Malletts Bay and Missisquoi Bay. Segment-specific calibration adjustments were applied to these two segments in which the second order phosphorus sedimentation coefficients were increased by a factor of four, to 400 m^3/g-yr.

The phosphorus sedimentation model calibration results are shown in Figure 28. The first plot (Figure 28A) compares observed and predicted phosphorus concentrations in each lake segment, using the two year (1990-1992) monitoring data given in Tables 22 and 23 without any calibration of the second order settling coefficients for Lake Champlain. Second order sedimentation rates of 100 m^3/g-yr were applied to all Lake Champlain segments in Figure 28A, as empirically derived by Walker (1987) from the U.S. Army Corps of Engineers reservoir data set. The error bars in Figure 28

A. No Calibration, 1990-1992 Loadings



B. Three Segment Calibration, 1990-1992 Loadings



C. Three Segment Calibration, 1991 Loadings

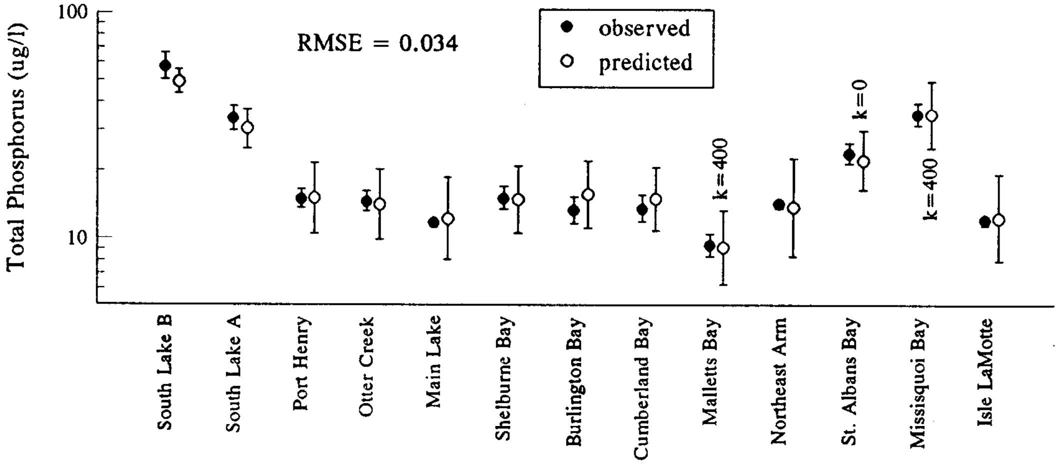


Figure 28. Phosphorus sedimentation model calibration results, comparing observed and model-predicted phosphorus concentrations in each lake segment. Error bars represent 95% confidence intervals for the observed and predicted mean values. Second order phosphorus sedimentation coefficients (k values) of 100 m^3/g-yr were used for all segments, except where indicated otherwise.

represent approximate 95% confidence intervals around the observed and predicted mean values. The error values for the predictions were calculated from the BATHTUB program first order error analysis procedure using both input data error (i.e. C.V. values from Tables 22 and 23) and phosphorus sedimentation model error, represented by a C.V. value of 0.45 for the second order settling terms (Walker, 1987).

Figure 28A shows that without any calibration, the fit between observed and predicted phosphorus concentrations was poor for some lake segments. The model generally over-predicted the observed phosphorus concentrations, although the differences were statistically significant in only a few segments. The root mean squared error (RMSE) calculated from the differences between observed and predicted concentrations ($\log_{10}$ scale) was 0.126 for the uncalibrated model.

Figure 28B shows the results for the same set of conditions except that the second order phosphorus sedimentation rates were adjusted for St. Albans Bay, Malletts Bay, and Missisquoi Bay as described above. The sedimentation rate calibration adjustment for these three segments eliminated all significant differences between observed and predicted phosphorus values. However, Figure 28B shows that some slight but systematic over-prediction of lake phosphorus concentrations remained in most segments when the two year (1990-1992) input data set was used. The calibration improved the RMSE value to 0.063.

Figure 28C compares observed and predicted lake phosphorus values calculated using the mean flows and phosphorus loading rates existing during the 1991 hydrologic base year (from Tables 23 and 24), with the same three-segment phosphorus sedimentation rate calibration used in Figure 28B. Figure 28C shows that a very close fit between observed and predicted values was obtained when tributary flow and loading rates more representative of long-term mean conditions were used in the model and compared with the 1990-1991 observed mean lake phosphorus concentrations. The systematic over-prediction was eliminated, and the RMSE was reduced to 0.034.

A likely explanation for the improved calibration fit to the 1991 hydrologic base year data is that the lake phosphorus concentrations measured during the two year study period reflected long-term mean flow and loading conditions more than the loading conditions existing during the specific monitoring period. Stream flows (and consequently phosphorus loadings and flow-weighted mean inflow concentrations for most tributaries) were somewhat higher during 1990 than in 1991 (see Figure 26). However, there were no statistically significant lakewide difference in total phosphorus concentrations between the two years, as discussed earlier.

Phosphorus response time calculations help explain why lake phosphorus concentrations did not immediately respond to the higher loadings during 1990. During the two year 1990-1992 monitoring period, Lake Champlain had a phosphorus loading rate of 884 mt/yr (Table 19) and a whole-lake mean phosphorus content of 328 mt (calculated from Tables 20 and 21), giving an average phosphorus residence time of 0.37 yrs. The lake phosphorus residence time model of Sonzogni *et al.* (1976) indicates that a period equal to three times the phosphorus residence time (about one year for Lake Champlain) would be required for a lake to establish an equilibrium with new steady-state loading conditions. This calculation assumes that no other factors such as sediment-water phosphorus interactions are operating to introduce further time lags into the system beyond what would be predicted by a simple wash-out model.

These phosphorus response time considerations indicate that the elevated phosphorus loading rates measured during 1990 would not have strongly influenced the mean lake phosphorus concentrations observed during 1990 and 1991. For this reason, it will be assumed that the 1991

hydrologic base year provides the best data set for use in calibrating the phosphorus sedimentation model because 1991 was more typical of long-term flow and loading conditions. The second order phosphorus sedimentation terms given in Figure 28C produced a very good fit between observed and predicted values, with calibration adjustments applied to only three segments.

The reasons for the unusually high apparent phosphorus sedimentation rates in Malletts Bay and Missisquoi Bay are unknown. Unique physical or chemical processes may be operating in these bays. Both bays are bounded by causeways with very narrow outlets, and are the most confined embayments in Lake Champlain. These morphologic characteristics may enhance the phosphorus trapping efficiency of these two bays.

The apparent phosphorus sedimentation rates estimated for Lake Champlain were compared with values derived from other lake data sets in order to gain additional perspective on the Lake Champlain values. Phosphorus net sedimentation rates in lakes are more commonly expressed as first order settling velocities. Data of DePalma *et al.* (1979) cited in Reckhow and Chapra (1983) produced a range of settling velocities of 0.7-38 m/yr for 50 temperate zone lakes. Chapra's (1979b) analysis suggested that a settling velocity of 16 m/yr provided the best empirical fit to data from 15 Canadian lakes. Phosphorus budget data for 11 segments of the Great Lakes produced apparent phosphorus settling velocities ranging from 10 to 37 m/yr (Chapra and Sonzogni, 1979). The global calibration results shown in Table 27 for the first order settling velocity model indicated that Lake Champlain as a whole had an apparent phosphorus settling velocity of about 38 m/yr. The same global calibration procedure using the 1991 hydrologic base year data produced a whole lake settling velocity estimate of 27 m/yr. These values are higher than the rates found for most lakes, but are within the ranges estimated for the Great Lakes and other multiple lake data sets.

In summary, a second order phosphorus sedimentation coefficient of 100 m^3/g-yr independently derived by Walker (1987) for the BATHTUB program from the U.S. Army Corps of Engineers data set produced an excellent fit between observed and predicted phosphorus concentrations in all segments of Lake Champlain when data from the 1991 hydrologic base year were used in the model and further calibration adjustments were made to three segments. The second order phosphorus sedimentation coefficients indicated in Figure 28C will be used in all subsequent modeling and load reduction analyses presented in this report. Issues related to model confirmation and the reliability of model predictions under altered loading conditions are discussed in the following section.

Model Confirmation

Reckhow and Chapra (1983) discuss several issues related to confirmation of water quality models. Ideally, a "severe" test should be applied in which there is a substantial change in conditions from those under which the model was calibrated.

A common procedure in the confirmation of water quality models is to calibrate the model using data obtained during one time period and test the calibrated model against data independently obtained during another time period. However, it was not possible to use this procedure for the steady-state model developed for Lake Champlain. Phosphorus residence time considerations discussed previously indicated that the two year study period was too short to provide independent loading and lake response relationships between the two years.

An alternative model confirmation approach is to empirically calibrate model coefficients to one cross-sectional lake data set and test the calibrated values against an independent set of lakes.

If the independently calibrated values produce a good fit between observed and model-predicted phosphorus concentrations in the test lakes, then confidence is gained in the model predictions.

The exchange flow rates and the phosphorus net sedimentation coefficients given in Table 23 and Figure 28C were obtained through model calibration procedures designed to produce a good fit between observed and predicted lake chloride and total phosphorus concentrations. Before using the model to predict the future response of the lake to different phosphorus loading rates, there is a need to provide some form of independent confirmation that the calibrated model will produce accurate predictions under loading conditions different from those used in the calibration procedure.

In the case of the exchange rates, confidence in the data used to calculate the chloride mass balance values for each lake segment was gained by the high level of accuracy demonstrated for the lakewide water and chloride input-output budgets (Table 19). Additional credibility in the calibration was established by the strong relationship between the calibrated exchange rates and the morphometry of the exchange interfaces (Figure 27). The relationship shown in Figure 27 provides independent evidence that the exchange values are physically realistic and internally consistent among the lake segments.

Confidence in the phosphorus sedimentation rate calibration is derived primarily from the fact that the second order phosphorus sedimentation coefficient of 100 m^3/g-yr used for most of the Lake Champlain segments was obtained from a prior analysis of an independent reservoir data set by (Walker, 1987). The sedimentation value that was calibrated and tested for the reservoir data produced excellent results for most segments of Lake Champlain without any further calibration for Lake Champlain (Figure 28C).

Segment-specific calibration procedures were applied to phosphorus sedimentation rates in three segments of Lake Champlain. St. Albans Bay is a special case where it was previously known that internal phosphorus loading played a major role in the phosphorus budget for the bay (Smeltzer, 1991). As noted earlier, prediction of the response of St. Albans Bay to external phosphorus loading changes will require knowledge of the future course of internal loading rates using the results of studies by Martin *et al.* (1994) and others. The model calibration procedure applied to St. Albans Bay in this report involved a fixed internal loading rate. While the internal loading rate used in the model for St. Albans Bay could be reduced or eliminated in order to simulate the expected long-term response of the bay to phosphorus loading reductions, this approach must be considered preliminary and unconfirmed pending the results of other on-going studies of internal phosphorus dynamics in St. Albans Bay.

Segment-specific calibration of the second order phosphorus sedimentation rates was also done for Malletts Bay and Missisquoi Bay where Walker's (1987) independently derived value of 100 m^3/g-yr was increased by a factor of four to provide a good fit between observed and predicted phosphorus concentrations in these two bays (Figure 28C). Since no independent test of the calibrated values for Malletts Bay and Missisquoi Bay is available, model predictions of the phosphorus response of these two bays to changes in loadings must also be considered to be relatively unconfirmed. Missisquoi Bay is a region of the lake where substantial phosphorus loading reductions will be needed, and additional research on the internal phosphorus dynamics of the bay would be particularly desirable in order to confirm and enhance the model's predictive capabilities for this lake segment.

PHOSPHORUS LOAD REDUCTION STRATEGY

Baseline Point and Nonpoint Source Phosphorus Loadings

The relative magnitude of point *vs.* nonpoint source phosphorus loadings to Lake Champlain is an issue of major management significance because of the implications for program and funding priorities. An examination of the relative magnitude of point and nonpoint source loadings between river basins and between political jurisdictions was used as a starting point in the process of developing a basin-wide phosphorus load reduction strategy for Lake Champlain.

Total phosphorus loading rates for the 1991 hydrologic base year were used to compare point and nonpoint source loadings from each state and sub-watershed. The point source phosphorus loadings for each tributary (from Table 15) were subtracted from the total loads (derived from Table 24) to estimate the nonpoint source component for each tributary. Atmospheric phosphorus loadings during 1991 and loadings from ungaged areas were also included in the nonpoint source estimates.

The simple subtraction procedure used to estimate the tributary nonpoint source component assumes that all phosphorus entering a stream, from either point or nonpoint sources, is eventually conveyed to the river mouth. If significant quantities of phosphorus are permanently attenuated along the stream course (e.g. in sediments within impoundments), then this procedure will underestimate the relative proportion of nonpoint source loading in the total load observed at the river mouth. However, in the absence of data on in-stream phosphorus transport processes in the Lake Champlain basin that would contradict this assumption, the best approach for the purpose of conducting a load reduction analysis is to assume that essentially all phosphorus discharged to a river is eventually conveyed to the river mouth. This assumption is consistent with point and nonpoint source phosphorus management approaches used in the Great Lakes and elsewhere (Lee *et al.*, 1985; DePinto *et al.*, 1986).

Point and nonpoint source total phosphorus loadings to Lake Champlain calculated for the 1991 hydrologic base year are summarized in Table 28 and Figure 29. Nonpoint sources dominate over point sources as contributors of total phosphorus to Lake Champlain. Overall, nonpoint sources compose about 71% of the total loading estimate of 647 mt/yr. Table 28 and Figure 29 also show that the majority (74%) of the watershed total phosphorus loadings to Lake Champlain are derived from the Vermont and Quebec side of the lake.

The natural background components of the nonpoint source total phosphorus loadings shown in Figure 29 were estimated from the relationship shown in Figure 30. Land use information corresponding to 17 of the monitored watersheds in the Lake Champlain basin were obtained from Budd and Meals (1994). The nonpoint source phosphorus concentrations for these streams (from Table 28) were plotted against the percentage of non-forested (i.e. agricultural or developed) land in each watershed. Figure 30 shows that a strong positive relationship existed between the percentage of agricultural and developed land in each watershed and the phosphorus concentrations in the streams. The relationship between stream phosphorus concentration and land use appeared to be the same for streams on both sides of the lake. The intercept of the regression line in Figure 30 suggests that the mean phosphorus concentration in Lake Champlain tributaries in their original forested state was about 0.015 mg/l (95% confidence interval = 0.010-0.021 mg/l).

The natural background total phosphorus loading rate to Lake Champlain was estimated by applying the 0.015 mg/l stream concentration to all tributaries and ungaged areas listed in Table 28 where current levels are higher than 0.015 mg/l, eliminating the point source loadings, and assuming

Table 28. Summary of point and nonpoint source total phosphorus loading to Lake Champlain for the 1991 hydrologic base year.

Source	Drainage Area (km^2)	Mean Flow (hm^3/yr)	Total Load (mt/yr)	Point Source Load (mt/yr)	Nonpoint Source Load (mt/yr)	Nonpoint Source Export (kg/ha-yr)	Nonpoint Source Conc. (mg/l)
VERMONT/QUEBEC							
Sampled Tributaries							
Winooski	2828	1543	83.8	24.3	59.6	0.21	0.039
Otter	2462	1119	109.7	62.3	47.4	0.19	0.042
Missisquoi	2223	1307	82.1	7.0	75.1	0.34	0.057
Lamoille	1909	1100	29.6	3.0	26.6	0.14	0.024
Poultney[1]	692	273	17.1	3.0	14.1	0.20	0.052
Pike	517	296	50.3	5.9	44.4	0.86	0.150
Lewis	209	90	5.2	0.0	5.2	0.25	0.058
Little Otter	185	55	5.4	0.0	5.4	0.29	0.098
Rock	152	69	28.9	0.0	28.9	1.90	0.419
LaPlatte	137	44	11.8	4.2	7.6	0.56	0.173
East	81	24	1.3	0.1	1.2	0.15	0.051
Malletts	76	31	1.7	0.0	1.7	0.22	0.055
Mill	59	26	3.5	0.0	3.5	0.59	0.135
Stevens	59	14	3.4	0.0	3.4	0.58	0.236
Stone Bridge	32	10	0.78	0.0	0.8	0.24	0.075
Indian	31	13	0.92	0.0	0.9	0.30	0.069
Ungaged Areas	429	175	14.9	0.0	14.9	0.35	0.085
Direct Wastewater Discharges		14	19.5	19.5	0.0		
Vermont/Quebec Total	12081	6204	469.9	129.3	340.6	0.28	0.055
NEW YORK							
Sampled Tributaries							
Saranac	1575	776	16.4	8.7	7.7	0.05	0.010
Ausable	1323	639	16.8	5.6	11.2	0.08	0.018
Mettawee/Barge Canal[2]	1098	487	37.1	3.4	33.7	0.31	0.069
Great Chazy	769	320	17.7	1.1	16.6	0.22	0.052
Bouquet	712	281	13.5	0.0	13.5	0.19	0.048
LaChute	702	273	1.1	0.0	1.1	0.02	0.004
Little Ausable	189	89	5.2	1.4	3.8	0.20	0.043
Salmon	175	55	1.7	0.0	1.7	0.10	0.031
Putnam	160	67	1.3	0.0	1.3	0.08	0.019
Little Chazy	139	44	3.2	0.0	3.2	0.23	0.073
Mill (Port Henry)	73	25	0.63	0.0	0.6	0.09	0.025
Mt. Hope	30	13	0.09	0.0	0.1	0.03	0.007
Highlands Forge	30	8.8	0.10	0.0	0.1	0.03	0.011
Hoisington	28	9.6	0.48	0.0	0.5	0.17	0.050
Mill (Putnam Sta.)	27	9.0	0.42	0.0	0.4	0.16	0.047
Ungaged Areas	599	249	6.8	0.0	6.8	0.11	0.027
Direct Wastewater Discharges		39	38.9	38.9	0.0		
New York Total	7629	3384	161.4	59.2	102.2	0.13	0.031
DIRECT PRECIPITATION	1130	915	15.4	0.0	15.4	0.14	0.017
LAKE TOTAL	20840	10503	646.7	188.5	458.2	0.22	0.044

[1] Includes some drainage area in New York.

[2] Includes some drainage area and one wastewater discharge in Vermont.

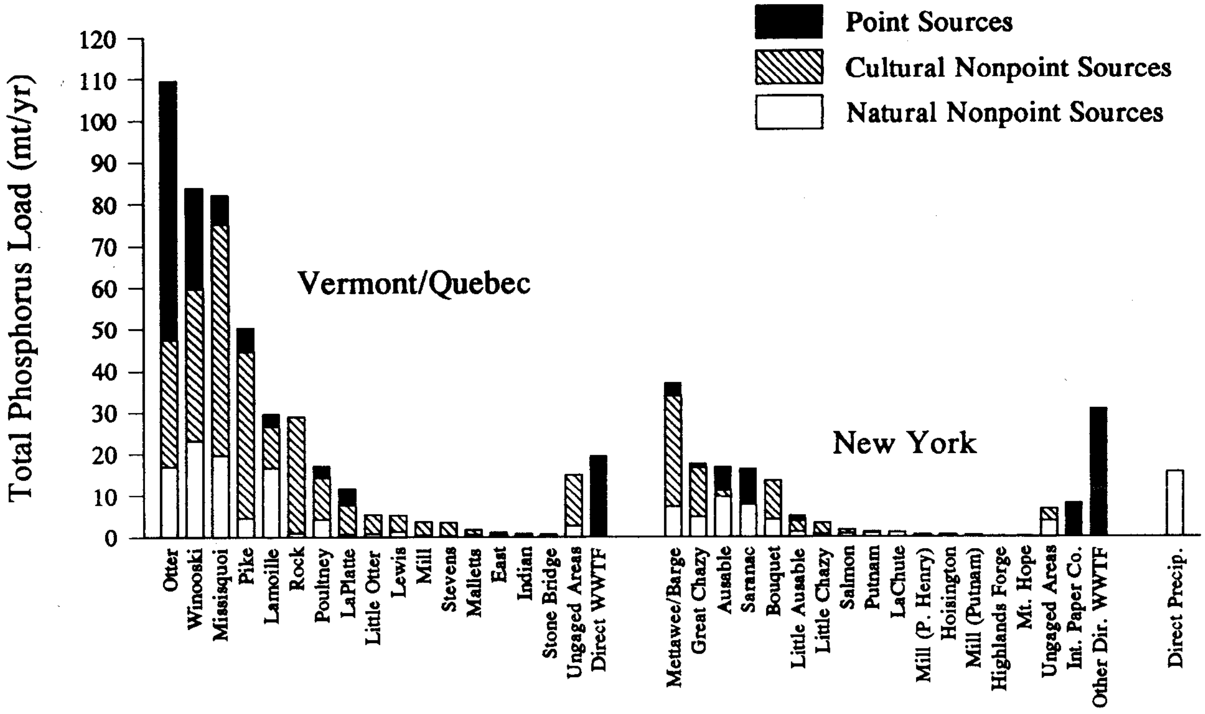


Figure 29. Summary of point and nonpoint source phosphorus loadings to Lake Champlain for the 1991 hydrologic base year.

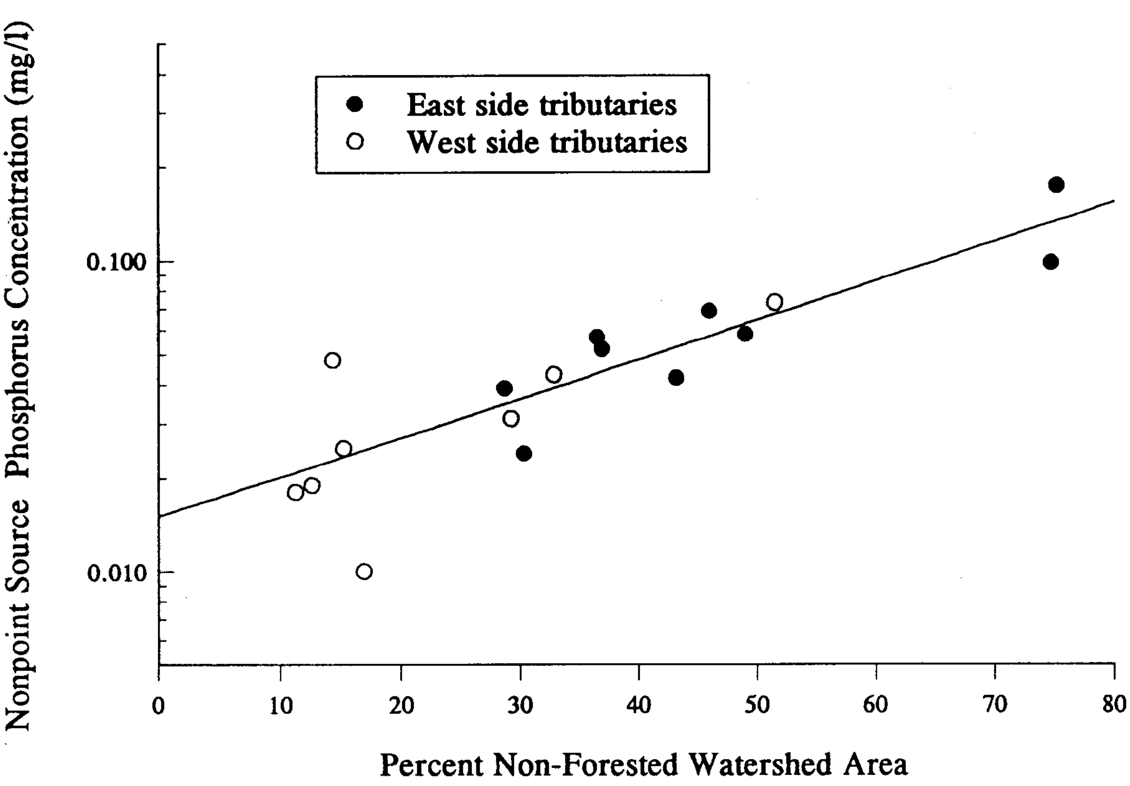


Figure 30. Relationship between nonpoint source total phosphorus concentration and the percentage of non-forested land area in 17 tributary watersheds in the Lake Champlain basin. The intercept of the regression line is at 0.015 mg/l.

that atmospheric phosphorus loadings have remained unchanged. Using this procedure, the natural background component of the phosphorus loading to Lake Champlain was estimated to be 151 mt/yr, or about 33% of the 458 mt/yr total nonpoint source load, and 23% of the 645 mt/yr total loading to the lake. The estimates of "natural nonpoint sources" shown in Figure 29 were based on the 0.015 mg/l concentration, with the balance of the total nonpoint source loading assumed to represent "cultural nonpoint sources". Apparently, human activities in the Lake Champlain Basin have increased phosphorus loading to the lake four-fold over natural background levels.

Earlier total phosphorus budgets for Lake Champlain (summarized in Bogdan, 1978) estimated the nonpoint source component to be 274-419 mt/yr, and only 46-58% of the total loading. The values given in Table 28 indicate a larger nonpoint source phosphorus loading rate of 458 mt/yr during 1991 (71% of the total loading). While it is possible that nonpoint source phosphorus loadings to Lake Champlain have increased since the 1970's, another likely reason for the difference is that the nonpoint loadings were under-estimated during the earlier studies, as suggested by Bogdan (1978), because of their failure to adequately sample high flow events or to account for concentration *vs.* flow dependence in the load estimation procedures.

The ranking of tributaries according to total phosphorus loading rates shown in Figure 29 is strongly influenced by the size of their respective drainage areas and is therefore not a good indication of the relative degree of phosphorus enrichment in each stream. The 31 major Lake Champlain tributaries are ranked according to their total phosphorus flow-weighted mean nonpoint source concentrations in Figure 31. The ranking based on nonpoint source phosphorus concentration shown in Figure 31 normalizes for the effects of different drainage basin sizes and provides a basis for identifying those watersheds with the most severe nonpoint source phosphorus impacts.

Figure 31 shows that in general, nonpoint source phosphorus concentrations are higher on the Vermont and Quebec side of the lake than on the New York side. Phosphorus concentrations are particularly high for the Rock, Stevens, LaPlatte, Pike, and Mill Rivers, which drain heavily agricultural regions of Vermont and Quebec

General Load Reduction Approach

The lake phosphorus mass balance model developed for this study was used to analyze phosphorus load reduction strategies to attain the in-lake criteria listed in Table 2. Several alternative approaches were considered and discussed by the States of Vermont and New York and the Lake Champlain Management Conference over a period of several years, as described in earlier drafts of this report and previous drafts of the Comprehensive Plan for Lake Champlain prepared by the Management Conference.

In 1996, the States of Vermont and New York and the U.S. Environmental Protection Agency negotiated a phosphorus reduction agreement for Lake Champlain. This agreement is presented in the Lake Champlain Management Conference (1996) plan titled *Opportunities for Action, an Evolving Plan for the Future of the Lake Champlain Basin.* The agreement establishes target phosphorus loads for point and nonpoint sources for each lake segment and for each state.

The load reduction agreement negotiated between the states first defined the point source loading targets for each state by assuming implementation of advanced wastewater treatment throughout the Lake Champlain Basin at the larger facilities where such treatment is the most cost-effective. The remaining load reductions necessary to attain the in-lake criteria were then targeted at specific lake segment watersheds using the lake phosphorus model with a minimum-cost optimization procedure, as

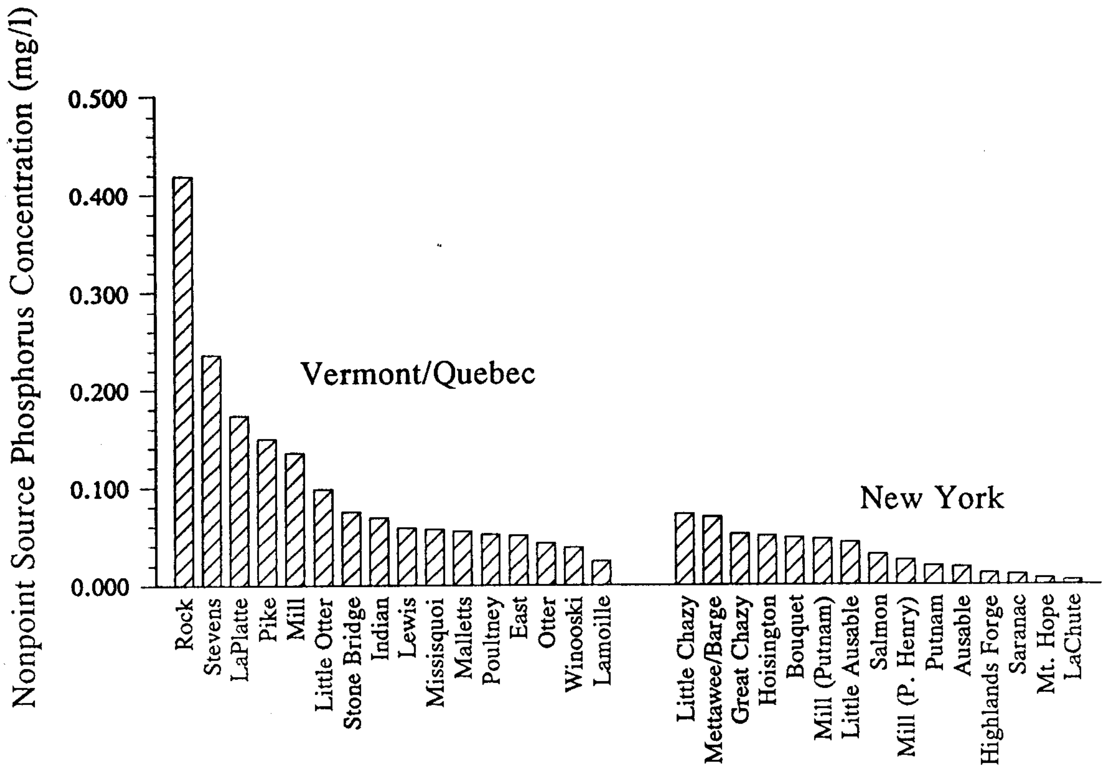


Figure 31. Nonpoint source total phosphorus concentrations (flow-weighted means) for 31 major Lake Champlain tributaries during the 1991 hydrologic base year.

described in more detail below. The states retain flexibility under the agreement to adjust the point and nonpoint source target loads within each state in the future, provided that the model is used to ensure that the in-lake criteria will be attained and the target loads for the other state remain unaffected by the adjustment.

Point Source Loading Targets

Current Point Source Policies

Point source phosphorus loading targets for Lake Champlain were developed using the current laws and regulations in Vermont, New York, and Quebec as starting points. Current policies in each jurisdiction applicable to Lake Champlain discharges are summarized in the report of the Lake Champlain Phosphorus Management Task Force (1993), and are reviewed below.

In Vermont, a phosphorus reduction statute (10 V.S.A. §1266a) enacted in 1992 established a basin-wide phosphorus effluent limit of 0.8 mg/l applicable to all discharges greater than 0.2 mgd, and exempting facilities using aerated lagoon treatment processes. Compliance with this section is required only to the extent that 100% state funding is provided to municipalities for the construction cost of phosphorus removal facilities.

The statutory 0.8 mg/l limit applies to 28 of the municipal discharges in Vermont listed in Table 15, of which 17 have already been upgraded to attain the 0.8 mg/l limit. All of the currently operating industrial discharges in the Vermont portion of the Lake Champlain Basin are meeting the 0.8 mg/l limit as well. Some industrial facilities such as the Edgar Weed Fish Culture Station and the Vermont Whey Co. have more stringent effluent phosphorus loading limits in their discharge permits that were derived from site-specific water quality considerations. Most of the other Vermont municipal facilities in the Lake Champlain basin are exempt from any specific phosphorus effluent limits under current policy. However, Vermont Water Quality Standards require that there be "no significant increase over currently permitted loadings" to Lake Champlain, and wastewater treatment facilities undergoing expansion have been required to maintain their current loadings under this rule.

In New York, a Department of Environmental Conservation, Division of Water, Technical and Operational Guidance Series document (1.3.6) recommends that *new* discharges in lake watersheds employ sub-surface disposal at facilities where the permitted flow is less than 10,000 gal/day, limit the permitted effluent phosphorus concentration to 1.0 mg/l at facilities with 10,000-50,000 gal/day permitted flow, and limit phosphorus to 0.5 mg/l at facilities with greater than 50,000 gal/day permitted flow. Under these guidelines, *existing* permitted discharges undergoing flow expansion must maintain their current phosphorus loading without increase, and other existing discharges must employ phosphorus removal treatment if the need is demonstrated through a special study or "detailed analysis".

Five discharges in the New York portion of the Lake Champlain Basin currently have phosphorus effluent limits in their state discharge permits. The International Paper Co. is limited to 88 lbs/day total phosphorus loading and a maximum concentration of 0.5 mg/l. The Wyeth-Ayerst, Chazy facility is limited to an effluent concentration of 0.5 mg/l. The Great Meadows and Washington Correctional Facilities have effluent phosphorus limits of 1.0 mg/l, and the Altona Correctional Facility is limited to 1.75 lbs/day. Two municipal discharges will have effluent phosphorus limits in the near future, including Champlain (5.6 lbs/day) and a new facility at Cadyville (0.35 lbs/day). No other municipal or industrial facilities in New York have phosphorus concentration or loading limits specified in their discharge permits under current policy.

In Quebec, there is an active sewer and wastewater treatment facility construction program in the Missisquoi Bay watershed that will result in the construction of treatment plants for all municipal and industrial discharges in the basin. All the Quebec facilities listed in Table 15 will be required to meet a 1.0 mg/l effluent limit.

Point Source Load Reduction Analysis

Table 29 lists all the wastewater treatment facilities in the Lake Champlain Basin that discharge significant quantities of phosphorus. Effluent monitoring data for average annual flows, total phosphorus concentrations, and phosphorus mass loading rates estimated for the 1991 base year are provided in Table 29, consistent with the values presented in Table 15. Substantial phosphorus load reductions occurred at some facilities between 1991 and 1995 as a result of phosphorus removal upgrades and other operational changes. Table 29 also shows the effluent monitoring data updated to 1995. In cases where effluent phosphorus concentration data for 1995 were not available, it was assumed that the 1990-1991 measured average value remained the same.

Updated information for the Quebec facilities was not available for 1995. For the purpose of Table 29 and subsequent load reduction analyses, it was assumed that the Quebec facilities listed in Table 15 are operating at their full permitted flows with effluent phosphorus concentrations of 1.0 mg/l.

Table 29 shows that the total point source load from Vermont and Quebec declined from 130 mt/yr in 1991 to 67 mt/yr in 1995. The reductions were primarily the result of reduced industrial influent loadings and operational changes at the Middlebury treatment plant, and construction of phosphorus removal facilities at Rutland City, Burlington Main, and Hinesburg. Point source loading from New York declined from 59 mt/yr in 1991 to 33 mt/yr in 1995. The decline in New York was primarily the result of reduced influent phosphorus loading to the Plattsburgh City treatment facility brought about by operational changes at the Georgia-Pacific plant and the closing of the Plattsburgh Air Force Base. An upgrade of the Saranac Lake treatment facility also resulted in reduced phosphorus loading from that municipality.

The data given in Table 29 were used to define preliminary point source loading targets for each segment of Lake Champlain. The preliminary loading targets assumed application of the Vermont phosphorus removal policy in both Vermont and New York. The Vermont statute limits the effluent phosphorus concentration to 0.8 mg/l at non-lagoon treatment facilities larger than 0.2 mgd permitted flow. For *all* facilities, the point source target values were calculated using either the permitted flow or 1.5 times the current (1995) flow, whichever was less. For facilities that are affected by the 0.8 mg/l policy, the loading targets were calculated using the minimum of either the current (1995) concentration or 0.8 mg/l. For facilities that are not affected by the 0.8 mg/l policy, the load targets were calculated using the current concentration.

The preliminary loading targets calculated in this manner for each facility are listed in Table 29. However, these targets were intended to apply in aggregate to each lake segment watershed, and not to individual facilities. Table 30 shows the loading rates and target loads from Table 29, summed by state and by lake segment. The preliminary point source target loads indicated in Table 30 were incorporated into the phosphorus reduction agreement endorsed by the Lake Champlain Management Conference (1996).

Table 29. **List of wastewater treatment facilities in the Lake Champlain Basin showing average annual flows, effluent total phosphorus concentrations, and loading rates estimated for 1991 and 1995. Target loads (calculated using 1.5X the 1995 flow where applicable, according to the procedure described in the text) and loads expected at full permitted flows are also indicated. Lake segments are numbered as in Table 30. Affected facilities (Y=yes, N=no) refer to the 0.8 mg/l policy described in the text. "ND" indicates no data available.**

Facility	Lake Segment	Affected Facility? (Y/N)	1991 Flow (mgd)	1995 Flow (mgd)	Permitted Flow (mgd)	1991 Conc. (mg/l)	1995 Conc. (mg/l)	Permitted Conc. (mg/l)	1991 Load (mt/yr)	1995 Load (mt/yr)	Target Load at 1.5X Current Flow (mt/yr)	Load at Permitted Flow (mt/yr)
Vermont/Quebec												
Benson	1	N	0.009	0.009	0.018	4.34	4.34	4.34	0.054	0.051	0.077	0.106
Castleton	1	Y	0.263	0.273	0.360	2.15	2.15	0.80	0.781	0.812	0.398	0.398
Fair Haven	1	Y	0.289	0.296	0.500	3.02	3.14	0.80	1.205	1.285	0.491	0.553
Poultney	1	Y	0.318	0.286	0.350	2.14	2.14	0.80	0.940	0.845	0.387	0.387
West Pawlet	1	N	0.020	0.011	0.040	6.30	6.30	6.30	0.174	0.099	0.148	0.348
Orwell	2	N	0.027	0.022	0.033	2.00	2.00	2.00	0.075	0.060	0.090	0.091
Brandon	4	Y	0.327	0.226	0.700	1.40	1.40	0.80	0.632	0.437	0.374	0.774
Green Mt. Trout Farm	4	N	ND	0.144	0.144	0.08	0.08	0.08	ND	0.016	0.016	0.016
Middlebury	4	Y	1.103	0.735	2.200	28.69	4.82	0.80	43.702	4.893	1.219	2.432
Pittsford	4	N	0.081	0.045	0.070	2.63	2.63	2.63	0.294	0.163	0.244	0.254
Pittsford Fish Hatchery	4	N	5.000	5.000	5.000	0.10	0.10	0.10	0.691	0.691	0.691	0.691
Proctor	4	N	0.248	0.229	0.325	2.18	2.18	2.18	0.747	0.689	0.979	0.979
Rutland City	4	Y	5.407	4.165	6.800	2.03	0.29	0.80	15.158	1.669	2.504	7.518
Vergennes	4	Y	0.467	0.330	0.660	0.70	0.35	1.00	0.451	0.157	0.236	0.912
Wallingford	4	N	0.110	0.108	0.120	2.98	2.98	2.98	0.453	0.444	0.494	0.494
West Rutland	4	Y	0.236	0.219	0.325	2.00	2.00	0.80	0.652	0.606	0.359	0.359
Barre City	5	Y	2.938	2.282	3.800	1.36	2.00	0.80	5.518	6.305	3.785	4.201
Burlington East	5	Y	0.805	0.714	1.000	0.68	0.28	0.80	0.756	0.277	0.388	1.106
Burlington North	5	Y	1.303	0.906	2.000	1.91	0.57	0.80	3.437	0.708	1.062	2.211
Essex Jct.	5	Y	1.487	1.517	2.750	0.78	0.67	0.80	1.602	1.406	2.110	3.040
IBM	5	N	1.957	3.062	3.500	0.26	0.17	0.17	0.703	0.736	0.842	0.842
Marshfield	5	N	0.018	0.017	0.045	3.94	3.94	3.94	0.098	0.094	0.142	0.245
Montpelier	5	Y	2.044	1.550	3.970	2.52	2.52	0.80	7.113	5.398	2.571	4.389
Northfield	5	Y	1.096	0.438	1.630	2.35	2.35	0.80	3.557	1.423	0.727	1.802
Plainfield	5	N	0.062	0.056	0.100	4.29	4.29	4.29	0.367	0.332	0.499	0.593
Richmond	5	Y	0.083	0.093	0.222	6.12	6.12	0.80	0.701	0.782	0.153	0.245
SouthBurlington Airport Park.	5	Y	1.313	1.312	2.300	0.76	0.57	0.80	1.378	1.029	1.544	2.543
Stowe	5	Y	0.147	0.130	0.250	0.22	0.17	0.80	0.045	0.030	0.045	0.276

Facility	Lake Segment	Affected Facility? (Y/N)	1991 Flow (mgd)	1995 Flow (mgd)	Permitted Flow (mgd)	1991 Conc. (mg/l)	1995 Conc. (mg/l)	Permitted Conc. (mg/l)	1991 Load (mt/yr)	1995 Load (mt/yr)	Target Load at 1.5X Current Flow (mt/yr)	Load at Permitted Flow (mt/yr)
Waterbury	**5**	**N**	**0.239**	**0.244**	**0.510**	**4.95**	**4.95**	**4.95**	**1.634**	**1.668**	**2.502**	**3.489**
Weed Fish Culture Station	**5**	**N**	**0**	**5.921**	**11.500**	**0**	**0.06**	**0.06**	**0**	**0.529**	**0.707**	**0.915**
Williamstown	**5**	**N**	**0.075**	**0.088**	**0.150**	**2.11**	**2.11**	**2.11**	**0.219**	**0.257**	**0.385**	**0.437**
Winooski	**5**	**Y**	**0.837**	**0.716**	**1.200**	**0.52**	**0.59**	**0.80**	**0.601**	**0.585**	**0.878**	**1.327**
Hinesburg	**6**	**Y**	**0.177**	**0.203**	**0.250**	**17.09**	**0.66**	**0.80**	**4.177**	**0.185**	**0.228**	**0.276**
Shelburne F.D.#1	**6**	**Y**	**0.306**	**0.268**	**0.310**	**0.67**	**0.37**	**0.80**	**0.283**	**0.138**	**0.159**	**0.343**
Shelburne F.D.#2	**6**	**Y**	**0.278**	**0.302**	**0.450**	**0.70**	**0.50**	**0.80**	**0.269**	**0.210**	**0.313**	**0.498**
South Burlington Bart. Bay	**6**	**Y**	**0.727**	**0.607**	**0.800**	**0.58**	**0.22**	**0.80**	**0.582**	**0.188**	**0.248**	**0.884**
Burlington Main	**7**	**Y**	**3.841**	**4.102**	**5.300**	**2.12**	**0.39**	**0.80**	**11.245**	**2.188**	**2.827**	**5.860**
Fairfax	**9**	**N**	**0.040**	**0.040**	**0.078**	**3.95**	**3.95**	**3.95**	**0.218**	**0.216**	**0.324**	**0.426**
Hardwick	**9**	**N**	**0.216**	**0.148**	**0.371**	**2.75**	**2.75**	**2.75**	**0.820**	**0.563**	**0.845**	**1.410**
Jeffersonville	**9**	**N**	**0.032**	**0.041**	**0.077**	**2.00**	**2.00**	**2.00**	**0.088**	**0.113**	**0.170**	**0.213**
Johnson	**9**	**Y**	**0.154**	**0.182**	**0.200**	**1.82**	**1.82**	**0.80**	**0.387**	**0.458**	**0.221**	**0.221**
Milton	**9**	**Y**	**0.163**	**0.135**	**0.225**	**0.56**	**0.56**	**0.56**	**0.126**	**0.105**	**0.157**	**0.174**
Morrisville	**9**	**Y**	**0.335**	**0.277**	**0.425**	**2.70**	**2.70**	**0.80**	**1.249**	**1.034**	**0.460**	**0.470**
Vermont Whey	**9**	**N**	**0.333**	**0.414**	**0.360**	**0.36**	**0.42**	**0.78**	**0.166**	**0.239**	**0.388**	**0.388**
Northwest State Correctional	**11**	**Y**	**0.018**	**0.024**	**0.040**	**0.17**	**0.21**	**0.50**	**0.004**	**0.007**	**0.010**	**0.028**
St. Albans City	**11**	**Y**	**2.114**	**2.383**	**4.000**	**0.27**	**0.49**	**0.50**	**0.788**	**1.614**	**2.421**	**2.764**
Agrimark	**12**	**N**	**0.044**	**0**	**0**	**15.38**	**0**	**0**	**0.935**	**0**	**0**	**0**
Enosburg Falls	**12**	**Y**	**0.364**	**0.316**	**0.450**	**2.65**	**6.94**	**0.80**	**1.332**	**3.032**	**0.498**	**0.498**
Newport Center	**12**	**N**	**0.018**	**0.013**	**0.042**	**0.10**	**0.10**	**0.10**	**0.002**	**0.002**	**0.003**	**0.006**
North Troy	**12**	**N**	**0.082**	**0.065**	**0.110**	**1.90**	**1.90**	**1.90**	**0.215**	**0.171**	**0.256**	**0.289**
Quebec Facilities	**12**	**N**	**1.543**	**1.543**	**1.373**				**8.524**	**1.896**	**1.896**	**1.896**
Richford	**12**	**N**	**0.347**	**0.276**	**0.380**	**1.04**	**1.04**	**1.04**	**0.498**	**0.396**	**0.546**	**0.546**
Rock Tenn	**12**	**N**	**2.515**	**0.181**	**3.500**	**0.40**	**0.40**	**0.40**	**1.389**	**0.100**	**0.150**	**1.935**
Sheldon Springs	**12**	**N**	**0.025**	**0.033**	**0.054**	**2.04**	**2.04**	**2.04**	**0.070**	**0.093**	**0.140**	**0.152**
Swanton	**12**	**Y**	**0.742**	**0.675**	**0.900**	**2.38**	**0.24**	**1.00**	**2.439**	**0.220**	**0.294**	**1.244**
Troy/Jay	**12**	**N**	**0**	**0.071**	**0.200**	**0**	**10.20**	**10.20**	**0**	**1.003**	**1.505**	**2.819**
Alburg	**13**	**N**	**0.069**	**0.085**	**0.130**	**0.03**	**0.04**	**0.04**	**0.003**	**0.005**	**0.007**	**0.008**
Vermont/Quebec TOTAL			**42.792**	**43.527**	**72.596**				**129.548**	**48.655**	**41.112**	**67.323**

Facility	Lake Segment	Affected Facility? (Y/N)	1991 Flow (mgd)	1995 Flow (mgd)	Permitted Flow (mgd)	1991 Conc. (mg/l)	1995 Conc. (mg/l)	Permitted Conc. (mg/l)	1991 Load (mt/yr)	1995 Load (mt/yr)	Target Load at 1.5X Current Flow (mt/yr)	Load at Permitted Flow (mt/yr)
New York												
Fort Ann	1	N	0.067	0.050	0.110	2.70	2.13	2.13	0.250	0.147	0.221	0.324
Granville	1	Y	0.793	0.700	0.650	1.50	1.75	0.80	1.643	1.693	0.719	0.719
Great Meadows Correctional	1	Y	0.371	0.280	0.400	2.00	0.50	0.80	1.025	0.193	0.276	0.442
Washington Correctional	1	Y	0.110	0.120	0.250	2.00	0.48	0.80	0.304	0.080	0.119	0.276
Whitehall	1	Y	0.539	0.600	0.600	0.96	0.72	0.80	0.715	0.597	0.597	0.663
Crown Point	2	N	0	0.020	0.060	0	2.11	2.11	0.000	0.058	0.087	0.175
International Paper Co.	2	Y	17.464	16.920	17.000	0.34	0.27	0.50	8.200	6.314	6.343	11.747
Ticonderoga	2	Y	0.902	0.800	1.500	1.12	0.59	0.80	1.395	0.652	0.978	1.658
Port Henry	3	Y	0.588	0.600	0.440	1.92	3.03	0.80	1.559	2.512	0.486	0.486
Westport	3	N	0.082	0.075	0.120	1.81	1.35	1.35	0.205	0.140	0.210	0.224
Ausable Forks	5	N	0.043	0.080	0.147	3.70	4.47	4.47	0.220	0.494	0.741	0.908
Keeseville	5	Y	0.264	0.234	0.300	3.25	2.98	0.80	1.185	0.964	0.332	0.332
Lake Placid	5	Y	1.341	1.300	2.500	2.27	2.30	0.80	4.204	4.132	2.156	2.764
Peru	5	Y	0.254	0.220	0.500	4.00	2.10	0.80	1.403	0.638	0.365	0.553
Peru/Valcour	5	N	0.005	0.003	0.048	2.27	0.42	0.42	0.016	0.002	0.003	0.028
Wadhams	5	N	0.006	0.006	0.015	5.10	3.20	3.20	0.042	0.027	0.040	0.066
Willsboro	5	N	0	0.060	0.125	0	5.20	5.20	0.000	0.431	0.647	0.898
Adirondak Fish Culture Station	8	N	2.623	2.600	3.600	0.05	0.03	0.03	0.181	0.108	0.149	0.149
Dannemora	8	N	0.699	0.611	1.500	3.08	2.65	2.65	2.973	2.238	3.356	5.493
Plattsburgh City	8	Y	8.402	7.700	16.000	1.74	0.68	0.80	20.190	7.236	10.854	17.690
Plattsburgh/Champlain Park	8	N	0.131	0.050	0.162	1.62	2.79	2.79	0.293	0.193	0.289	0.625
Saranac Lake	8	Y	1.884	1.350	2.800	2.06	1.01	0.80	5.360	1.884	2.239	3.096
St. Armand	8	N	0.032	0.027	0.060	3.89	5.07	5.07	0.172	0.189	0.284	0.420
Altona Correctional	13	N	0.066	0.080	0.080	2.00	0.71	0.71	0.182	0.078	0.078	0.078
Champlain	13	Y	0.323	0.380	0.400	2.05	1.20	0.80	0.914	0.630	0.442	0.442
Rouses Point	13	Y	0.814	0.810	2.000	1.62	1.47	0.80	1.821	1.646	1.343	2.211
Wyeth-Ayerst, Chazy	13	N	0.039	0.078	0.078	83.80	0.50	0.50	4.513	0.054	0.054	0.054
Wyeth-Ayerst, Rouses Point	13	N	ND	0.225	0.225	ND	0.23	0.23	ND	0.072	0.072	0.072
New York TOTAL			37.842	35.979	51.670				58.964	33.402	33.482	52.595

Table 30. **Point source phosphorus loads to Lake Champlain, grouped by state and lake segment. Estimated 1991 and 1995 loads, target loads (calculated using 1.5 times the 1995 flow where applicable, according to the procedure described in the text), and loads expected at full permitted flows are indicated.**

	Vermont/Quebec				**New York**			
Lake Segment	**1991 Load (mt/yr)**	**1995 Load (mt/yr)**	**Target Load at 1.5X Current Flow (mt/yr)**	**Load at Permitted Flow (mt/yr)**	**1991 Load (mt/yr)**	**1995 Load (mt/yr)**	**Target Load at 1.5X Current Flow (mt/yr)**	**Load at Permitted Flow (mt/yr)**
1. South Lake B	3.2	3.1	1.5	1.8	3.9	2.7	1.9	2.4
2. South Lake A	0.1	0.1	0.1	0.1	9.6	7.0	7.4	13.6
3. Port Henry	0.0	0.0	0.0	0.0	1.8	2.7	0.7	0.7
4. Otter Creek	62.8	9.8	7.1	14.4	0.0	0.0	0.0	0.0
5. Main Lake	27.7	21.6	18.3	27.7	7.1	6.7	4.3	5.5
6. Shelburne Bay	5.3	0.7	0.9	2.0	0.0	0.0	0.0	0.0
7. Burlington Bay	11.2	2.2	2.8	5.9	0.0	0.0	0.0	0.0
8. Cumberland Bay	0.0	0.0	0.0	0.0	29.2	11.8	17.2	27.5
9. Malletts Bay	3.1	2.7	2.6	3.3	0.0	0.0	0.0	0.0
10. Northeast Arm	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
11. St. Albans Bay	0.8	1.6	2.4	2.8	0.0	0.0	0.0	0.0
12. Missisquoi Bay	15.4	6.9	5.3	9.4	0.0	0.0	0.0	0.0
13. Isle LaMotte	0.0	0.0	0.0	0.0	7.4	2.5	2.0	2.9
TOTAL	129.5	48.7	41.1	67.3	59.0	33.3	33.5	52.6

Tables 29 and 30 also show the phosphorus loads that would occur with growth to the full permitted flow capacity at all facilities, assuming full implementation of the 0.8 mg/l phosphorus policy in both Vermont and New York. Although this level of growth is not likely to occur at all facilities, the future point source phosphorus loads under this condition would be 67 mt/yr from Vermont and Quebec, and 53 mt/yr from New York.

Point Source Phosphorus Removal Costs

In Vermont, municipalities directed to reduce phosphorus from their wastewater treatment facility discharges pursuant to 10 V.S.A. §1266a currently receive a 100% state grant on eligible project costs for the construction of facilities necessary to achieve the directed level of phosphorus reduction. Annual operating costs are normally borne by the sewer system users. New York currently has no general state requirement for wastewater phosphorus removal in the Lake Champlain Basin. However, the passage of the New York State Environmental Bond Act in November 1996 provides a funding mechanism for the construction of phosphorus removal facilities in the New York portion of the basin.

The capital construction costs for implementing the Vermont phosphorus removal statute in the Vermont portion of the Lake Champlain Basin are listed in Table 31. The state grant project costs in Table 31 were provided by the Facilities Engineering Division of the Vermont Department of Environmental Conservation. A total of 17 facilities have already been upgraded at a cost of $15,731,937. The remaining 11 facilities will require $9,940,000, of which $300,000 in state funds have already been provided to Enosburg Falls.

Phosphorus removal costs to attain a 0.8 mg/l effluent limit at the 13 affected New York municipal and state facilities listed in Table 29 were estimated by Holmes and Artuso (1996). The total capital cost to upgrade these 13 facilities was estimated to be $3,389,300. However, the City of Plattsburgh has indicated that an additional $5,000,000 may be required to consistently achieve a 0.8 mg/l effluent phosphorus limit in the future at that facility.

Nonpoint Source Loading Targets

General Load Reduction Modeling Approach

The lake phosphorus mass balance model developed for Lake Champlain was used to determine the watershed nonpoint source loading values necessary to achieve the in-lake phosphorus criteria listed in Table 2. The procedure assumed that point source loads were at the target levels indicated in Table 30 for each state and lake segment. The load reduction modeling procedure was used to define nonpoint source loading targets for the watersheds of each of the 13 lake segments shown in Figure 1.

The nonpoint source load reduction targeting procedure used the 1991 base year nonpoint source loads (Table 28) as a starting point in the analysis. The 1991 base year loads were modified to account for load reductions expected from recently implemented agricultural best management practices (BMPs) in Vermont and New York, and for reductions anticipated as a result of the adoption in 1995 of Accepted Agricultural Practice (AAP) Rules in Vermont. The remaining nonpoint source load reductions necessary to attain the in-lake criteria were then targeted to specific lake segment watersheds using the lake phosphorus model with a minimum-cost optimization procedure, as described in more detail below. Information on the cost and effectiveness of agricultural and urban phosphorus control practices was used to support the optimization procedure.

Table 31. Phosphorus removal upgrade schedule and project construction costs for Vermont municipal wastewater treatment facilities in the Lake Champlain Basin.

Facility	Completion Date	Phosphorus Removal Project Cost ($)
Stowe	1979	489,636
Vergennes	1980	812,961
Winooski	1980	(not avail.)
Essex Jct.	1985	1,405,000
South Burlington Air. Park.	1987	974,000
South Burlington Bart. Bay	1987	122,000
St. Albans and NW Corr.	1987	2,756,000
Shelburne #1	1988	386,964
Swanton	1989	1,358,380
Burlington East	1991	(included in 'Main')
Burlington Main	1991	1,922,000
Burlington North	1991	(included in 'Main')
Shelburne #2	1992	386,774
Rutland City	1993	171,400
Hinesburg	1994	508,874
Barre City	1995	3,667,948
Johnson	1995	770,000
Enosburg Falls	1998-2001[1]	500,000
Castleton	1998-2001[1]	570,000
Morrisville	1998-2001[1]	520,000
Middlebury	1998-2001[1]	1,900,000
Brandon	1998-2001[1]	600,000
Poultney	1998-2001[1]	992,000
Fair Haven	1998-2001[1]	510,000
West Rutland	1998-2001[1]	1,300,000
Northfield	1998-2001[1]	500,000
Richmond	1998-2001[1]	1,020,000
Montpelier	1998-2001[1]	1,528,000
		25,671,937

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[1] Date of completion will depend on the rate of state and federal funding.

All other model input data, calibration coefficients, and error terms remained the same in the load reduction modeling analysis as documented in the Model Development section of this report, except for the internal phosphorus loading rate applied to St. Albans Bay. Research by Martin *et al.* (1994) indicated that sediment phosphorus levels and net internal loading rates in St. Albans Bay are gradually declining following the significant point source phosphorus loading reduction that occurred in 1986. While the internal loading rate to St. Albans Bay that will exist when the system eventually re-equilibrates with its new loadings is not known with precision, it was assumed for the purpose of the load reduction modeling procedure that the current internal loading rate of 3.3 mg/m^2-day will decline to zero, and that the second order phosphorus net sedimentation rate will remain at zero (i.e. that phosphorus will behave conservatively in the bay). The long-term sediment-water mass balance model developed for St. Albans Bay by Martin *et al.* (1994) indicated that net internal loading to the system should actually decline to negative values (i.e. that a net flux of phosphorus from the water column to the sediments will eventually be restored).

To support the optimization procedure, the lake phosphorus mass balance model developed using the BATHTUB program was transferred to a spreadsheet format. Model equations and optimization solutions were conducted using numeric tools provided by the spreadsheet program, as described in more detail below.

Nonpoint Source Control Cost and Effectiveness

Agricultural Controls

Information on the cost and expected phosphorus reduction effectiveness of agricultural nonpoint source controls in the Lake Champlain Basin was provided by the U.S. Natural Resources Conservation Service (R. Croft, personal communication). Cost data and agricultural phosphorus runoff models developed for local application by the Vermont Office of the U.S. Natural Resources Conservation Service were used to estimate the cost-effectiveness of the treatments listed in Table 32. Costs and phosphorus reduction potential were estimated on a per animal unit basis (1 au = 1,000 lbs = 454 kg animal weight). Costs were expressed on a present value basis, including capital costs as well as annual operation and maintenance costs, assuming a 5% discount rate over a 30-year period, following Holmes and Artuso (1995). The treatments considered in the analysis were limited to dairy situations, which represent the majority of the farming activities in the Lake Champlain Basin.

Individual farm data for the Vermont and New York portions of the Lake Champlain Basin were compiled from farm inventories conducted by the Vermont and New York Offices of the U.S. Natural Resources Conservation Service and the Vermont Department of Agriculture, Food, and Markets. These inventories contained information on the treatment practices in place and the additional practices needed on each farm. Using the farm inventory data, the U.S. Natural Resources Conservation Service applied the values listed in Table 32 (derived from Vermont agricultural phosphorus runoff model runs on over 200 farms in the basin) to estimate the maximum potential phosphorus load reduction and the cost of full implementation of all needed treatments on each farm.

The original farm inventory data were coded for each U.S. Natural Resources Conservation Service 11-digit hydrologic unit in the Lake Champlain Basin. The farm data were distributed for this analysis to the appropriate monitored tributary or ungaged watershed (as listed in Table 33) using a geographic information system (GIS) procedure with the assistance of the Vermont Center for Geographic Information.

Table 32. **Cost and assumed phosphorus reduction effectiveness of agricultural nonpoint source controls in the Lake Champlain Basin, from U.S. Natural Resources Conservation Service models. Costs are total capital and annual operation and maintenance costs, expressed on a present value basis (30 years at 5% discount rate). "NA" indicates data not available and treatment not considered.**

Treatment	Cost ($/au)	Phosphorus Reduction (kg/au/yr)
Milkhouse effluent treatment	81	0.091
Waste utilization	334	0.136
Barnyard runoff treatment	130	0.227
Erosion control	341	0.363
Grazing management	45	0.227
Nutrient management	108	0.091
Riparian zone management	NA	NA

A list of the maximum potential agricultural phosphorus reductions and costs for each tributary and ungaged area is given in Table 33. The maximum phosphorus reduction potential and costs are grouped by state and by lake segment watershed in Table 34.

Data for Quebec farms were not available and are not included in Tables 33 and 34. For the purpose of the load reduction optimization procedure discussed below, it was assumed that the potential agricultural phosphorus reduction in the Quebec portion of the Missisquoi Bay watershed was approximately equal to the potential reduction in the Vermont portion (20 mt/yr), since about half of the watershed of Missisquoi Bay is in Quebec. Costs of agricultural nonpoint source controls in Quebec were assumed to be equal to the values given in Tables 33 and 34.

Tables 33 and 34 show the costs and phosphorus reductions expected from full implementation of the Vermont AAP rules in each watershed. The Vermont AAP data in Tables 33 and 34 were derived by applying the U.S. Natural Resources Conservation Service Vermont agricultural phosphorus runoff models for treatments at all farms where such practices do not now exist, but would be required under the Vermont AAP rules. Phosphorus reductions from AAP implementation were estimated to be 50% of the maximum potential reduction obtainable from waste utilization treatment only, or 0.068 kg/au/yr. Costs for AAP implementation were estimated to be $58/au (present value basis).

Tables 33 and 34 also show the phosphorus load reductions estimated from recently implemented BMPs in Vermont and New York. Agricultural BMPs implemented since 1987 were assumed to have produced their water quality effects after the 1990-1991 monitoring period for this study. The U.S. Natural Resources Conservation Service used the values derived from the Vermont agricultural phosphorus runoff models to estimate the reductions that should be credited for these recently implemented BMPs, with no future cost, as part of the load reduction modeling procedure.

The data shown in Tables 33 and 34 indicate that recently implemented BMPs have reduced phosphorus loading to Lake Champlain by 29 mt/yr, relative to the 1991 base year nonpoint source loading rate of 443 mt/yr (Table 28, excluding direct precipitation). The remaining maximum potential future phosphorus reduction from agricultural sources is 101 mt/yr, including 75 mt/yr in Vermont, and 26 mt/yr in New York. An additional, but unknown, load reduction potential exists in the Quebec portion of the Missisquoi Bay watershed. Implementation of the Vermont AAP rules is expected to produce a 7 mt/yr reduction distributed among the watersheds as indicated in Tables 33 and 34.

The load reduction modeling analyses described below focused on the portion of the future potential reductions that could be targeted using the minimum-cost optimization procedure (i.e. excluding the Vermont AAP reductions which are mandatory throughout Vermont). The average unit cost ($millions/mt-yr) of these future targeted reductions are listed for each watershed in Tables 33 and 34. These average unit costs for targeted BMPs represent the cost to implement all the potentially available practices in each watershed (excluding the AAPs), divided by the phosphorus load reduction expected from their implementation. The Lake Champlain Basin average unit cost for future targeted agricultural BMP implementation is about $1.1 million/mt-yr. The relative magnitude of these unit costs between watersheds was used by the optimization procedure to produce watershed loading targets representing the minimum-cost alternative.

Table 33. Potential phosphorus reduction and cost for implementation of agricultural nonpoint source control practices in the Lake Champlain Basin, grouped by tributary watershed. Costs are total capital and annual operation and maintenance costs, expressed on a present value basis (30 years at 5% discount rate). Data for Quebec farms were not available and are not included in this table.

Lake Segment	Tributary	Maximum Potential Future Reduction (mt/yr)	Expected Vermont AAP Reduction (mt/yr)	Recent BMP Reduction (mt/yr)	Cost ($) of Maximum Future Reduction	Cost ($) of Vermont AAPs	Unit Cost of Future Targeted Reduction ($ millions/mt-yr)
1. South Lake B NY	Mettawee NY	7.38	0.00	0.000	9,706,414	0	1.315
1. South Lake B NY	Mt. Hope	0.09	0.00	0.000	129,903	0	1.465
1. South Lake B NY	Poultney NY	0.41	0.00	0.000	576,810	0	1.407
1. South Lake B NY	Ungaged NY 1	0.17	0.00	0.000	218,192	0	1.252
1. South Lake B VT	Mettawee VT	3.46	0.43	0.000	3,909,650	368,010	1.169
1. South Lake B VT	Poultney VT	2.25	0.27	0.000	2,525,256	227,592	1.160
1. South Lake B VT	Ungaged VT 1	0.43	0.04	0.288	442,976	34,584	1.057
2. South Lake A NY	LaChute	0.18	0.00	0.000	228,468	0	1.286
2. South Lake A NY	Mill (Putnam Sta.)	0.26	0.00	0.000	318,152	0	1.243
2. South Lake A NY	Putnam	0.01	0.00	0.282	7,062	0	0.778
2. South Lake A NY	Ungaged NY 2	0.70	0.00	0.058	793,372	0	1.134
2. South Lake A VT	East	0.87	0.08	0.588	904,525	70,619	1.057
2. South Lake A VT	Ungaged VT 2	0.96	0.09	0.651	1,001,249	78,170	1.057
3. Port Henry NY	Hoisington	0.04	0.00	0.072	41,414	0	1.024
3. Port Henry NY	Mill (Port Henry)	0.06	0.00	0.003	69,419	0	1.180
3. Port Henry NY	Ungaged NY 3	0.34	0.00	0.685	342,238	0	0.992
3. Port Henry VT	Ungaged VT 3	0.33	0.03	0.225	346,864	27,081	1.057
4. Otter Creek NY	Ungaged NY 4	0.02	0.00	0.028	15,847	0	1.024
4. Otter Creek VT	Lewis	1.83	0.10	0.000	1,660,493	88,682	0.911
4. Otter Creek VT	Little Otter	3.09	0.30	0.000	3,302,087	258,859	1.092
4. Otter Creek VT	Otter	18.97	1.61	7.367	19,173,200	1,375,818	1.025
4. Otter Creek VT	Ungaged VT 4	0.20	0.02	0.088	216,934	18,680	1.128
5. Main Lake NY	Ausable	0.24	0.00	0.036	260,261	0	1.102
5. Main Lake NY	Bouquet	0.59	0.00	0.734	689,664	0	1.162
5. Main Lake NY	Highlands Forge	0.00	0.00	0.000	0	0	0.000
5. Main Lake NY	Little Ausable	1.15	0.00	0.250	1,482,483	0	1.291

Lake Segment	Tributary	Maximum Potential Future Reduction (mt/yr)	Expected Vermont AAP Reduction (mt/yr)	Recent BMP Reduction (mt/yr)	Cost ($) of Maximum Future Reduction	Cost ($) of Vermont AAPs	Unit Cost of Future Targeted Reduction ($ millions/mt-yr)
5. Main Lake NY	Salmon	0.37	0.00	0.000	441,016	0	1.191
5. Main Lake NY	Ungaged NY 5	0.29	0.00	0.026	356,156	0	1.238
5. Main Lake VT	Ungaged VT 5	0.49	0.05	0.254	522,613	43,052	1.088
5. Main Lake VT	Winooski	6.58	0.82	0.886	7,140,920	701,046	1.118
6. Shelburne Bay	LaPlatte	0.93	0.09	0.000	966,184	77,314	1.053
6. Shelburne Bay	Ungaged VT 6	0.01	0.00	0.008	16,575	1,488	1.153
7. Burlington Bay	Ungaged VT 7	0.00	0.00	0.000	0	0	0.000
8. Cumberland Bay	Saranac	0.68	0.00	0.035	870,914	0	1.286
8. Cumberland Bay	Ungaged NY 8	2.28	0.00	0.406	2,794,250	0	1.226
9. Malletts Bay	Indian	0.13	0.02	0.049	148,505	13,174	1.145
9. Malletts Bay	Lamoille	8.40	0.81	2.667	8,773,174	689,070	1.065
9. Malletts Bay	Malletts	0.33	0.04	0.122	369,348	32,766	1.145
9. Malletts Bay	Ungaged VT 9	0.18	0.02	0.071	191,164	16,035	1.102
10. Northeast Arm	Stone Bridge	0.26	0.02	0.307	267,679	20,173	1.039
10. Northeast Arm	Ungaged VT 10	2.23	0.18	1.442	2,196,718	153,564	0.996
11. St. Albans Bay	Mill	1.27	0.09	0.000	1,224,931	79,512	0.976
11. St. Albans Bay	Stevens	1.26	0.09	0.000	1,222,242	79,338	0.976
11. St. Albans Bay	Ungaged VT 11	0.24	0.02	0.018	232,651	15,219	0.978
12. Missisquoi Bay	Missisquoi	16.34	1.30	7.288	16,162,069	1,109,965	1.001
12. Missisquoi Bay	Pike	1.57	0.11	1.018	1,508,441	90,563	0.970
12. Missisquoi Bay	Rock	1.42	0.10	0.924	1,368,604	82,167	0.970
12. Missisquoi Bay	Ungaged VT 12	0.70	0.05	0.520	675,894	43,610	0.972
13. Isle LaMotte NY	Great Chazy	8.10	0.00	1.022	10,127,844	0	1.250
13. Isle LaMotte NY	Little Chazy	1.09	0.00	0.223	1,540,350	0	1.418
13. Isle LaMotte NY	Ungaged NY 13	1.46	0.00	0.087	2,021,832	0	1.381
13. Isle LaMotte VT	Ungaged VT 13	0.49	0.04	0.244	482,171	33,297	0.990
BASIN TOTAL		101.15	6.84	28.972	109,985,178	5,829,447	1.104

Table 34. Potential phosphorus reduction and cost for implementation of agricultural nonpoint source control practices in the Lake Champlain Basin, grouped by state and lake segment watershed. Costs are total capital and annual operation and maintenance costs, expressed on a present value basis (30 years at 5% discount rate). Data for Quebec farms were not available and are not included in this table.

Lake Segment Watershed	Maximum Potential Future Reduction (mt/yr)	Expected Vermont AAP Reduction (mt/yr)	Recent BMP Reduction (mt/yr)	Cost ($) of Maximum Future Reduction	Cost ($) of Vermont AAPs	Unit Cost of Future Targeted Reduction ($ millions/mt-yr)
Vermont						
1. South Lake B	6.14	0.74	0.29	6,877,882	630,186	1.158
2. South Lake A	1.84	0.17	1.24	1,905,774	148,789	1.057
3. Port Henry	0.33	0.03	0.23	346,864	27,081	1.057
4. Otter Creek	24.09	2.04	7.46	24,352,715	1,742,039	1.026
5. Main Lake	7.07	0.87	1.14	7,663,533	744,098	1.116
6. Shelburne Bay	0.95	0.09	0.01	982,759	78,802	1.054
7. Burlington Bay	0.00	0.00	0.00	0	0	0.000
9. Malletts Bay	9.04	0.88	2.91	9,482,192	751,045	1.070
10. Northeast Arm	2.49	0.20	1.75	2,464,397	173,736	1.001
11. St. Albans Bay	2.77	0.20	0.02	2,679,824	174,069	0.976
12. Missisquoi Bay	20.03	1.56	9.75	19,715,007	1,326,305	0.995
13. Isle LaMotte	0.49	0.04	0.24	482,171	33,297	0.990
Vermont Total	75.25	6.84	25.03	76,953,118	5,829,447	1.040
New York						
1. South Lake B	8.05	0.00	0.00	10,631,319	0	1.320
2. South Lake A	1.14	0.00	0.34	1,347,053	0	1.179
3. Port Henry	0.44	0.00	0.76	453,071	0	1.020
4. Otter Creek	0.02	0.00	0.03	15,847	0	1.024
5. Main Lake	2.64	0.00	1.05	3,229,581	0	1.225
8. Cumberland Bay	2.96	0.00	0.44	3,665,164	0	1.240
13. Isle LaMotte	10.66	0.00	1.33	13,690,025	0	1.285
New York Total	25.90	0.00	3.95	33,032,059	0	1.275
BASIN TOTAL	101.15	6.84	28.97	109,985,178	5,829,447	1.104

Urban Controls

Information on the cost and expected phosphorus reduction effectiveness of nonpoint source phosphorus controls on urban land was obtained and modified from Holmes and Artuso (1995), based on published reports of programs implemented in other parts of the country. Two categories of urban BMPs were considered by Holmes and Artuso (1995). Non-structural urban BMPs or minor capital improvements such as more frequent street sweeping, installation of sediment traps in catch basins, and public education efforts were assumed to cost $45 to $180 annually per hectare of urban land treated, with a phosphorus load reduction effectiveness of 10%. Structural urban BMPs such as construction of detention basins or stormwater treatment facilities were assumed to have a capital cost, including land acquisition, of $5,638 to $13,252 per hectare treated, with an annual operating cost of 5% of the capital cost. A phosphorus reduction effectiveness of 50% was assumed for structural urban BMPs.

Costs of urban BMPs were expressed on a present value basis (5% discount rate over a 30-year period) for the optimization analysis, consistent with the agricultural cost information. Urban BMP costs were assumed to increase linearly between the minimum and maximum values given above as a function of the proportion (0-100%) of the total urban land area treated in each watershed, as would approximately be the case if the least expensive controls were implemented first. The total urban land area in each lake segment watershed used in this analysis are listed in Table 35. The urban land areas given in Table 35 were modified from the mid-1970s values used by Holmes and Artuso (1995) in proportion to the population increase estimated for each lake segment watershed between 1970 and 1990, using U.S. Census data.

Current phosphorus loadings from urban land in each lake segment watershed (Table 35) were calculated by Holmes and Artuso (1995) using land use information in Budd and Meals (1994) with the measured nonpoint source loads provided by this study. Percent phosphorus reductions from urban BMP implementation relative to the total urban loads given in Table 35 were estimated in the optimization analysis from the proportion of urban land treated in each watershed with structural and non-structural controls at the assumed treatment effectiveness rates.

There were several types of “urban” nonpoint source control practices that were not considered in this analysis, due to a lack of cost-effectiveness information. These include practices suitable for rural residential areas and other lower density developments. Examples of such practices include protection of riparian zones with buffer areas, limitations on the extent of impervious surfaces in a watershed, better back road maintenance, and erosion control at construction sites. Better cost-effectiveness information on these practices should be obtained in the Lake Champlain Basin so that they can be considered as more specific phosphorus reduction plans are prepared for each targeted watershed.

Minimum-Cost Optimization Procedure

The lake phosphorus model developed and calibrated using the BATHTUB program was transferred to a spreadsheet format in order to conduct a minimum-cost optimization analysis, following approaches used by Chapra et al. (1983) and Holmes and Artuso (1995). Numeric solution techniques provided by the spreadsheet program (Novell, Inc., 1994) were used to find the least-cost combination of nonpoint source reductions in each lake segment watershed that would achieve the in-lake phosphorus concentration criteria (Table 2).

Table 35. **Urban land areas and current urban nonpoint source loading estimates used in the phosphorus load reduction targeting procedure.**

Lake Segment Watershed	Urban Area (ha)	Current Urban Load (mt/yr)
South Lake B, NY	2,565	3.24
South Lake B, VT	2,909	3.59
South Lake A, NY	1,963	0.84
South Lake A, VT	26	0.01
Port Henry, NY	908	0.69
Port Henry, VT	11	0.01
Otter Creek, NY	0	0.00
Otter Creek, VT	6,714	5.71
Main Lake, NY	8,049	9.97
Main Lake, VT	15,429	17.52
Shelburne Bay, VT	2,538	1.49
Burlington Bay, VT	1,704	0.23
Cumberland Bay, NY	6,252	2.81
Malletts Bay, VT	6,330	4.49
Northeast Arm, VT	1,447	0.44
St. Albans Bay, VT	1,773	2.09
Missisquoi Bay, VT/Que.	3,181	19.08
Isle LaMotte, NY	2,265	2.36
Isle LaMotte, VT	243	0.20

The model mass balance equations for each lake segment (equation 1) were written into the spreadsheet version of the model using the flow routing scheme shown in Figure 25 and the calibrated values for the exchange rates and phosphorus sedimentation coefficients. The spreadsheet program used iterative solution techniques to solve the series of non-linear equations for the phosphorus concentration in each lake segment.

Table 36 lists the values for the model terms for each lake segment used in the analysis. Point source loads were assumed to be at their target values, from Table 30. Watershed nonpoint source loads were initially set to their 1991 base year values and then varied during the optimization procedure. Other phosphorus mass transfer terms such as direct precipitation and intake withdrawals were held at their 1991 levels. The net internal loading rate to St. Albans Bay was set to zero, as discussed above. Net flows from gaged and ungaged watershed runoff, direct precipitation, evaporation, and water withdrawals were routed cumulatively through the lake segments to provide the values given in Table 36 for advective flows to downstream segments.

Before the optimization procedure was initiated, the load reductions attributed to recently implemented agricultural BMPs (from Table 34) were subtracted from the 1991 base year nonpoint source loads for each lake segment. No future cost was accumulated in the optimization procedure for this prior reduction. The reductions expected from non-targeted implementation of the Vermont AAP rules (Table 34) were also subtracted from the remaining nonpoint source loads, and the cost of AAP implementation was included in the total cost estimate. The optimization procedure then varied the extent of further agricultural and urban nonpoint source load reductions in each segment watershed until the optimization parameters described below were satisfied.

The spreadsheet load reduction targeting procedure required the specification of optimization goals, variables, and constraints. The optimization goal was to minimize the total (present value) cost of nonpoint source load reductions in the Lake Champlain Basin. Unit costs ($ per mt/yr) of agricultural load reduction in each watershed were derived from the values given in Table 34. Costs of urban nonpoint source controls were modified from Holmes and Artuso (1995), as described above.

The variables in the procedure were the extent of future agricultural and urban nonpoint source load reductions targeted to each lake segment watershed and each state. The spreadsheet optimization procedure iteratively varied the agricultural loads and the percent of the urban land treated with BMPs in each watershed until the minimum-cost goal was achieved within the constraints specified.

The key constraint on the optimization procedure was that the model-predicted phosphorus concentrations in each lake segment must be less than or equal to the criteria values. Additional constraints were specified such that the future agricultural load reductions (including Vermont AAP implementation) in each watershed must be less than or equal to the maximum potential reduction in each watershed, as given in Table 34. Other constraints in the procedure limited the agricultural and urban loads to their present values or less within each lake segment watershed (i.e. nonpoint source loading increases were not allowed).

The in-lake criteria used as constraints in the optimization procedure were modified from the values given in Table 2 for two lake segments. Compliance with the 25 μg/l criterion for the South Lake B segment was not required in the analysis because it was found that this value could not be attained with any realistic combination of point and nonpoint source reduction measures. The 54 μg/l criterion value specified in the Vermont Water Quality Standards was used for the South Lake B segment instead. The criterion for the Missisquoi Bay segment was modified slightly upward to 27

Table 36. **Values for the terms used in the spreadsheet version of the lake phosphorus model and the load reduction optimization procedure.**

Lake Segment	In-Lake Phosphorus Criterion (mg/l)	VT Target Point Load (mt/yr)	NY Target Point Load (mt/yr)	VT 1991 Watershed Nonpoint Load (mt/yr)	NY 1991 Watershed Nonpoint Load (mt/yr)	Other Loads[1] (mt/yr)	Total Load (mt/yr)	Advective Flow to Downstream Segment (hm^3/yr)	Exchange Flow with Downstream Segment (hm^3/yr)	Phosphorus Sedimentation Coefficient ($m^3/g\text{-}yr$)	Segment Volume (km^3)
1. South Lake B	0.054[4]	1.5	1.9	24.8	24.3	0.1	52.6	830	712	100	0.008
2. South Lake A	0.025	0.1	7.4	2.4	3.5	-0.0	13.3	1,263	1,259	100	0.125
3. Port Henry	0.014	0.0	0.7	0.4	2.6	1.0	4.7	1,360	13,998	100	1.463
4. Otter Creek	0.014	7.1	0.0	58.9	0.1	0.4	66.5	2,643	49,427	100	0.955
5. Main Lake	0.010	18.3	4.3	60.3	31.8	5.6	120.3	7,239	8,861	100	16.787
6. Shelburne Bay	0.014	0.9	0.0	11.1	0.0	-0.0	12.1	58	4,816	100	0.140
7. Burlington Bay	0.014	2.8	0.0	0.3	0.0	0.1	3.2	7	2,986	100	0.063
8. Cumberland Bay	0.014	0.0	17.2	0.0	8.8	0.1	26.1	828	8,672	100	0.063
9. Malletts Bay	0.010	2.6	0.0	29.8	0.0	0.7	33.2	988 (188[2])	272 (52[2])	400	0.722
10. Northeast Arm	0.014	0.0	0.0	3.2	0.0	3.4	6.5	2,019	1,968	100	3.380
11. St. Albans Bay	0.017	2.4	0.0	7.2	0.0	0.1	9.8	43	1,844	0	0.023
12. Missisquoi Bay[3]	0.027[4]	5.3	0.0	151.9	0.0	1.2	158.4	1,720	297	400	0.205
13. Isle LaMotte	0.014	0.0	2.0	0.6	20.9	2.5	25.9	9,672		100	1.892
TOTAL		41.1	33.5	350.8	91.9	15.2	532.5				25.826

[1] Includes net sum of direct precipitation, water withdrawals, and internal loading.
[2] Value for the northern interface of Malletts Bay with the Northeast Arm.
[3] Includes loads from Quebec.
[4] Modified from the value given in Table 2.

µg/l for similar reasons. The phosphorus concentration criteria for each lake segment used as constraints in the optimization procedure are given in Table 36.

The results of the nonpoint source load reduction optimization procedure are given in Table 37. No urban sources were targeted for reduction because of their higher cost relative to agricultural controls. The potential reductions available from agricultural sources, in combination with the point source loading limits specified in the procedure, were sufficient to attain the in-lake criteria without seeking more expensive reductions from urban areas. However, as noted above, the model did not consider some possibly less expensive control measures for rural residential or suburban land. These "urban" sources should be considered as part of future phosphorus reduction implementation plans for individual watersheds. The agricultural nonpoint source load reduction targets given in Table 37 should be treated as total nonpoint source reduction targets, and not exclusively the responsibility of the agricultural community, for the present purpose of developing a basin-wide phosphorus reduction strategy.

As shown in Table 37, a total nonpoint source reduction of 78.2 mt/yr relative to the 1991 base year value will result in attainment of the in-lake criteria at minimum basin-wide cost. This reduction includes 29.0 mt/yr from recently implemented BMPs, 6.8 mt/yr expected from compliance with the Vermont AAP rules, and 42.4 mt/yr from future reductions optimally targeted to specific watersheds. Most of the nonpoint reduction (73.1 mt/yr) must come from sources in Vermont and Quebec, including 47.5 mt/yr in the Missisquoi Bay watershed alone.

The future nonpoint source reductions indicated by the procedure range from 0-100% of the maximum potential agricultural reduction among the various watersheds (from Table 34). Overall, the future reductions shown in Table 37 (including Vermont AAP implementation) are 41% of the maximum potential for the basin as a whole.

The total (present value) cost to attain the nonpoint source load reductions indicated in Table 37 is $49.0 million, including $47.7 in Vermont and Quebec and $1.4 million in New York. The majority of the cost ($37.3 million) is for implementation measures in the Missisquoi Bay watershed. The $49.0 million cost represents the minimum value resulting from the optimization procedure.

Load Reduction Strategy Results

Summary of Allowable Loads and Load Reduction Targets

The load reduction targeting strategy for the Lake Champlain Basin resulting from the procedure described above is summarized in Table 38. Table 38 provides a basis for establishing allowable phosphorus loading rates from each state and each lake segment watershed. A total point and nonpoint source phosphorus load reduction of 192 mt/yr relative to the 1991 base year value will be required to attain the in-lake criteria, including 162 mt/yr from sources in Vermont and Quebec and 31 mt/yr from New York sources. Table 38 also shows the load reduction requirements relative to the estimated 1995 loading rates, taking into account the reductions accomplished since 1991. Relative to the 1995 values, the future load reduction requirement totals 57 mt/yr, including 56 mt/yr in Vermont and Quebec and 1 mt/yr in New York.

Predicted Future Phosphorus Concentrations

Model predictions using mean values for all model input terms without error analysis were used to determine compliance with the criteria values during the spreadsheet based load reduction

Table 37. Results of nonpoint source load reduction optimization procedure, showing targeted agricultural load reductions and costs.

Lake Segment Watershed	Recent BMP Reduction (mt/yr)	Vermont AAP Reduction (mt/yr)	Future Targeted BMP Reduction (mt/yr)	Total Reduction (mt/yr)	Percent of Maximum Potential Reduction	Total Present Value Cost ($ millions)
Vermont/Quebec						
South Lake B	0.29	0.74	4.51	5.54	86%	5.8
South Lake A	1.24	0.17	0.40	1.81	31%	0.6
Port Henry	0.23	0.03	0.00	0.26	10%	0.0
Otter Creek	7.45	2.04	0.21	9.71	9%	2.0
Main Lake	1.14	0.87	0.00	2.01	12%	0.7
Shelburne Bay	0.01	0.09	0.00	0.10	10%	0.1
Burlington Bay	0.00	0.00	0.00	0.00	0%	0.0
Malletts Bay	2.91	0.88	0.00	3.79	10%	0.8
Northeast Arm	1.75	0.20	0.00	1.95	8%	0.2
St. Albans Bay	0.02	0.20	0.00	0.22	7%	0.2
Missisquoi Bay	9.75	1.56	36.15	47.46	94%	37.3
Isle LaMotte	0.24	0.04	0.00	0.28	8%	0.0
Vermont/Quebec Total	25.02	6.84	41.27	73.13	51%	47.7
New York						
South Lake B	0.00	0.00	0.00	0.00	0%	0.0
South Lake A	0.34	0.00	1.14	1.48	100%	1.3
Port Henry	0.76	0.00	0.00	0.76	0%	0.0
Otter Creek	0.03	0.00	0.02	0.04	100%	0.0
Main Lake	1.05	0.00	0.00	1.05	0%	0.0
Cumberland Bay	0.44	0.00	0.00	0.44	0%	0.0
Isle LaMotte	1.33	0.00	0.00	1.33	0%	0.0
New York Total	3.95	0.00	1.16	5.11	4%	1.4
BASIN TOTAL	28.97	6.84	42.43	78.24	41%	49.0

Table 38. Summary of Lake Champlain phosphorus load reduction procedure results, showing 1991 and 1995 loads (mt/yr), allowable target loads, and load reductions required to attain the in-lake criteria at minimum basin-wide cost.

	Point Sources				Nonpoint Sources				Total				
Lake Segment	1991 Load	1995 Load	Allowable Load	Reduction from 1991	1991 Load	1995 Load	Allowable Load	Reduction from 1991	1991 Load	1995 Load	Allowable Load	Reduction from 1991	Reduction from 1995
Vermont													
South Lake B	3.2	3.1	1.5	1.7	24.8	24.5	19.3	5.5	28.0	27.6	20.8	7.2	6.8
South Lake A	0.1	0.1	0.1	-0.0	2.4	1.1	0.6	1.8	2.4	1.2	0.6	1.8	0.5
Port Henry	0.0	0.0	0.0	0.0	0.4	0.2	0.1	0.3	0.4	0.2	0.1	0.3	0.0
Otter Creek	62.8	9.8	7.1	55.7	58.9	51.4	49.2	9.7	121.7	61.2	56.3	65.4	4.9
Main Lake	27.7	21.6	18.3	9.4	60.3	59.1	58.3	2.0	88.0	80.7	76.6	11.4	4.1
Shelburne Bay	5.3	0.7	0.9	4.4	11.1	11.1	11.0	0.1	16.4	11.8	12.0	4.5	-0.1
Burlington Bay	11.2	2.2	2.8	8.4	0.3	0.3	0.3	0.0	11.5	2.5	3.1	8.4	-0.6
Malletts Bay	3.1	2.7	2.6	0.5	29.8	26.9	26.1	3.8	32.9	29.7	28.6	4.3	1.0
Northeast Arm	0.0	0.0	0.0	0.0	3.2	1.4	1.2	2.0	3.2	1.4	1.2	2.0	0.2
St. Albans Bay	0.8	1.6	2.4	-1.6	7.2	7.2	7.0	0.2	8.0	8.9	9.5	-1.4	-0.6
Missisquoi Bay	15.4	6.9	5.3	10.1	151.9	142.1	104.4	47.5	167.3	149.1	109.7	57.6	39.3
Isle LaMotte	0.0	0.0	0.0	-0.0	0.6	0.3	0.3	0.3	0.6	0.3	0.3	0.3	0.0
Vermont Total	129.6	48.7	41.1	88.5	350.8	325.8	277.7	73.1	480.4	374.5	318.8	161.6	55.7
New York													
South Lake B	3.9	2.7	1.9	2.0	24.3	24.3	24.3	0.0	28.2	27.0	26.2	2.0	0.8
South Lake A	9.6	6.9	7.4	2.2	3.5	3.1	2.0	1.5	13.1	10.1	9.4	3.7	0.7
Port Henry	1.8	2.7	0.7	1.1	2.6	1.8	1.8	0.8	4.3	4.5	2.5	1.8	2.0
Otter Creek	0.0	0.0	0.0	0.0	0.1	0.1	0.0	0.0	0.1	0.1	0.0	0.0	0.0
Main Lake	7.1	6.7	4.3	2.8	31.8	30.8	30.8	1.0	38.9	37.5	35.1	3.8	2.4
Cumberland Bay	29.2	11.8	17.2	12.0	8.8	8.3	8.3	0.4	38.0	20.2	25.5	12.4	-5.3
Isle LaMotte	7.4	2.5	2.0	5.4	20.9	19.5	19.5	1.3	28.3	22.0	21.5	6.8	0.5
New York Total	59.0	33.3	33.5	25.5	91.9	88.0	86.8	5.1	150.9	121.3	120.3	30.6	1.0
BASIN TOTAL	188.5	82.0	74.6	113.9	442.7	413.8	364.5	78.2	631.3	495.7	439.1	192.2	56.6

optimization procedure. With this approach, the confidence level for attainment of the phosphorus criteria is approximately 50% for lake segments currently exceeding their criterion.

An alternative procedure was considered in which compliance was required at the upper (e.g. 90%) level of the distribution of predicted segment phosphorus concentrations, based on the BATHTUB program error analysis. It was found that attainment of the in-lake criteria at the 90% confidence level in all lake segments was not a practical possibility for any reasonable combination of nonpoint source loading reductions, and this requirement was therefore not included in the procedure.

The model-predicted phosphorus concentrations in each lake segment resulting from the target loading values given in Table 38 are compared in Figure 32 with the in-lake criteria. The 95% confidence limits for the predicted phosphorus distributions shown in Figure 32 were calculated from BATHTUB program error analysis procedures, and indicate the degree of uncertainty in the predictions. It was assumed for this purpose that the error terms (C.V. values) for the tributary concentrations given in Table 24 would remain the same as loadings were reduced. Figure 32 indicates that even if the loading targets given in Table 38 are attained, there is a significant possibility that the in-lake criteria will still be exceeded in some lake segments, although the majority of the lake segments would be expected to come into compliance with their phosphorus criteria.

1996 Lake Champlain Phosphorus Load Reduction Agreement

In June 1996, the States of Vermont and New York and the U.S. Environmental Protection Agency negotiated an agreement on a process for developing phosphorus loading targets for Lake Champlain. The Lake Champlain Management Conference endorsed this agreement and incorporated it into the draft comprehensive basin plan titled *Opportunities for Action, An Evolving Plan for the Future of the Lake Champlain Basin* (Lake Champlain Management Conference, 1996).

The agreement established preliminary total phosphorus loading targets for each state and lake segment watershed, consistent with the load reduction targeting procedure described above and listed in Table 38. The agreement committed each state to achieving the allowable phosphorus loads for each lake segment within the next 20 years, and to making 25% progress toward the loading targets within the next five years. Each state retains the flexibility to adjust point and nonpoint source loading values between lake segment watersheds, provided that the lake phosphorus model developed by this study is used to ensure that the in-lake criteria will be attained without affecting the other state's loading targets.

As an initial implementation output under the June 1996 agreement, both states prepared Lake Champlain Phosphorus Reduction Plans in October 1996. These plans list specific point and nonpoint source phosphorus control measures that will be taken to accomplish the load reduction goals (Vermont Agency of Natural Resources, 1996; New York State Department of Environmental Conservation, 1996).

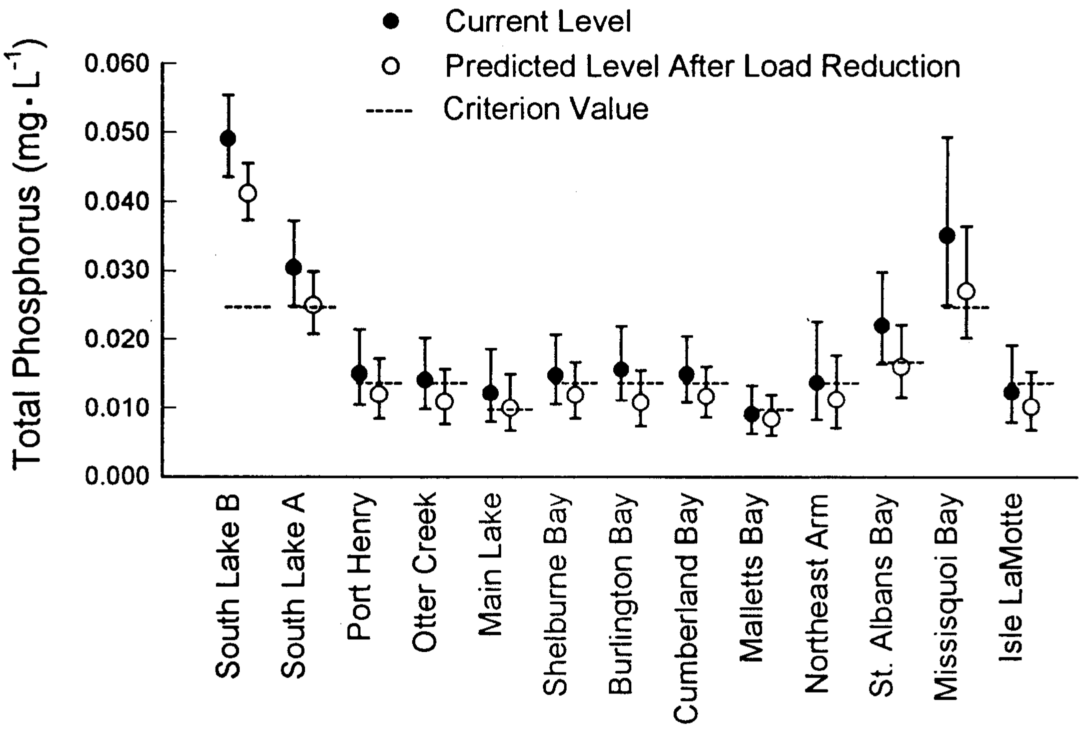

Figure 32. Phosphorus load reduction modeling results. Predicted total phosphorus concentrations in Lake Champlain segments following targeted loading reductions are compared with the 1991 measured levels and the in-lake phosphorus criteria values. Error bars show 95% confidence intervals for the 1991 existing and predicted mean phosphorus concentrations.

CONCLUSIONS AND RECOMMENDATIONS

The ultimate purpose of this study was to recommend basin-wide point and nonpoint source phosphorus load reduction targets for Lake Champlain that, if implemented, will attain the in-lake water quality criteria endorsed by Vermont, New York, and Quebec in the 1993 Lake Champlain Water Quality Agreement. The database and lake phosphorus model developed for this study are suitable for this purpose. Water and chloride input-output budgets derived from the data balanced within close tolerances, indicating that the major mass balance model terms were estimated accurately. Model terms estimated through calibration procedures received independent confirmation in most respects. The calibrated exchange flow rates between lake segments were consistent with lake morphometry. The phosphorus model sedimentation rates were consistent with independently derived values from a national reservoir data set for most segments of Lake Champlain. A model error analysis was conducted that produced statistical estimates of the uncertainty of all model predictions.

The lake phosphorus mass balance model developed by this study was used with a minimum-cost optimization procedure to establish phosphorus loading targets for each state and each lake segment watershed. These loading targets were accepted by the States of Vermont and New York and the Lake Champlain Management Conference in its 1996 Comprehensive Plan titled *Opportunities for Action, An Evolving Plan for the Future of the Lake Champlain Basin*. The plan commits the states to achieving the target loads within 20 years, contingent on the availability of state and federal funding.

The lake modeling analysis indicated that a basin-wide total allowable phosphorus loading rate of 439 mt/yr, distributed among the lake segment watersheds as indicated in Table 38, will achieve the in-lake criteria at minimum cost. This represents a 192 mt/yr (30%) reduction from the 1991 base year watershed loading rate of 631 mt/yr, and a 57 mt/yr (11%) reduction from the estimated 1995 loading rate of 496 mt/yr. The criteria endpoints for the South Lake B and Missisquoi Bay segments were modified in the analysis from the values given in Table 2 so that the loading targets would be practically attainable.

The steady-state, multiple-segment, mixed reactor modeling approach used in this study is appropriate for the purpose of identifying basin-wide phosphorus load reductions required to attain the in-lake phosphorus criteria for each lake segment, which are expressed on a spatial and seasonal average basis. However, the data and lake model were not designed to analyze the impacts of individual phosphorus discharges on a more local scale within a lake segment, or to simulate seasonal phosphorus dynamics.

Recommendations

1. **The States of Vermont and New York, with assistance from the U.S. Environmental Protection Agency and the U.S. Department of Agriculture, should proceed to implement point and nonpoint source phosphorus load reductions in the Lake Champlain Basin according to the June 1996 phosphorus reduction agreement (Lake Champlain Management Conference, 1996) and the October 1996 phosphorus reduction implementation plans submitted by each state.**

2. **The States of Vermont and New York should maintain the database and modeling capability on Lake Champlain developed by this study so that the model can be used to analyze alternative in-lake phosphorus criteria or different phosphorus load reduction strategies that might be considered in the future. The states retain flexibility under the 1996 Lake Champlain Phosphorus Reduction Agreement to adjust the loading targets, provided that the model is used to ensure that the in-lake criteria will be achieved without affecting the other state's obligations. The lake model should be refined and enhanced as additional research information becomes available for Lake**

Champlain on hydrodynamic processes, internal phosphorus dynamics, in-stream phosphorus transport and transformation, and lake ecosystem response to phosphorus loading.

3. The State of Vermont and the Province of Quebec should work cooperatively to attain the phosphorus load reductions required for Missisquoi Bay. The Lake Champlain Steering Committee established by the Memorandum of Understanding on Environmental Cooperation on the Management of Lake Champlain should designate a Missisquoi Bay Phosphorus Management Task Force. The purpose of the Task Force should be to conduct studies and discussions in order to reach technical and political agreement on the amount of phosphorus load reductions to be obtained from each jurisdiction within the Missisquoi Bay watershed, and on the means for implementation of the reductions.

4. Missisquoi Bay is the lake segment where the greatest nonpoint source phosphorus load reductions will be required. However, the model phosphorus sedimentation term used for Missisquoi Bay is relatively unconfirmed because of the additional calibration adjustment that was necessary to accurately simulate the observed phosphorus levels in the bay. For this reason, Missisquoi Bay should be a high priority region of Lake Champlain in which to conduct additional research on internal phosphorus processes in the lake. This research should be used to confirm or modify mass balance model terms representing internal phosphorus processes, and refine the load reduction recommendations for Missisquoi Bay if necessary.

5. St. Albans Bay is a lake segment where strong spatial concentration gradients and seasonal phosphorus dynamics have a major effect on water quality conditions. A more detailed modeling analysis of St. Albans Bay by Smeltzer *et al.* (1994) showed that excessive accumulations of phosphorus in the inner bay, especially during August when wind induced exchange mixing is at a minimum, may continue to cause occasional summer algae blooms even if the 17 Fg/l water quality criterion (Table 2) is attained on an annual average basis in the center of the bay. The major immediate source of phosphorus to St. Albans Bay during the August problem period is internal loading from bay and wetland sediments. The load reduction procedure for the present study was conducted with the assumption that internal phosphorus loading rates in St. Albans Bay will decline over time as expected (Martin *et al.* 1994) . However, nonpoint source phosphorus loading from St. Albans Bay tributaries remain high (Figure 29) and may contribute to long-term sediment phosphorus accumulation and recycling back to the water column during the critical summer season. Reduction in nonpoint source phosphorus loading to St. Albans Bay should therefore be considered as part of an overall phosphorus management strategy for Lake Champlain.

6. The lake modeling and load reduction analysis indicated that the 25 Fg/l phosphorus criterion for the South Lake B segment (Table 2) is probably not attainable with any practical combination of point and nonpoint source loading reductions within that segment watershed. The criterion for the South Lake B segment should therefore be re-examined during future revisions of the New York, Quebec, and Vermont Water Quality Agreement on Lake Champlain, based on further research on phosphorus sources and impacts in the South Lake region. The mass balance model should be extended to include a time-dependent analysis of all major phosphorus processes in this particularly dynamic region of Lake Champlain.

7. The States of Vermont and New York should continue to operate the Long-Term Water Quality and Biological Monitoring Program for Lake Champlain, which began in 1992. This program uses sampling locations and methods comparable to those employed for the present study, and expands the number of parameters monitored. The long-term monitoring project is designed to

assess progress toward attainment of the watershed load reduction targets and the in-lake phosphorus criteria over a period of several years or decades.

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