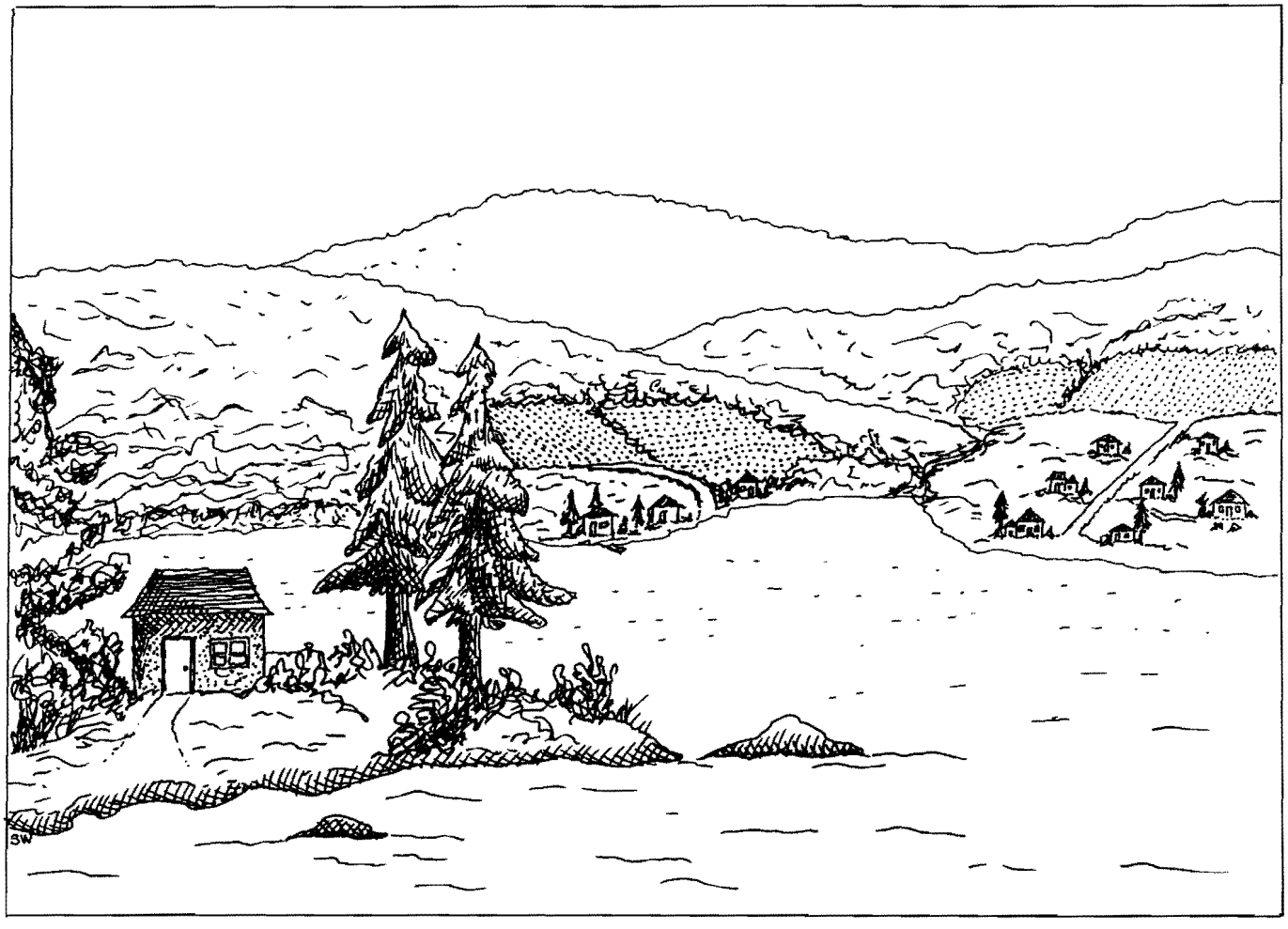
LAKE IROQUOIS
DIAGNOSTIC - FEASIBILITY STUDY
1982 - 1985
FINAL REPORT



STATE OF VERMONT

AGENCY OF ENVIRONMENTAL CONSERVATION

DEPARTMENT OF WATER RESOURCES

AND ENVIRONMENTAL ENGINEERING

WATER QUALITY DIVISION

MONTPELIER, VERMONT

Lake Iroquois
Diagnostic-Feasibility Study

Final Report - June 1985

as authorized by the Revised
Regulations for Section 314 of the
Clean Water Act of 1977, P.L. 95-217

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Several lake residents assisted with the study. Mary Madore maintained stage records on the lake and the outlet, monitored precipitation, and researched historical information on Lake Iroquois. Henry Gulla and Ginger Isham also monitored precipitation in the watershed. Many other Lake Iroquois residents cooperated with the study by permitting the installation of weirs and wells on their property.

Other contributions are acknowledged at appropriate points within the text.

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INTRODUCTION

Eutrophication is a natural process that occurs in all lakes starting from the time the lake is formed. Man's activities, however, can greatly accelerate this process of nutrient enrichment and severely impact the suitability of a lake for various uses. Phosphorus is commonly the nutrient in shortest supply in lakes. Increased phosphorus availability promotes the growth of algae and aquatic plants. This can result in reduced water clarity, increased oxygen depletion, limited recreational opportunities, and a generally reduced aesthetic appeal. Activities such as land clearing, road ditching, landscaping and stream channelization can greatly increase surface runoff and the amount of phosphorus delivered to the lake. Agricultural runoff can also be high in phosphorus due to erosion from cultivated areas, the use of fertilizers, and animal wastes. The eutrophication impact of greater phosphorus loads to a lake can be magnified when this phosphorus is internally recycled.

Lake Iroquois was selected to undergo a diagnostic-feasibility study primarily as a result of its ranking in the Vermont Lake Classification Survey (Vermont Department of Water Resources and Environmental Engineering, 1980). Lake Iroquois was one of 14 lakes which possessed a combination of high spring phosphorus concentrations, a large percentage of the watershed under cultural influence, and a high level of public use. These lakes were seen as priorities for restoration activities.

The general objectivies of the Lake Iroquois diagnostic-feasibility study were the following:

1. Develop phosphorus and hydrologic budgets for the lake.
2. Identify and quantify phosphorus sources within the watershed and lake.
3. Relate phosphorus loading to water quality parameters, and develop predictive models to determine water quality responses to changes in phosphorus loading.
4. Study the history of eutrophication in Lake Iroquois through paleolimnological techniques.

5. Select and evaluate potential watershed and in-lake treatment techniques designed to improve water quality through phosphorus limitation or other means.

Funding for this study was provided by the U.S. Environmental Protection Agency, Clean Lakes Program, as authorized under Section 314 of the Clean Water Act of 1977, P.L. 95-217, with matching funds provided by the State of Vermont.

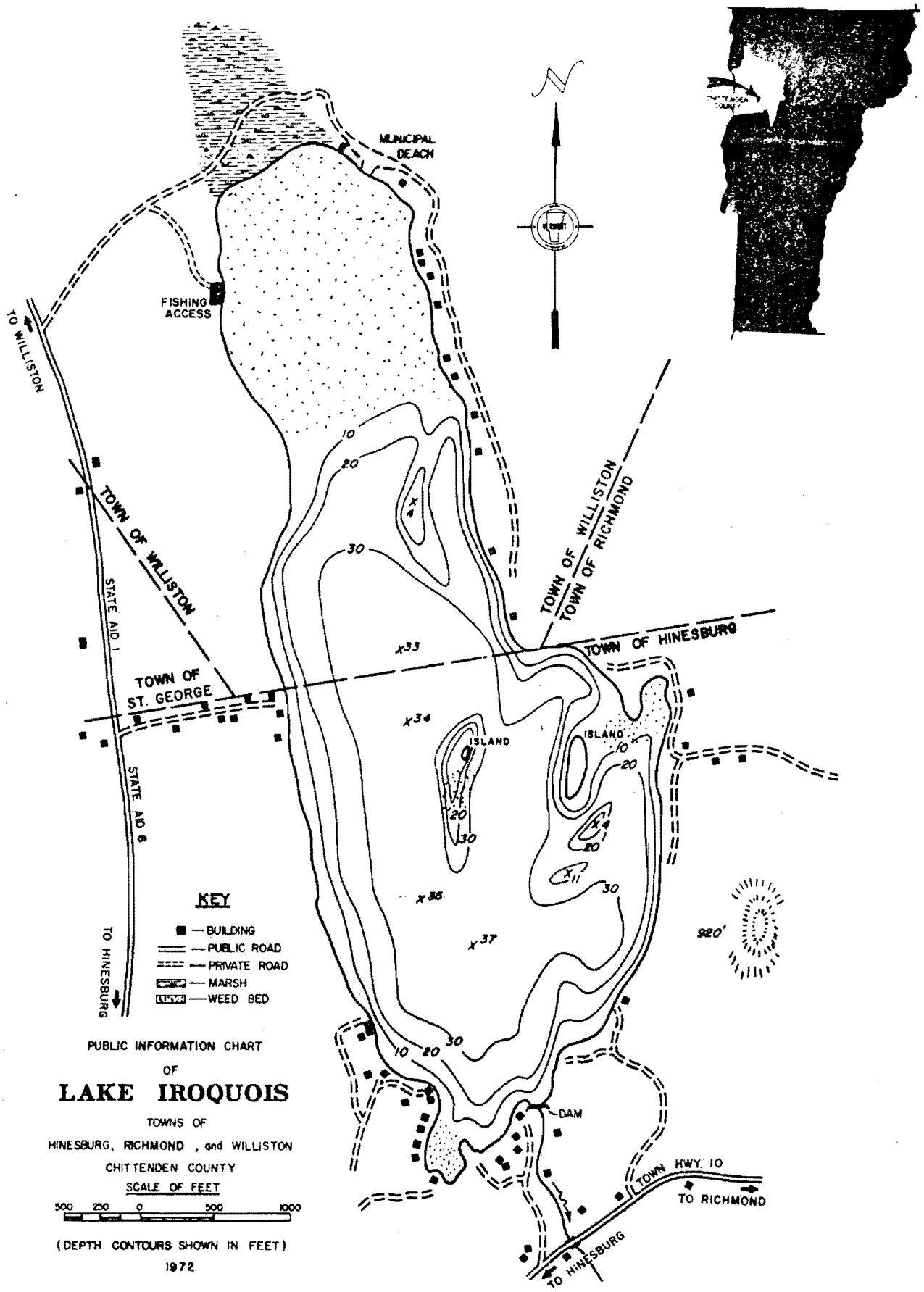


Figure 1. Lake Iroquois location and morphometry.

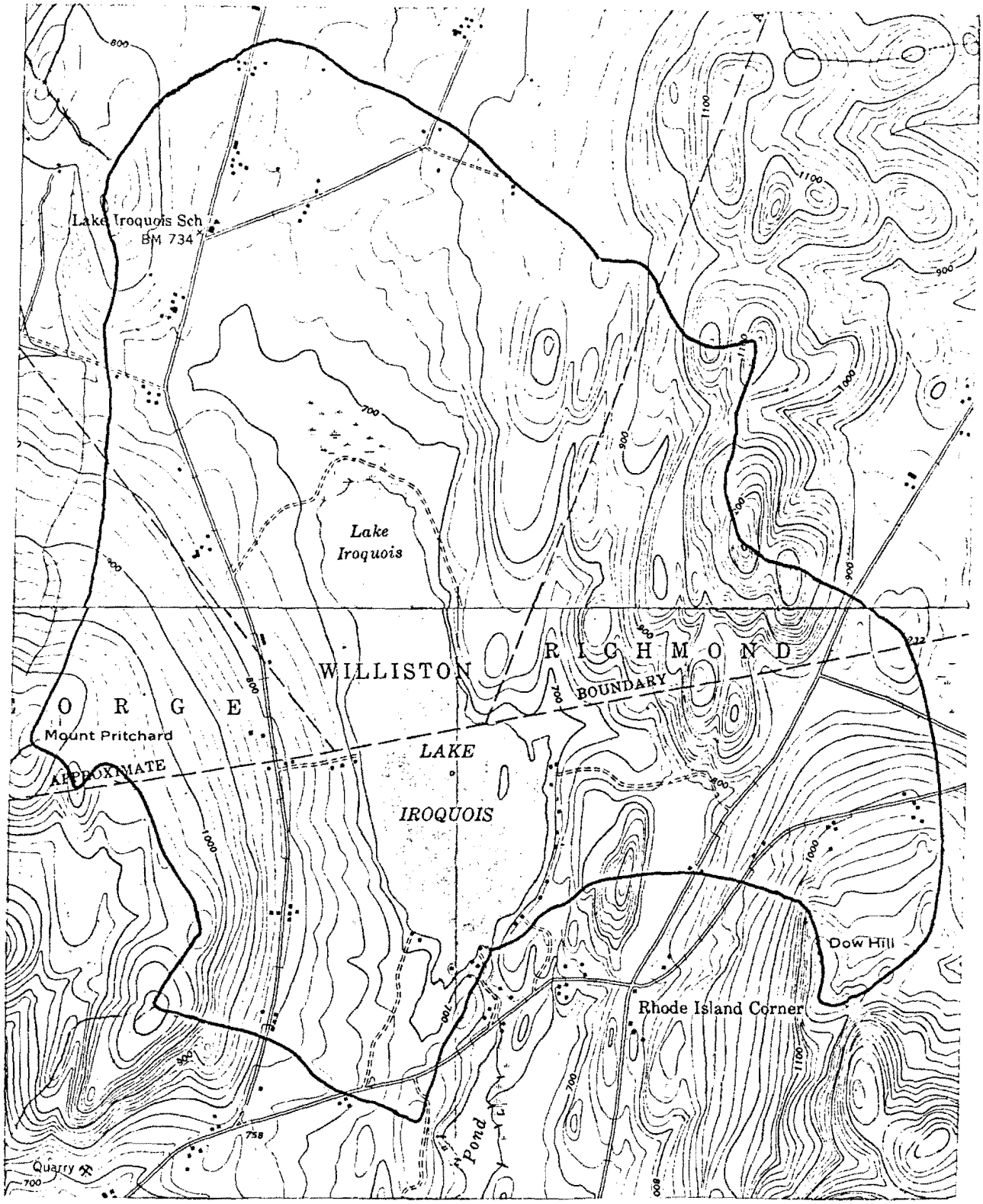
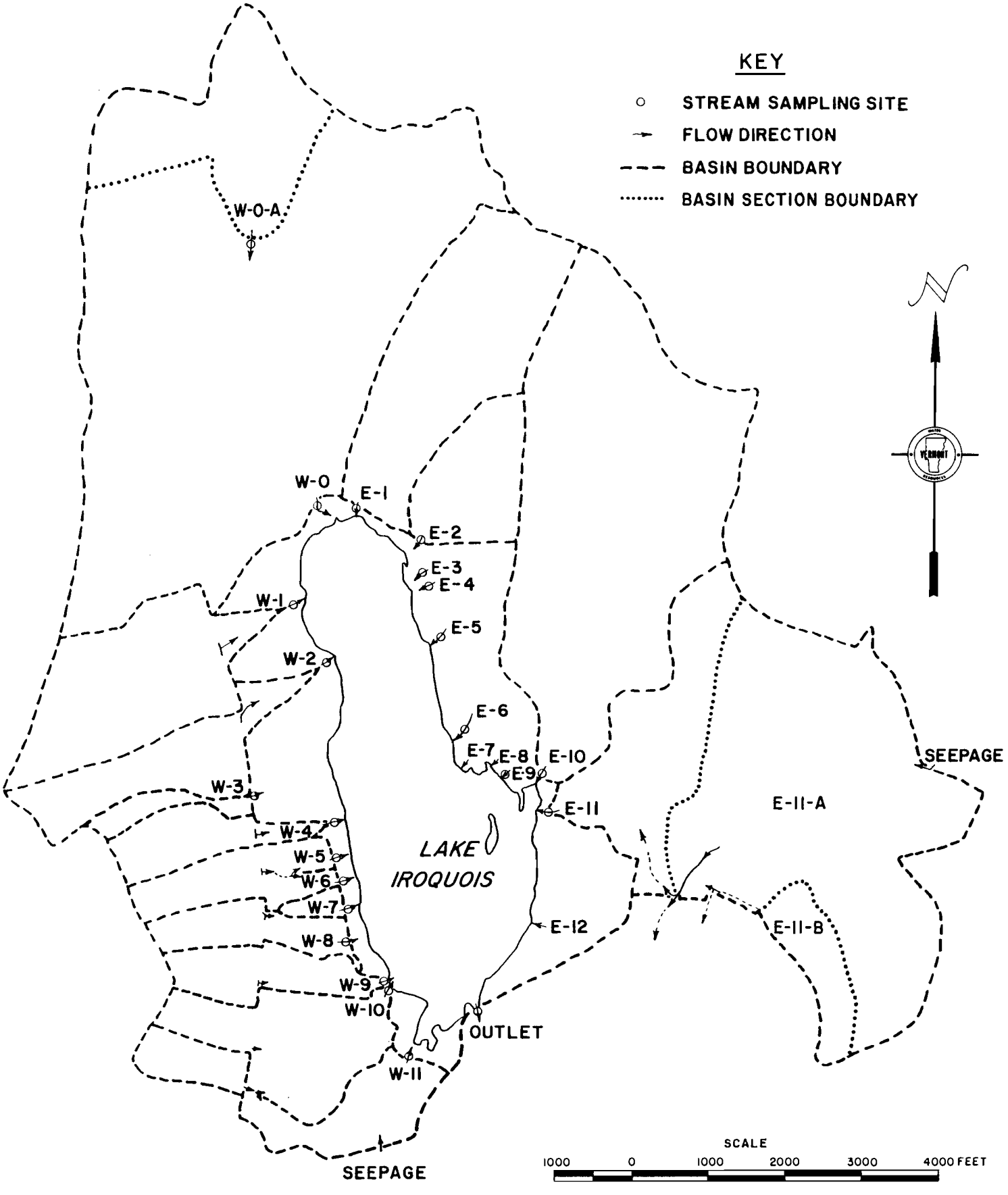


Figure 2. Lake Iroquois watershed topography. Scale = 1:24,000. Elevations are in feet.

Figure 3.

LAKE IROQUOIS WATERSHED AND SUB-BASIN BOUNDARIES



BACKGROUND AND DESCRIPTIVE INFORMATION

Location and Physical Description

Lake Iroquois is a 98.7 hectare (244 acre) lake located in Chittenden County in northwestern Vermont (Figure 1). Maximum depth is 11.3 meters (37 ft.), and average depth is 5.9 meters (19 ft.). More specific morphometric data is given in Table 1. The lake has a NNW-SSE orientation and a maximum fetch of 2.0 km. The lake was naturally formed, but the water level has been raised and is presently being controlled by a masonry and concrete dam located at the lake's south end.

The watershed area is 889.6 hectares (2198 acres). Watershed topography is shown in Figure 2. Figure 3 shows the locations of 16 surface inflow sites and their associated drainage basins which have been identified at Lake Iroquois. Eight additional small surface inflow sites with undetermined drainage basin boundaries are also indicated along the lake's east shore. Divergent streamflow occurs at the boundary of the W-10 and W-11 basins, in the W-5 and W-6 combined basin, and at two points in the E-11 basin. In the E-11 basin, portions of the divergent flow leave the watershed at each of the two points of occurrence. Seepage from two shallow impoundments just outside the watershed contributes to surface flows at sites W-11 and E-11. Three sub-basin sections are also delineated in Figure 3. These sections are separately evaluated for hydrologic or phosphorus yield characteristics in later sections of this report.

Water Chemistry and Biological Characteristics

Lake Iroquois is a soft water lake with total hardness values in the vicinity of 45-52 mg/l (as $CaCO_3$). The lake has a high total alkalinity (40 mg/l as $CaCO_3$) which makes it relatively insensitive to acidic deposition. pH values normally range between 7.4 and 8.5, although during the summer the hypolimnetic layer tends to drop below 7.0. The lowest pH

recorded during the 1982-1983 sampling was 6.6. Table 2 gives concentrations of the major ions and other chemical parameters during the 1983 spring overturn.

Table 1. Lake Iroquois morphometry.

Sample Depth (m)	Stratum Boundary Depth (m)	Area ($10^6 m^2$)	Stratum Volume ($10^6 m^3$)
	0	0.941	
0			.449
	0.5	0.856	
1			.792
	1.5	0.731	
2			.690
	2.5	0.650	
3			.624
	3.5	0.599	
4			.586
	4.5	0.564	
5			.548
	5.5	0.531	
6			.512
	6.5	0.492	
7			.464
	7.5	0.437	
8			.398
	8.5	0.360	
9			.307
	9.5	0.257	
10			.187
	10.5	0.124	
			.033
	11.3	0.00	

Total Volume = $5.590 \times 10^6 m^3$

Table 2. Concentrations of major ions and related parameters for Lake Iroquois water.

PARAMETER	RESULTS OF AN APRIL 14TH, 1983 SAMPLE[1]
Calcium (mg/l)	15.8
Magnesium (mg/l)	2.9
Sodium (mg/l)	5.7
Potassium (mg/l)	1.5
Total Hardness (mg/l as $CaCO_3$)	51
Chloride (mg/l)	10.2
Sulfate (mg/l)	9.8
Total Alkalinity (mg/l as $CaCO_3$)	40
pH	7.45
Conductivity (umhos)	141

[1]This was a 0-10m hose sample composited over 3 sites (see Figure 9). Results are similar to records of sample results from various dates in previous years back to 1968, except for pH, which is a more variable parameter.

Table 3 shows historical summer dissolved oxygen profiles and demonstrates that the lake frequently reaches anoxic conditions in the hypolimnion. The one profile taken in July of 1940 indicates that the lake may have been experiencing a lower oxygen depletion rate than it has at present. However, this profile was produced by different analytical methods and may not, therefore, be comparable.

Lake Iroquois has been classified as a eutrophic lake (Vermont Department of Water Resources and Environmental Engineering, 1980). Table 4 shows historical (1964-1980) chlorophyll-a concentrations. More recent (1981-1983) chlorophyll dynamics are discussed elsewhere in this report. Between 1973 and 1977, 635 kg of the algicide copper sulfate was applied by local residents each year. No chlorophyll records exist to document the effect of this application. 1983 was the first year in which chlorophyll-a levels were measured at one meter intervals rather than as a depth integrated composite. This more intensive sampling revealed a substantial hypolimnetic or metalimnetic algal population through most of the summer with chlorophyll-a concentrations as high as 100 ug/l at one point (Figure 4). This population had gradually migrated up into the epilimnion by the end of August. It is possible that such a metalimnetic algal population could have been present in previous years.

Table 5 lists the dominant algal genera found in historical samples. Generally the summer months are dominated by blue-green algae. This historical information reflects the enumeration of algal cells rather than biovolume, and therefore the genera listed as dominant may not have composed the bulk of the algal volume. Tables 6 and 7 contain a more quantitative description of the algal types present in 1982 and 1983. In both 1982 and 1983, *Cryptomonas* spp. composed greater than 5% of the biovolume throughout most of the year. This genus was far less dominant in 1983, however, than in 1982. Cyanophytes, particularly *Aphanizomenon flos-aquae* and *Anabaena* sp., were more dominant in the summer of 1983. The large chlorophyll-a peaks which occurred

Table 3. Historical summer dissolved oxygen profiles.

Date	Depth (m)	D.O. (ppm)
7-03-40	0	9.7
	3	9.5
	6	8.9
	8	7.6
	9	2.1
	10	1.2
	11	0.5
8-08-68	0	8.2
	1	8.1
	2	8.4
	3	8.3
	5	8.2
	6	7.1
	7	0.1
	8	0.0
	9	0.0
	10	0.0
6-14-76	0	9.3
	6	8.3
	8	4.0
	10	1.8
7-12-76	0	7.5
	6	3.5
	10	0.1
8-09-76	0	8.4
	5	8.3
	7	0.8
	9	0.0
6-17-77	0	9.5
	6	8.5
	8	4.5
	10	0.0
7-25-77	0	8.3
	7	3.5
	10	0.3
6-16-78	5	9.3
	6	9.3
	7	8.9
	8	4.3
	9	3.0
7-13-78	7	6.4
	8	2.8
	9	0.3
	10	0.4
	10.5	0.2
8-18-78	6	6.7
	7	4.1
	8	0.8
	9	0.2
	10	0.1

Table 4. 1964-1980 Chlorophyll-a concentrations.

Date	Chlorophyll-a (ug/l)	Date	Chlorophyll-a (ug/l)	Date	Chlorophyll-a (ug/l)
6-03-64	22	9-22-78	9	6-13-80	4
6-24-69	11	9-28-78	18	6-18-80	9
8-04-69	28	10-03-78	14	6-26-80	8
10-20-69	19	10-13-78	22	7-03-80	5
		10-19-78	10	7-10-80	3
5-31-77	2	10-24-78	12	7-24-80	7
6-27-77	7			7-31-80	13
7-25-77	14	6-06-79	4	8-07-80	14
		6-14-79	6	8-14-80	7
6-08-78	3	6-20-79	8	8-21-80	8
6-18-78	1	6-28-79	14	8-29-80	18
6-22-78	1	7-06-79	23		
6-30-78	4	7-11-79	17		
7-07-78	3	7-25-79	7		
7-13-78	3	8-02-79	6		
7-20-78	6	8-17-79	8		
7-28-78	8				
8-03-78	20				
8-11-78	54				
8-18-78	27				
8-24-78	31				
8-31-78	18				
9-07-78	10				
9-14-78	11				

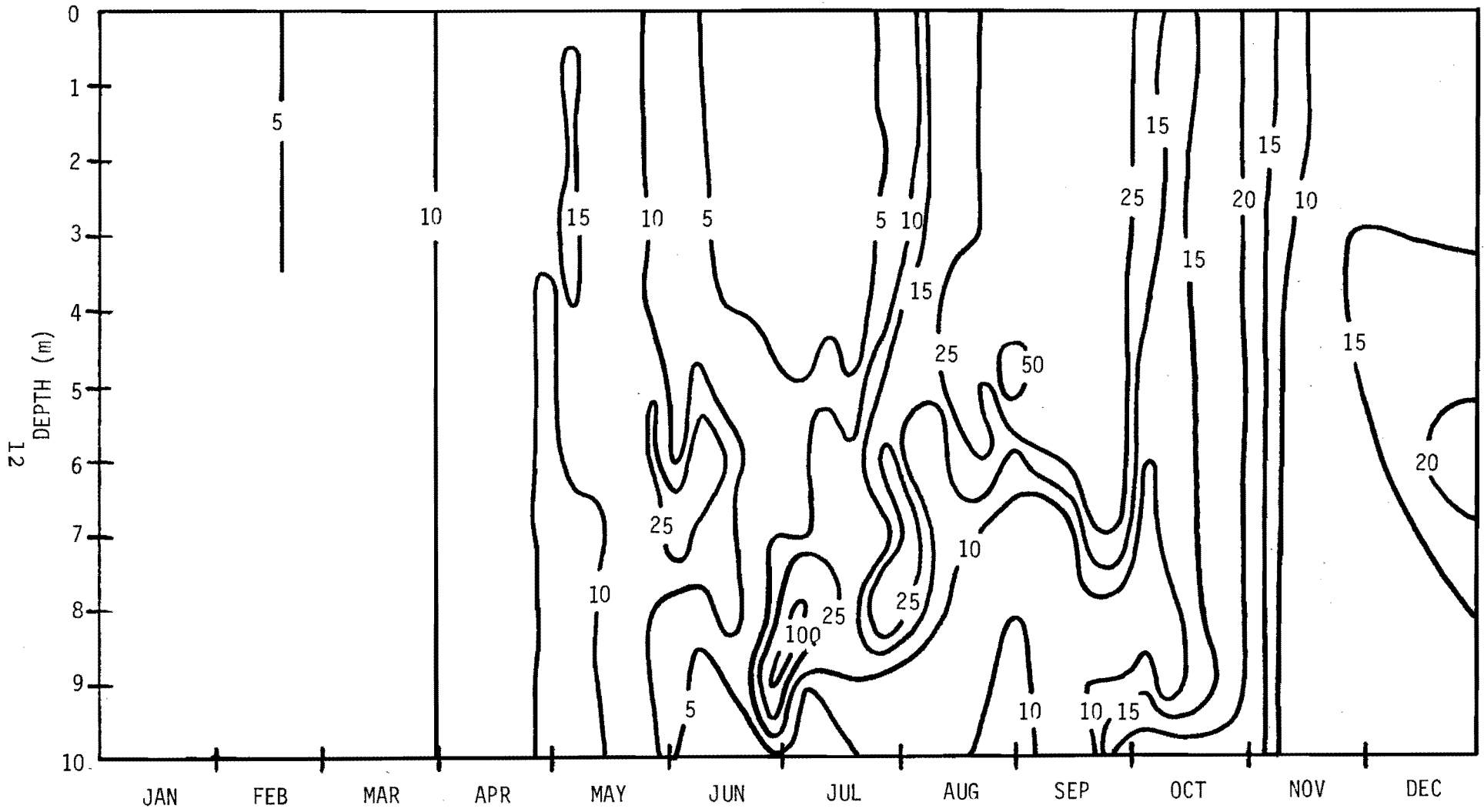


Figure 4. 1983 depth-time isopleths for chlorophyll-a concentrations.

Table 5. Dominant algal genera in quantitatively analyzed lake samples collected prior to 1982.

Sample Date	Algal genera composing 5% or more of the total cells liter present (in order of dominance)
10-04-76	Coelosphaerium Microcystis Cryptomonas/Rhodomonas
05-31-77	Microcystis Oscillatoria Croomonas Coelosphaerium
07-25-77	Aphanizomenon Oscillatoria
1978 (date unknown)	Anabaena Rhaphiopsis Coelosphaerium Single-celled chlorophytes Oscillatoria
1981 (date unknown)	Ceratium Aphanizomenon Stephanodiscus

Table 6. Biovolume (u^3/liter x 10^6) and number of cells (cells/liter x 10^6) of dominant algal classes in 1982.

Algal Type	5/25 Biovolume	5/25 #	6/22 Biovolume	6/22 #	7/20 Biovolume	7/20 #	8/24 Biovolume	8/24 #
Diatoms	1640	0.708	33	0.011	9	0.014	57	0.042
Chlorophyta	172	1.508	38	0.44	20	0.322	35	0.394
Cyanophyta	0	0.0	2	0.619	41	10.302	33	8.185
Cryptomonas Rhodomonas	231	0.577	1233	0.519	398	0.181	940	0.723
Other[1]	36	0.018	12	0.007	50	0.028	226	0.081
TOTAL	2079	2.811	1319	1.596	518	10.847	1290	9.425
Dominant[2] Genera	Stephanodiscus (72%) Cryptomonas (8%)		Cryptomonas (93%)		Cryptomonas (76%) Trachelomonas (10%)		Cryptomonas (72%) Trachelomonas (10%) Ceratium (8%)	

1 Includes Pyrrophyta, non-diatom Chrysophyta, Trachelomonas.

2 Genera composing greater than 5% of biovolume

Table 6 (cont.). Biovolume (u^3/liter x 10^6) and number of cells (cells/liter x 10^6) of dominant algal classes in 1982.

Algal Type	9/22 Biovolume	9/22 #	10/27 Biovolume	10/27 #	11/17 Biovolume	11/17 #	12/7 Biovolume	12/7 #
Diatoms	88	0.109	1085	3.165	149	0.119	353	0.148
Chlorophyta	14	0.207	116	2.412	26	0.354	16	0.263
Cyanophyta	66	4.155	35	0.358	29	0.356	5	0.091
Cryptomonas Rhodomonas	804	0.721	1286	0.910	1246	1.303	1100	0.635
Other[1]	332	0.074	99	0.063	41	0.037	47	0.042
TOTAL	1303	5.266	2620	6.908	1492	2.169	1521	1.179
Dominant[2] Genera	Cryptomonas (60%) Ceratium (17%) Trachelomonas (8%)		Cryptomonas (47%) Asterionella (28%) Stephanodiscus (8%)		Cryptomonas (77%) Stephanodiscus (8%) Rhodomonas (7%)		Cryptomonas (71%) Stephanodiscus (21%)	

1 Includes Pyrrophyta, non-diatom Chrysophyta, Trachelomonas.

2 Genera composing greater than 5% of biovolume

Table 7. Biovolume (u^3/liter x 10^6) and number of cells (cells/liter x 10^6) of dominant algal classes in 1983.

Algal Type	2/17 Biovolume	2/17 #	4/14 Biovolume	4/14 #	5/12 Biovolume	5/12 #	6/15 Biovolume	6/15 #
Diatoms	68	0.245	102	0.168	763	0.830	129	0.426
Chlorophyta	32	1.043	57	1.064	88	1.433	741	1.199
Cyanophyta	5	0.092	6	1.420	70	1.115	45	0.902
Cryptomonas Rhodomonas	143	0.128	432	0.852	466	0.795	114	0.234
Other[1]	174	0.142	830	1.377	1606	0.490	308	1.271
TOTAL	422	1.650	1427	4.811	2993	4.663	1337	4.032
Dominant[2] Genera	Cryptomonas (32%) Asterionella (15%) single-celled chlorophytes (5%)		Peridinium (33%) Cryptomonas (24%) Dinobryon (21%) Rhodomonas (6%)		Peridinium (48%) Cryptomonas (13%)		Batryococcus (52%) Dinobryon (21%) Cryptomonas (7%) Asterionella (6%)	

1 Includes Pyrrophyta, non-diatom Chrysophyta, Trachelomonas.

2 Genera composing greater than 5% of biovolume

Table 7 (cont.). Biovolume (u^3/liter x 10^6) and number of cells (cells/liter x 10^6) of dominant algal classes in 1983.

Algal Type	7/13 Biovolume	7/13 #	8/17 Biovolume	8/17 #	9/14 Biovolume	9/14 #	10/12 Biovolume	10/12 #
Diatoms	22	0.057	24	0.028	39	0.064	61	0.213
Chlorophyta	42	1.129	56	1.541	37	1.072	271	2.249
Cyanophyta	667	5.288	1519	93.411	1967	1.072	209	16.938
Cryptomonas Rhodomonas	83	0.114	117	0.177	146	151.609	954	1.200
Other[1]	808	0.270	70	0.050	62	0.035	126	0.071
TOTAL	1622	6.848	1786	95.207	2251	153.015	1621	20.666
Dominant[2] Genera	chrysophyte (42%) Anabaena (40%) Trachelomonas (8%)		Aphanizomenon (63%) Anabaena (21%) Cryptomonas (6%)		Aphanizomenon (87%) Cryptomonas (6%)		Cryptomonas (55%) Aphanizomenon (11%) Trachelomonas (8%) Staurastrum (8%) single celled chlorophytes (6%)	

1 Includes Pyrrophyta, non-diatom Chrysophyta, Trachelomonas.

2 Genera composing greater than 5% of biovolume

Table 7 (cont.). Biovolume (u^3/liter x 10^6) and number of cells (cells/liter x 10^6) of dominant algal classes in 1983.

Algal Type	11/16 Biovolume	#	12/29 Biovolume	#
Diatoms	2334	1.164	132	0.256
Chlorophyta	155	0.689	30	0.418
Cyanophyta	343	26.380	152	11.720
Cryptomonas Rhodomonas	231	0.263	288	0.739
Other[1]	50	0.028	37	0.021
TOTAL	3120	28.524	639	13.154
Dominant[2] Genera	Stephanodiscus (66%) Aphanizomenon (11%) Cryptomonas (7%) Melosira (6%)		Cryptomonas (30%) Aphanizomenon (24%) Rhodomonas (15%) Cyclotella (10%) Trachelomonas (6%) Asterionella (5%)	

1 Includes Pyrrophyta, non-diatom Chrysophyta, Trachelomonas.

2 Genera composing greater than 5% of biovolume

in August and September of 1983 were composed mostly of Aphanizomenon flos-aquae. In addition to different species compositions between the two years, the total biovolume during the summer months was consistently higher in 1983 than in 1982.

Lake Iroquois supports a substantial macrophyte population covering about 37% of the lake area. An extensive macrophyte survey done in 1982 (see Aquatic Macrophytes section) revealed an average macrophyte density of 68 g dry wt./m^2. The four most dominant species found in the 1982 survey were Elodea Nuttallii, Heteranthera dubia, Potamogeton praelongus and Potamogeton zosterformis. Lake residents say they have observed increases in macrophyte abundance in the past 10 to 20 years. One survey performed by the Vermont Department of Water Resources and Environmental Engineering in 1977 delineates past areas of little to no plant growth which currently have dense coverage. Whether this represents an actual trend towards more extensive growth, however, cannot be substantiated due to year to year variability.

Table 8 contains a list of the fish species present. Generally, the lake supports a warmwater fishery with yellow perch, pumpkinseeds and golden shiners composing the bulk of the population. A 1977 creel survey revealed that the majority of fishermen were fishing for bass, although over 96% of the catch was composed of yellow perch (Anderson, 1977). The 1968 fish survey on Lake Iroquois revealed the yellow perch population to be overabundant and stunted in growth (Anderson, 1969).

Present and Historical Lake Uses

Lake Iroquois has long been popular for swimming, fishing, and various boating activities. In 1958 the public swimming beach at the north end of the lake was leased by the towns of Williston, Richmond, Hinesburg, and St. George. The beach receives heavy public use during the summer, both on the weekends and during the week when Red Cross swimming lessons are offered. In addition to those participating in the town-sponsored swimming lesson program, a minimum of 1500 people are estimated to make

Table 8. Fish species present[1].

Smallmouth bass (*Micropterus dolomieui*)
Largemouth bass (*Micropterus salmoides*)
Chain pickerel (*Esox niger*)
Northern pike (*Esox lucius*)
Walleye (*Stizostedion vitreum*)
Yellow perch (*Perca flavescens*)
Pumpkinseed (*Lepomis gibbosus*)
Rock bass (*Ambloplites rupestris*)
Brown bullhead (*Ictalurus nebulosus*)
American eel (*Anguilla rostrata*)
Common white sucker (*Catostomus commersoni*)
Golden shiner (*Notemigonus crysoleucas*)
Emerald shiner (*Notropis atherinoides*)
Black-chin shiner (*Notropis heterodon*)
Bluntnose minnow (*Pimephales notatus*)
Banded killifish (*Fundulus diaphanus*)

[1]Based on Vermont Fish and Game Department survey results, 1940-77. Stocking of lake trout (*Salvelinus namaycush*) was mentioned in the 1940 survey report.

upon an adequate stocking source and resolution of the apparent conflict of use existing between boaters and fishermen.

Currently there are no restrictions on motor size or speeds, so the lake provides an opportunity for many types of boating activities. During the 1982 summer season, the boating uses observed included water skiing, sailing, wind surfing, and canoeing. Other observed uses of Lake Iroquois include ice skating, snowmobiling and cross country skiing.

The aesthetic and recreational attractions of the lake have resulted in the development of numerous seasonal and permanent homesites around the lake. There are 92 residences located along the lakeshore, a high number for a Vermont lake of this size. The first seasonal homes were established around 1890. Shoreline development proceeded at an irregular, but generally gradual rate to the present. Water quality is of immediate importance to shoreline residents as it affects both recreational use and property values. Most residents also use the lake as a domestic water supply for all purposes except drinking water.

Residents from various sections of the lake's shoreline have formed four separate lake associations. Representatives from each association comprise the Lake Iroquois Action Committee. The Action Committee applied approximately 635 kg of copper sulfate to the lake each summer from 1973 to 1977 in an effort to control nuisance algal blooms. In 1979 a weed harvester was constructed under the guidance of the Action Committee and the towns of Williston and Hinesburg with funds provided in part by a State grant. Organizational and mechanical problems delayed the start of harvesting until the summer of 1984.

Some form of outlet control structure has existed at the lake since the early 1800's. Historical records indicate the first small dam was built in 1816 to regulate flows for a downstream grist mill. A sawmill was built at the dam site in 1822. Conflicting records exist for the construction date of the present dam. Dating results of a sediment core indicate this dam was probably built between 1900 and 1905 (see Paleolimnological Study section). The dam is believed to have raised the

use of the beach annually, based on sales of seasonal user permits. Swimming activities also take place at numerous sites around the lake's shoreline.

A high density of macrophytes exists both in the vicinity of the public beach and along much of the shoreline (see Figure 48) reducing the quality of swimming. Frequent raking of the public beach is required to remove macrophytes and algal scums that accumulate on the shoreline. Algal turbidity and floating and suspended plant fragments were frequently noted in the swimming area during 1983. Swimming is a major recreational use of the lake which is significantly impaired by present lake conditions.

A boat launch owned and maintained by the Vermont Fish and Wildlife Department was acquired in 1963, and provides access to the fishery resource of Lake Iroquois. In the past, the lake was noted for its good northern pike and walleye fishing in the winter and summer and was also known to support a good perch fishery. State fishery investigations indicate that the quality of the fishery has declined. Maintenance of the walleye population was dependent on stocking, which has been discontinued. The decline in the northern pike population may be due to competition from inadvertently introduced chain pickerel and the opening of the lake to ice fishing. Yellow perch, which showed excellent growth in 1940, have shown poor growth in more recent years. This is probably due to competition from pumpkinseeds (introduced after 1940), and to less effective population control through predation by the declining northern pike and walleye populations. A survey of ice fishermen in 1969 revealed a catch of 0.4 fish per man-hour. The amount of fishing activity in the summer appeared to be limited by the large amount of boating activity on the lake (Anderson, 1969). A creel survey in 1977 revealed that fishermen expended 631 man-hours in the winter and 3,589 man-hours in the summer of that year. In the winter, one-third of the fishermen were residents of the local Chittenden County and two-thirds were from Massachusetts. In the summer, 83% were residents of Chittenden County (Anderson, 1977). Reclamation was recommended for Lake Iroquois in 1977, contingent

lake level about seven feet and to have created most of the shallow northern bay. The dam is owned by the Iroquois Manufacturing Company which presently has control over water levels in the lake. A petition process does exist by which the State of Vermont may set regulations governing water levels and thus prohibit excessive lake drainage that would be detrimental to recreational use. Currently the Iroquois Manufacturing Company uses the storage capacity for flow augmentation of the Lower Pond dam downstream of Lake Iroquois. (The Lower Pond dam supplies cooling water for manufacturing processes). The Iroquois Manufacturing Company has recently projected plans, still in the feasibility study phase, to develop a hydroelectric facility on a small pond downstream of Lower Pond.

Geology

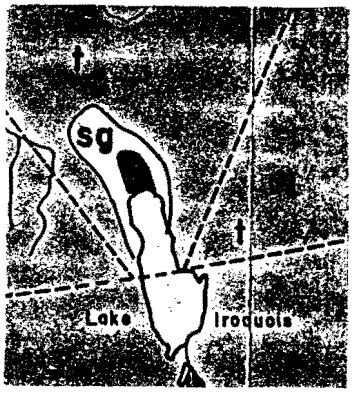
Lake Iroquois is located within the foothills of the Green Mountains. The surficial material of the watershed is largely till deposited during the Wisconsin glacial stage, about 13,000 years ago (Figure 5). The till blanket is generally less than 10 feet thick on the uplands, with greater thicknesses found in low-lying areas (Stewart, 1973). Till depths up to 62 feet are documented by domestic well drilling records. The high sand content of most tills in this region increases the permeability above that which is normally characteristic of till material. However, subsoil layers are frequently cemented or contain higher percentages of clay which reduces their permeability.

A deposit of lacustrine sand and gravel is located at the north end of the lake. Well drilling records from sites near the margin of this deposit indicate a thick cemented layer of sandy and gravelly material.

Two bedrock formations underlie the watershed (Figure 6). The north-south band on the east side of the lake is a segment of the Pinnacle formation. This is a schistose greywacke, a highly metamorphosed rock in which quartz, alibite, serecite, biotite, and chlorite are the dominant minerals. The remainder of the

FIGURE 5.

Surficial Material of the Lake Iroquois Area (adapted from Stewart 1973).



LEGEND

t

Glacial Till - generally thin over bedrock, much bedrock exposed, unsorted, poorly drained, bouldery surface. Low water potential.

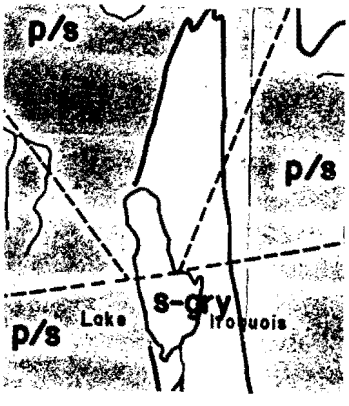
sg

Lacustrine and Marine Sands and Gravels - Marine deposits only along Lake Champlain. Predominantly sand and pbebly sand, well drained above the water table. Good source for sand. Moderate to high water potential below the water table.

Peat and Muck - Swampy, poorly drained areas. water table at or near the surface.

FIGURE 6.

Bedrock of the Lake Iroquois Area (adapted from Stewart 1973).



LEGEND

s-gry

Schistose Greywacke. Complex, light to dark grey color, highly metamorphosed, sandy. Generally resistant to erosion, but dark minerals susceptible to chemical weathering.

Complex, highly metamorphosed rocks of the Green Mountains consisting primarily of Phyllites, Gneisses, Schists with varying amounts of Greenstones and Amphibolites. Highly fractured.

- g - gneiss
- s - schist
- p - phyllite
- gr - greenstone
- p/s - undifferentiated phyllites and schists
- s-qtz - schists with interbedded quartzite
- gr/am - undifferentiated greenstones and amphibolites
- s/gn-qtz - undifferentiated schists and gneisses with interbedded quartzite
- p/s-qtz - undifferentiated phyllites and schists with interbedded quartzite

area is underlain by the Underhill formation, a greenish quartzitic schist (quartz-serecite-albite-chlorite-biotite) with serecite-quartz-chlorite phyllite common in the lower part (Doll et al., 1961).

Soils

Soils in the Lake Iroquois watershed are predominantly stony loams, with slopes ranging from 0-60%. Surface permeabilities are moderate. The dominant soils present all have fragipans at 1-3 feet, or bedrock at 1-1.5 feet (Lyman rocky loam). Fragipan permeabilities are generally an order of magnitude less than surface permeabilities, resulting in lateral drainage above the fragipan, and an increased potential for surface runoff, since the surface soil layers can be more easily saturated.

Soils belonging to the Lyman-Marlow association comprise 40% of the watershed land area. In most areas, the Lyman and Marlow soils are not separately delineated, but are mapped as a complex (Figure 7 and Table 9). The Lyman soils are somewhat excessively drained and shallow to bedrock. They are rocky or very rocky in most places and are the steepest soils in the association. The Marlow soils are well drained, loamy, and stony. They have a slowly permeable fragipan within 30 inches of the surface. These soils are moderately steep or steep.

Peru soils occupy 27% of the land area. These soils are moderately well drained and loamy and have a slowly permeable fragipan that is less than 34 inches from the surface. Peru soils are mainly gently sloping to sloping.

Cabot soils, which comprise 26% of the land area, are similar to the Peru soils in texture and also have a fragipan, but Cabot soils have a higher water table than Peru soils and are less sloping. The fragipan of Cabot soils is within 24 inches of the surface. Cabot soils occupy lower positions than Peru soils and are mainly level to gently sloping.

Miscellaneous soil types account for the remaining 7% of the land area. These soils have generally formed in low-lying or wet areas, or in exposed sand and gravel deposits.

Figure 7. Lake Iroquois Watershed Soils Map (Adapted from Allen 1974).

Table 9. Soil descriptions.

Soil Symbol	Soil Type	Depth to bedrock (ft)	Depth to seasonal high water table (ft)	Fragipan depth (ft)	Surface permeability (in/hr)	Soil erodibility coefficients (K)
Au	Au Gres Fine sandy loam	5+	0.5-1	none	2.0-6.3	.15
Ca	Cabot stony silt loam	5+	0-1	1-2	0.63-2.0	.28
Cb	Cabot extremely stoney silt loam	5+	0-1	1-2	0.63-2.0	.28
Cs	Colton and Stetson soils	4+	4+	none	2.0-6.3	.17
Dd	Duane and Deerfield soils	5+	1-2	none	2.0-20.0	.17
Lf	Limerick silt loam, very wet	5+	0-0.5*	none	0.63-2.0	.49
Lm	Lyman/Marlow rocky loams	1-1.5/5+	4+/1-1.5	none/1-3	0.63-6.3	.26
Ly	Lyman/Marlow very rocky loams	1-1.5/5+	4+/1-1.5	none/1-3	0.63-6.3	.26
Ma	Marlow stony loam	5+	1-1.5*	1-3	0.63-2.0	.20
Me	Marlow extremely stony loam	5+	1-1.5*	1-3	0.63-2.0	.20
Mp	Muck and Peat	5+	0-0.5	none	--	--
Pc	Peacham stony silt loam	5+	0-0.5	0.5-2	0.63-2.0	.28
Pe	Peru stony loam	5+	1-2*	1.5-3	0.63-2.0	.20
Ps	Peru extremely stoney loam	5+	1-2*	1.5-3	0.63-2.0	.20
Rk	Rock land	--	--	--	--	--
Sk	Scarboro loam	5+	0-0.5	none	0.63-6.3	.17
St	Stetson gravelly fine sandy loam	4+	4+	none	2.0-6.3	.17

*perched water table

The shoreline soils around Lake Iroquois all have severe limitation ratings for septic tank disposal fields (Table 10). Limitations include high water table, slow permeability, slopes, and shallowness to bedrock.

Table 10. Septic tank disposal field limitations for shoreline soils.

Soil Symbol	Soil Type	Estimated Degree and Dominant Kind of Limitation
CaA	Cabot stony silt loam	Severe: high water table; slow permeability
CaC	Cabot stony silt loam	Severe: high water table; slow permeability; slopes
CbA	Cabot extremely stony silt loam	Severe: high water table; slow permeability
CbD	Cabot extremely stony silt loam	Severe: high water table; slopes; slow permeability
CsD	Colton and Stetson soils	Severe: slopes
LyD	Lyman/Marlow very rocky loams	Severe: shallow to bedrock; slow permeability; slopes
MeC	Marlow extremely stony loam	Severe: slow permeability; slopes
PsE	Peru extremely stony loam	Severe: slopes; slow permeability
StC	Stetson gravelly fine sandy loam	Severe: slopes

Land Use

Figure 8 and Table 11 depict land use types and distribution in the Lake Iroquois watershed. The eastern half of the watershed is predominantly woodland, while the western half is characterized by more diversified residential and agricultural land uses. Land use breakdowns for individual sub-basins are shown in Table 12.

Table 11. Land use in the Lake Iroquois watershed.

Land Use	% of Watershed
Woodland	53.4
Hayfield	11.6
Water	10.0
Pasture	9.7
Idle grassland	8.5
Residential	5.1
Wetland	0.9
Row crops	0.8

Figure 8. LAKE IROQUOIS WATERSHED LAND USE

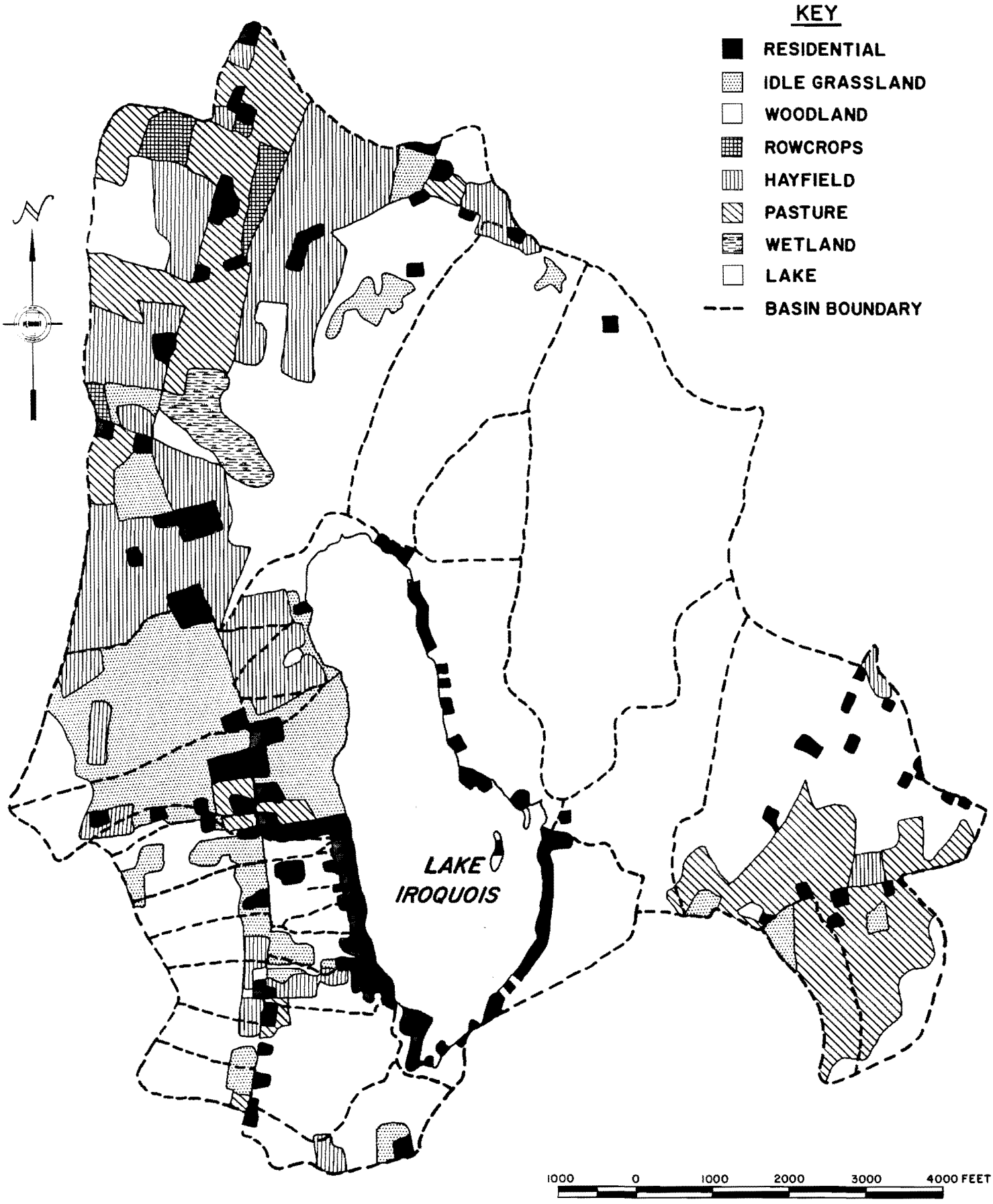


Table 12. Sub-basin land use.

% of Sub-Basin in Each Land Use

sub-basin*/area (ha)	Woodland	Hayfield	Water	Pasture	Idle Grassland	Residential	Wetland	Row crops
W-0/243.8	30.1	35.2	--	17.6	5.1	5.1	3.7	3.2
W-1/43.6	12.1	16.6	--	--	66.7	4.5	--	--
W-2/24.3	20.6	7.7	--	6.0	49.6	16.1	--	--
W-3/4.9	40.0	12.0	--	10.0	24.0	14.0	--	--
W-4/13.5	64.5	--	--	--	32.6	2.9	--	--
W-5,6/15.9	75.9	--	--	--	11.1	13.0	--	--
W-7/2.2	95.5	--	--	--	--	4.5	--	--
W-8/12.0	60.7	8.2	--	--	23.0	8.2	--	--
W-9/14.6	61.1	22.1	--	1.3	5.4	10.1	--	--
W-10/33.4	90.3	1.5	--	1.8	2.3	4.1	--	--
W-11/20.4	76.4	6.7	--	1.9	11.1	3.8	--	--
E-1/54.3	95.7	2.3	--	--	1.4	0.5	--	--
E-2/19.8	100.0	--	--	--	--	-	--	--
E-10/122.7	99.7	--	--	--	--	0.3	--	--
E-11/164.5**	64.5	1.7	--	29.3	2.1	2.4	--	--
(unsampled areas)								
EAST SHORE/57.3	86.2	--	--	--	--	13.8	--	--
WEST SHORE/42.4	18.7	21.0	--	3.7	29.3	27.2	--	--

* See Figure 8 for sub-basin locations.

** This area contributes only a portion of its drainage to the lake due to divergent stream flows.

METHODS

Lake Sampling

Lake sampling sites are shown in Figure 9. Tables 13, 14, and 15 give the sampling schedules during 1981, 1982, and 1983. Parameters measured and their abbreviations used in these tables are as follows.

Dissolved Oxygen (D.O.)	Alkalinity (Alk.)
Total Phosphorus (TP)	Total Hardness (T-Hard)
Dissolved Phosphorus (DP)	Calcium (Ca)
Ammonium (NH_4-N)	Magnesium (Mg)
Nitrate plus Nitrite (NO_3+ NO_2-N)	Sodium (Na)
	Potassium (K)
Total Kjeldahl Nitrogen (TKN)	Chloride (Cl^-)
Total Iron (T-Fe)	Sulfate (SO_4)
Dissolved Iron (D-Fe)	Total Silica (SiO_2)
Total Manganese (T-Mn)	Temperature (Temp.)
Dissolved Manganese (D-Mn)	Secchi Disc Transparency
Conductivity (Cond.)	Chlorophyll-a (Chl-a)
pH	Phytoplankton

All lake and stream water quality data collected during the course of this study has been stored in the U.S. Environmental Protection Agency's STORET system and is available on request. Some additional sampling was also conducted through Vermont's Lay Monitoring Program and Spring Phosphorus Sampling Program during the study period. This data is presented in Warren (1984).

During 1981 and 1982, lake samples from discrete depths were collected with a Kemmerer water sampler. During 1983, lake samples were collected with a peristaltic pump. Filtration for dissolved parameters was done in-line at the time of sampling, using a 0.45 um membrane filter. Hose samples (see Tables 13-15) were collected by slowly lowering a weighted garden hose end to the appropriate depth, tightly pinching the upper end, raising

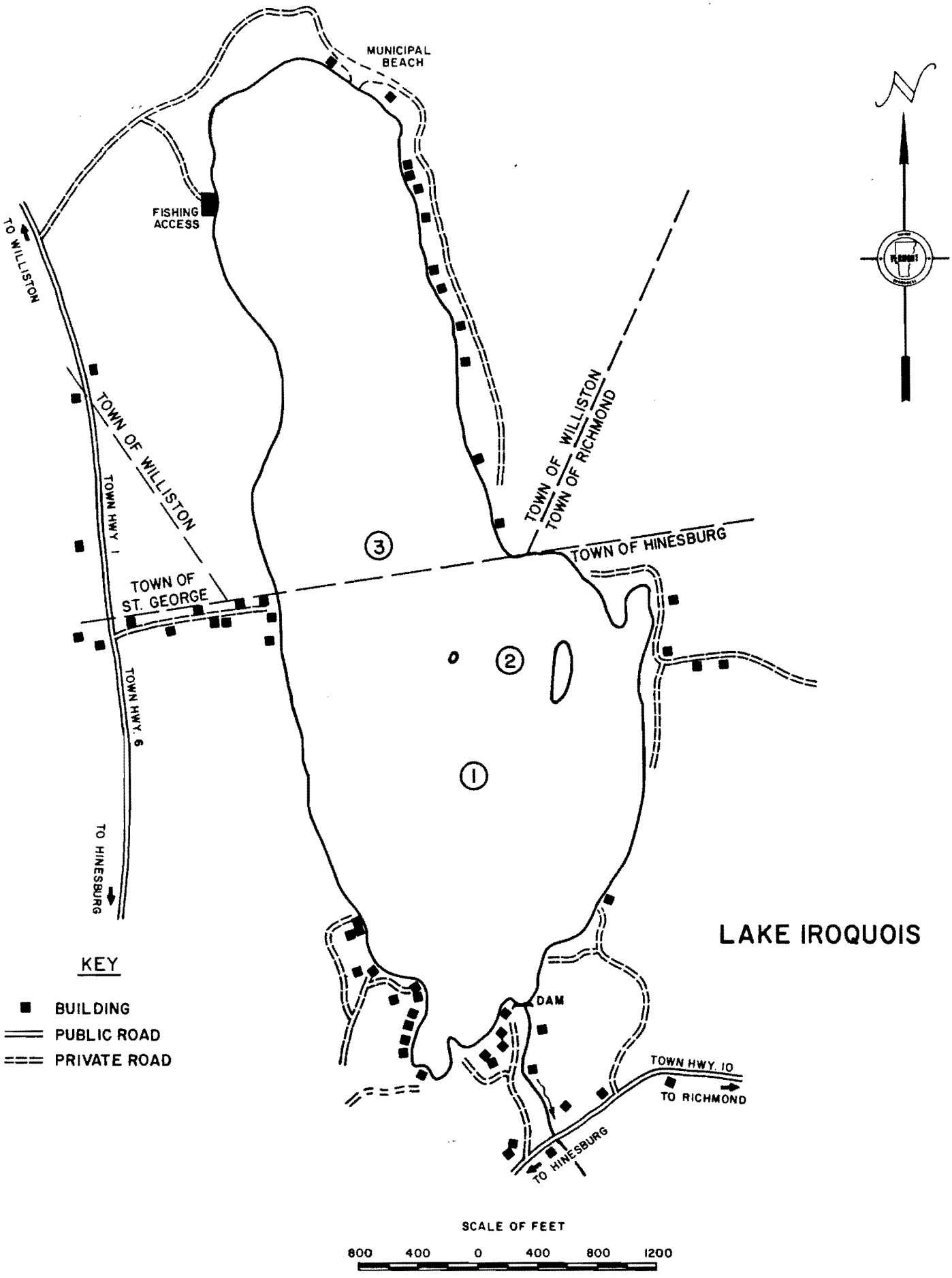


Figure 9. Lake sampling sites.

Table 13. 1981 Lake sampling schedule and parameters.[1]

Date	D.O., Temp. (every meter)	TP (every meter)	Chl-a (hose sample)[2]	Secchi Disc Transparency
5-12	X	X	X	X
5-15	X			X
5-21	X			X
5-28	X	X	X	X
6-2	X			X
6-8	X	X	X	X
6-11	X			X
6-19	X			X
6-26	X	X	X	X
7-7				X
7-10	X	X	X	X
7-25	X	X	X	X
8-6	X	X	X	X
8-21	X	X	X	X
9-22	X	X	X	X
10-6	X	X	X	X
10-23	X	X	X	X
11-3	X	X	X	X
11-17	X	X	X	X

[1]all samples were collected at site 1 (Figure 9)
[2]hose samples were taken to twice the Secchi disc transparency depth

Table 14. 1982 Lake sampling schedule and parameters.

Date	D.O., Temp. (every meter)	TP (every meter)	TP (0,2,8 m sites 2,3)	Chl-a (hose sample)[2]	Phytoplankton (hose sample)[2]	Secchi Disc Transparency	Cl^- (every 2 m)	TKN, NH_4-N, NO_3+NO_2-N (every meter)	pH, T-Fe T-Mn (1, 7,8,9,10m)	Cond., Alk. T-Hard, SiO_2, SO_4 (1, 10 m)
1-7	X	X								
1-27	X	X								
2-9		X								
3-1		X								
3-16	X	X								
3-30	X	X								
4-23	X	X		X		X	X			
5-1	X	X		X		X	X			
5-10	X	X		X		X	X			
5-17	X	X		X		X	X			
5-24	X	X	X	X	X	X	X			
6-1	X	X	X	X		X	X			
6-3	X									
6-8	X	X	X	X		X				
6-14	X	X	X	X		X	X			
6-22	X	X	X	X	X	X	X			
6-28	X	X	X	X		X	X			
7-1								X	X	X
7-6	X	X	X	X		X	X			
7-12	X	X		X		X	X			
7-15						X				X
7-20	X	X	X	X	X	X				

Parameters[1]

Table 14 (continued).

Date	D.O., Temp. (every meter)	TP (every meter)	TP (0,2,8 m sites 2,3)	Chl-a (hose sample)[2]	Phytoplankton (hose sample)[2]	Secchi Disc Transparency	Cl^- (every 2 m)	TKN, NH_4-N, NO_3+NO_2-N (every meter)	pH, T-Fe T-Mn (1 7,8,9,10m)	Cond., Alk. T-Hard, SiO_2, SO_4 (1, 10 m)
7-27	X	X		X		X	X			
7-29								X	X	X
8-2	X	X		X		X	X			
8-9	X	X		X		X	X			
8-20	X	X		X		X	X			
8-24	X	X		X	X	X				
8-26	X	X		X		X	X			
9-2	X	X		X		X	X			
9-9	X	X		X		X	X			
9-16	X	X		X		X	X	X	X	X
9-23	X	X		X	X	X	X			
9-28	X	X		X		X	X			
10-6	X	X		X		X	X			
10-15	X	X		X		X	X			
10-21	X	X		X	X	X	X	X	X	X
10-27	X	X		X		X				
11-4	X	X		X		X				
11-10	X	X		X		X				
11-17	X	X		X	X	X				
12-1	X	X		X		X		X	X	X
12-7	X	X		X	X	X				

[1] all samples were collected at site 1 (Figure 9) unless otherwise indicated

[2] hose samples were taken to twice the Secchi disc transparency depth

Table 15. 1983 lake sampling schedule and parameters.

Date	D.O., Temp. (every meter)	TP (every meter)	TP (0,2,8m; sites 2,3)	Chl-a (every meter)	Phytoplankton (hose sample)[2]	Secchi Disc Transparency	TKN, NH_4-N, NO_3+NO_2-N (every meter)	T-Fe, D-Fe, T-Mn, D-Mn, DP, pH (2, 4, 6, 7, 8, 9, 10 m)[3]	Ca, Mg, Na, K, Cl, SO_4, Alk, Cond., pH[4], SiO_2, TP, DP, NH_4-N, NO_3+NO_2-N (3 site hose composite)
2-17	X	X		X	X				
3-15	X	X		X					
4-14	X	X	X	X	X	X			X
4-21	X	X		X		X			
4-29	X	X		X		X			
5-5	X	X		X		X			
5-12	X	X		X	X	X			
5-19	X	X	X	X		X	X	X	
5-27	X	X		X		X			
6-2	X	X		X		X			
6-9	X	X		X					
6-15	X	X	X	X	X	X	X	X	
6-22	X	X		X		X			
6-29	X	X		X		X			
7-5	X	X		X		X			
7-13	X	X		X	X	X			
7-19	X	X	X	X		X	X	X	
7-28	X	X		X		X			
8-5	X	X		X		X			
8-9	X	X		X		X			
8-17	X	X	X	X	X	X	X	X	

Table 15. (continued).

Date	D.O., Temp. (every meter)	TP (every meter)	TP (0,2,8m; sites 2,3)	Chl-a (every meter)	Phytoplankton (hose sample)[2]	Secchi Disc Transparency	TKN, NH_4-N, NO_3+NO_2-N (every meter)	T-Fe, D-Fe, T-Mn, D-Mn, DP, pH (2, 4, 6, 7, 8, 9, 10 m)	Ca, Mg, Na, K, SO_4, Alk, Cond., pH, SiO_2, TP, DP, NH_4-N, NO_3+NO_2-N (3 site hose composite)
8-23	X	X		X		X			
8-31	X	X		X					
9-7	X	X		X		X			
9-14	X	X		X	X	X			
9-20	X	X	X	X		X			
9-28	X	X		X		X	X	X	
10-5	X	X		X		X			
10-12	X	X		X	X	X		X	
10-19	X	X	X	X		X		X	
10-27	X	X		X		X		X	
11-2	X	X		X		X		X	
11-9	X	X		X		X		X	
11-16	X	X		X	X	X		X	
11-29	X	X		X		X			
12-29	X	X		X	X				

[1]all samples were collected at site 1 (Figure 9) unless otherwise indicated

[2]the phytoplankton hose samples were taken twice the Secchi disc transparency depth

[3]these parameters sampled only at 3, 6 and 9 m after 10/12/83.

the lower end with an attached rope, and emptying the hose into a clean bucket. The water was thoroughly mixed in the bucket and then transferred to a sample bottle. Temperature measurements were made with a YSI tele-thermometer. During 1982 and 1983, tele-thermometer measurements were calibrated on each sampling date with a standard mercury thermometer accurate to ±0.1°C. Daily calibration was not practiced during 1981, so temperature data from that year is probably less accurate.

Lake Volume and Phosphorus Mass Estimation

A staff gage attached to the face of the dam at the lake's south end was used to monitor lake volume fluctuations. A record of staff gage measurements is given in Appendix A. A staff gage reading of 18.72 ft. was selected to represent the lake surface elevation which corresponded to the lake volume estimate derived from the bathymetric map (Figure 1). Variations in staff gage readings from this datum were used to quantify lake volume fluctuations.

Lake phosphorus mass estimates were made by applying concentrations of phosphorus samples taken at one meter intervals to their associated strata volumes (Table 1). Confidence intervals for lake phosphorus mass estimates were determined from the variability of total phosphorus sample analyses. Samples were collected at 0, 2, and 8 m depths at three sites in the lake on seven occasions during 1983. Samples from each site were analyzed on separate dates to maximize potential analytical variability. A plot of the log of concentration variance vs. the log of concentration mean was developed. Additional high concentration points were added to the plot from two and three site sample data collected from various lakes in Vermont through the Department of Water Resources spring phosphorus sampling program. This was done since few high concentration samples existed in the original data set. A regression line for the plot

was then used to estimate variances for the mean phosphorus concentrations of the strata. A volume weighted standard deviation for the total lake phosphorus mass estimate was determined from strata variances using the following formula:

$$w_i = \frac{v_i}{v_t} \qquad s = \frac{s_i^2 \, w_i^2}{n_i}$$

where: v_i = volume of stratum i
v_t = total lake volume
s_i^2 = variance of the mean P concentration of stratum i
n_i = number of samples used to determine the mean P concentration of stratum i
s = standard deviation of the total lake P mass

Ninety-five percent confidence intervals were estimated as ±two standard deviations. The confidence intervals thus determined were used to evaluate the significance of week to week changes in lake phosphorus mass. They do not account for inaccuracies in lake volume estimation. Variances of mean phosphorus concentrations of hypolimnetic strata may have been underestimated during periods of peak phosphorus concentrations due to the small number of truly representative three site samples taken. The confidence intervals were probably most accurate for epilimnetic phosphorus mass and mean concentration estimates.

Evaluation of a second set of three site samples collected during 1982 produced very similar variance estimates although separate dates of analysis for separate sites was not practiced. Variance estimates produced from the 1983 data set were used in confidence interval estimates for all three years of lake phosphorus data.

Phosphorus Vertical Transport Estimation

A vertical transport model described by Stauffer and Lee (1974) was used to evaluate the magnitude and timing of phosphorus gains in the epilimnion resulting from periodic, wind-

induced episodes of metalimnetic and hypolimnetic erosion during the thermally-stratified summer period. Temperature and phosphorus concentration profiles prior to a period of downward migration of the epilimnetic boundary, and temperature profiles after the period were used to estimate vertical transport of phosphorus. Phosphorus was assumed to have the same turbulent exchange coefficient as heat. Epilimnetic boundaries were determined from relative thermal resistance to mixing (RTRM) values (Kuether, 1980). An RTRM of 30 defined the epilimnetic and hypolimnetic boundaries.

The phosphorus vertical transport estimates probably have low levels of accuracy due to various limitations of the data used. Profile data was collected on a weekly basis during which time significant net heat losses or gains may have occurred. Also, profile data was largely based on samples and measurements from a single site, and so provided only approximations of true lake-wide mean values for the needed parameters. Although such limitations were recognized, the phosphorus vertical transport estimates were useful in identifying periods of transport, providing rough quantification of transport, and studying a major mechanism influencing epilimnetic phosphorus concentrations during the thermal stratification period.

Surface Flows

Flow Measurement

V-notch weirs were installed on 13 inflowing streams (W-1, W-2, W-4, W-5, W-6, W-8, W-10, W-11, E-1, E-2, E-6, E-10, and E-11; see Figure 3) during the summer and fall of 1982 to facilitate flow measurements. Stage-discharge rating curves were developed for culverts on two additional streams (W-3 and W-9). The stream at the north end of the lake (W-O) could not be effectively rated due to beaver activity, so instantaneous flow measurements were made with a Marsh-McBirney current meter. The remaining surface flows were too small to justify regular monitoring.

During 1983, continuous flow recorders were operated at weirs for two streams (W-5 and E-10). Mean daily flows were derived from the continuous flow records for these streams. Estimated mean daily flows from stream W-6 were added to the mean daily flows from stream W-5 to produce a combined W-5/W-6 basin mean daily flow record. Divergent streamflow prevented basin separation. Estimates of mean daily flow for stream W-6 were made by applying regression relationships of instantaneous flows between the two streams to the mean daily flow record for stream W-5. Five regressions between stream W-5 and stream W-6 were applied over the course of the year since some major flow events altered the distribution of flow at the point of divergence. R^2 values for all regressions exceeded 0.97.

Instantaneous flow measurements were made on streams without continuous recorders at least twice weekly during March-May and November, and approximately weekly for the remainder of the year. Mean daily flow estimates for these streams were made by relating their instantaneous flow measurements to mean daily flow hydrographs (log of mean daily flow vs. day) of the streams with continuous flow records, through graphical overlays. Generally, a best fit was obtained for all high flow periods at a point where flows were basin areal proportional. Adjustments to this basin areal proportional relationship were made for some streams during the summer and early fall low flow period (see Table 16). Flows for drainage areas not monitored (12% of the total watershed land area) were assumed to be basin areal proportional to combined W-5/W-6 basin flows.

During most of 1982, no continuous flow records existed and only instantaneous flow measurements were made on inflowing streams, usually at the time of sample collection. To develop estimates of mean daily flows for Lake Iroquois streams during 1982, mean daily flow records from a nearby stream were used (4.5 mi. west; station number 3 in the Laplatte River watershed study being conducted by the University of Vermont). This stream has a watershed size (153 ha) similar to E-10 (123 ha), but has more clay soils and a flashier flow regime than E-10. From January 1

Table 16. 1983 mean daily flow determination guidelines.

Stream	Base Stream	Dates	Conversion Factor[1]
W-0	E-10	all	1.987
W-1	W-2[2]	all	1.794
W-2	W-5	1/1-6/5	1.528
		6/6-9/30	0.226
		10/1-11/4	4.088
		11/5-12/31	1.528
W-3	W-5	all	0.157
W-4	W-5	all	1.006
W-8	W-5	all	0.755
W-9	W-5	all	0.918
W-10	E-10	1/1-6/29	0.272
		6/30-7/30	0.187
		7/31-8/10	0.033
		8/11-8/31	0.187
		9/1-9/4	0.033
		9/5-9/30	0.187
		10/1-11/3	0.041
		11/4-12-31	0.272
W-11	E-10	1/1-6/8	0.166
		6/9-7/30	0.750
		7/31-8/10	0.269
		8/11-9/20	0.750
		9/21-9/23	0.269
		9/24-10-4	0.750
		10/5-12/31	0.166
E-1	W-5	1/1-6/6	3.415
		6/7-10/4	0.943
		10/5-12/31	3.415
E-2	W-5	1/1-9/30	1.245
		10/1-10/13	0.000
		10/14-12/31	1.245
E-11	E-10	1/1-6/8	0.926
		6/9-7/30	4.238
		7/31-8/10	1.548
		8/11-9/20	4.238
		9/21-9/24	1.548
		9/25-10/4	4.238
		10/5-11/4	2.069
		11/5-12/31	0.926
east shore unsampled areas	W-5	all	3.604
west shore unsampled areas	W-5	all	2.805

[1] base stream flow x conversion factor = stream flow
[2] as determined from W-5

to August 31, 1983, a common record of mean daily flows existed for the Laplatte basin stream and for stream E-10. A regression of the square root of the mean daily flows for the two streams was developed ($R^2 = 0.73$). This regression appeared superior to either flow vs. flow, or log flow vs. log flow regressions, which were also evaluated. To minimize the effects of a slight non-linearity in the relationship, separate high and low flow regressions were developed. These were applied to the 1982 mean daily flow record of the Laplatte basin stream to produce an estimated 1982 mean daily flow record for stream E-10. Mean daily flows for other Lake Iroquois streams were then assumed to be basin areal proportional to the estimated E-10 flows.

Outflow estimates for 1983 are based on a series of rating curves developed for both a lake level staff gage attached to the face of the dam and an in-stream staff gage below the dam. Instantaneous flow measurements used to develop the rating curves were made with a Marsh-McBirney current meter. Instantaneous flow measurements or staff gage readings were made at least twice weekly during March-May and November and approximately weekly for the remainder of the year. Readings during the peak spring and fall flows were made on a daily basis, with the assistance of a lakeshore resident. Instantaneous flow measurements and staff gage readings were assumed to represent mean daily flows in order to produce mean daily outflow record. Mean daily outflow for days lacking measurements were determined by interpolation.

Instantaneous outflow measurements were also made during 1982. However, efforts towards developing rating curves during the first half of the year were unsuccessful due to changes in stream channel configuration. Therefore, outflow during 1982 was estimated as a residual in short-term (5 day) water balances.

Surface Flow Sampling

Fifteen inflowing streams (W-0, 1, 2, 3, 4, 5, 6, 8, 9, 10, 11, E-1, 2, 10, 11; see Figure 3) and the outlet were sampled regularly between March 24, 1982 and December 28, 1983.

The remaining smaller streams were sampled occasionally to allow comparisons to the regularly sampled streams. Grab samples were collected approximately twice weekly during high flows and weekly during low flows. Sampling was scheduled to coincide with high flows as much as was feasible. Samples were collected and analyzed for total phosphorus and dissolved phosphorus. During 1982, stream samples were also analyzed for chloride. Dissolved phosphorus and chloride samples were filtered in the lab on the day of collection using 0.45 um membrane filters and GF/A glass fiber filters, respectively. A complete list of stream sampling results is available on request.

Loading Estimation

Total phosphorus and dissolved phosphorus stream loads for each year were determined through a stratified sample technique with groups based upon flow interval (Walker, 1983). Flow interval boundaries for the mean daily flow record for each stream were established by two to five equally spaced intervals on a square root of flow scale. The maximum mean daily stream flow for each year determined the range of the scale. A flow-weighted mean concentration for each interval was calculated from those mean daily flows in each respective interval that had associated sample concentrations. This flow weighted mean concentration was then applied to all mean daily flows within the interval to produce daily loadings. Daily loadings were summed to produce short-term and annual loading estimates. Variance estimates for each flow interval's loading contribution and for total annual loading were also calculated.

Surface inflow loading estimates for the unsampled areas of the watershed were based on loading data from representative watershed streams. Flow-weighted mean phosphorus concentrations for the flow intervals of stream W-4 were applied to similar flow intervals for the unsampled west shore areas. Flow interval phosphorus concentrations from stream E-11 were applied to similar flow intervals for the unsampled east shore areas.

Surface flows from unsampled areas were assumed to be basin areal proportional to continuously monitored streamflows, as mentioned previously.

Chloride loadings from streams were calculated for 1982 by integrating instantaneous loading measurements over time. Instantaneous loading was determined by multiplying instantaneous flow by the chloride concentration. Instantaneous flow measurements made on stream W-0 were used to estimate flows at all sites by assuming basin areal proportionality. No attempt was made to estimate the additional unsampled loading which occurred between January 1 and March 23. Chloride loadings from all sampled streams were composited (adjusting for volume) to produce a mean chloride concentration for inflow.

Lake phosphorus losses through surface outflow were determined from mean daily outflow values and associated total phosphorus concentrations of outflow samples. Phosphorus concentrations for unsampled dates were estimated by interpolating between concentrations found on sampled dates.

Groundwater

Wells

A total of 43 monitoring wells were installed around the perimeter of the lake (see Figure 10 for locations). Eleven wells (groups A and C below) were installed in October of 1982. The remaining wells (group B) were installed during the early summer of 1983. Materials used to construct the wells are described below.

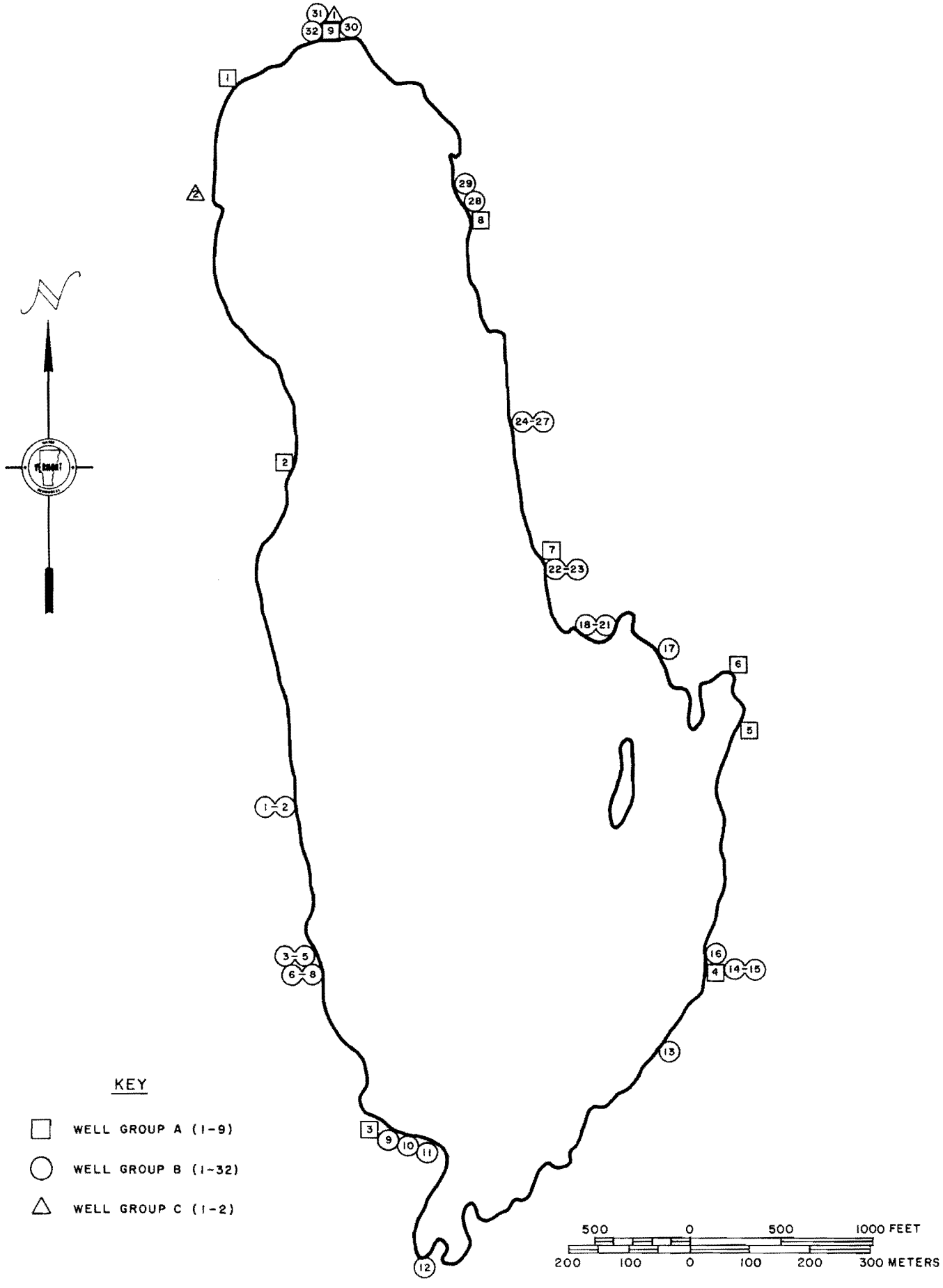


Figure 10. Well locations.

WEll GROUP	WELL NUMBERS	MATERIALS
A	1-9	casing - 0.68 cm inside dia. flexible vinyl tubing screen - casing is joined to a 20.3 cm long slotted polyethylene tube (1.3 cm inside dia.) wrapped with fiberglass filter fabric; lower end is sealed with a rubber stopper
B	1-32	casing - 1.3 cm inside dia. P.V.C. pipe screen - lower 91.4 cm* of casing is perforated and wrapped with fiberglass filter fabric; lower end is sealed with a rubber stopper *(wells 4, 20, and 26 have 30.5 cm screens)
C	1-2	casing - 10 cm inside dia. P.V.C. pipe screen - lower 30.5 cm of casing is slotted and wrapped with fiberglass filter fabric; lower end is sealed with a P.V.C. end cap

Well Numbers C-1 and C-2 were installed with a hand operated soil auger. All other wells were installed using a 2 or 3 meter section of 1.9 cm inside dia. steel pipe. Two lag bolts were loosely fitted into each end of the steel pipe which was then hammered into the soil to the desired depth (the top bolt protected the pipe from being damaged by the sledge hammer; the bottom bolt prevented the pipe from being filled with soil as it was driven). The top bolt was then removed and the well pipe was inserted into the steel pipe. The well pipe was then held in place while the steel pipe was pulled out. The bottom bolt remained in the soil below the well (technique from Lee, 1978).

Most group A and B wells were installed at a depth of approximately 1.2 meters below the ground surface at a distance less than 1 meter from the lake high water line. Group A wells were located in sites which were felt to be relatively unaffected by septic systems and thus represented background conditions.

Most group B wells were located down gradient from septic systems. These wells were used to evaluate the movement of septic system pollutants to the lake. Well numbers B-4, B-21, and B-26 were installed somewhat deeper (2-3 m) and had shorter screens to provide some indication of depth related variations in pollutant concentrations (See Appendix B for individual well descriptions.)

Group C wells were installed at a depth of approximately 2 meters below the ground surface at a distance of about 50 meters from the shoreline. These wells were also felt to be representative of background conditions. Weekly water level measurements were made on these wells to provide a record of water table fluctuations (Appendix C).

Most group B wells were sampled every two weeks from late June to the end of October during 1983. Generally, other wells were sampled at least monthly during the summer of 1983. Some wells were also sampled during 1982 (See Appendix D for a list of sampling dates for individual wells.)

Wells were sampled for total dissolved phosphorus, chloride, nitrate and nitrite, ammonium, and specific conductivity. Results are listed in Appendix D. A battery powered peristaltic pump fitted with an in-line filter was used to collect samples. Wells were flushed prior to sample collection by pumping until dry or until three well volumes of water were removed. Phosphorus and nitrogen samples were field filtered through a 0.45 um membrane filter. Filters were flushed with at least 10 ml of water prior to sample collection. Chloride and conductivity samples were filtered in the lab through a GF/A glass fiber filter on the day of collection.

Piezometers

Forty-two lake bed piezometers were installed around the perimeter of the lake during the summer of 1982 (Figure 11). Piezometers were constructed identically to group A wells and were installed with a steel pipe in the same manner as group A

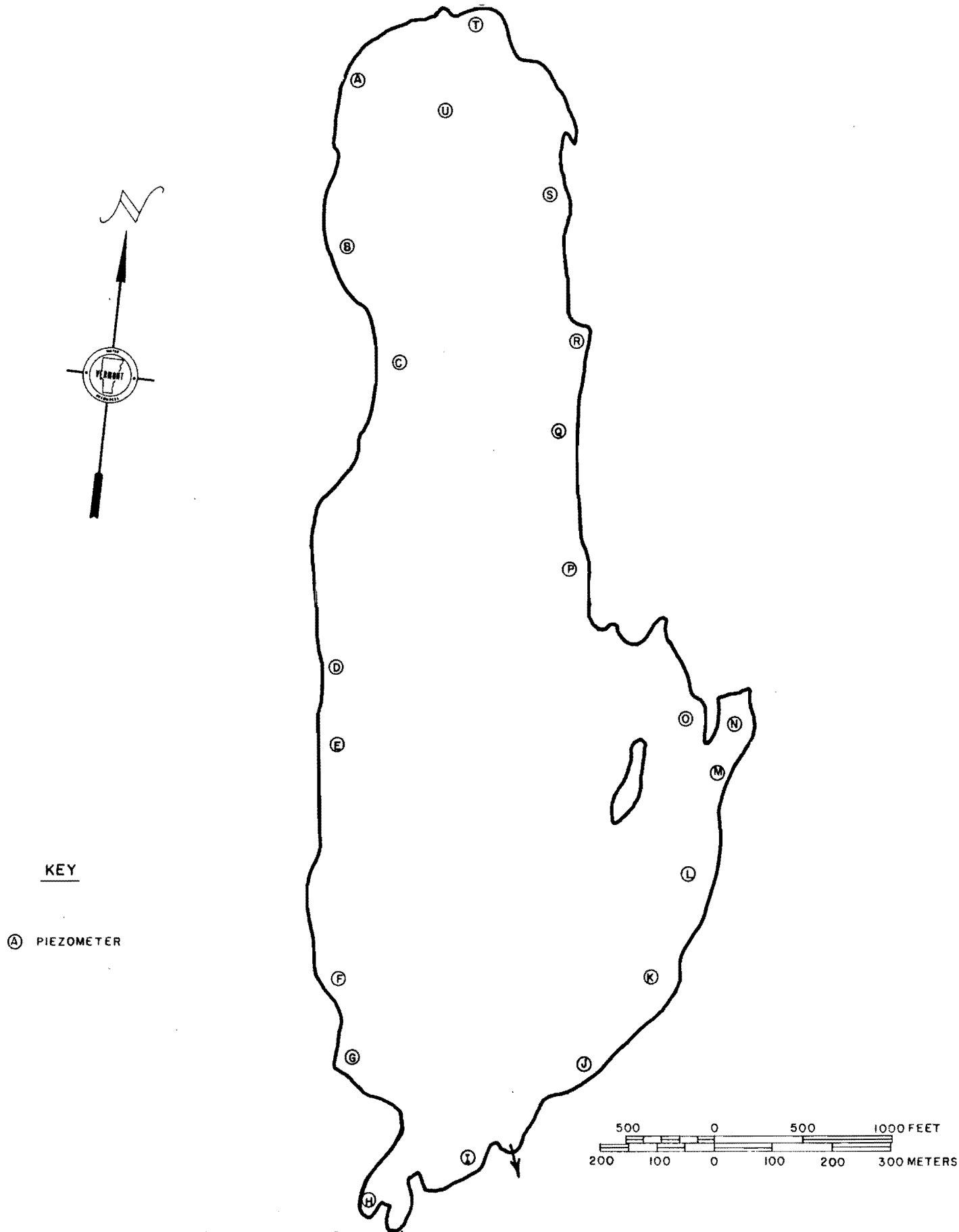


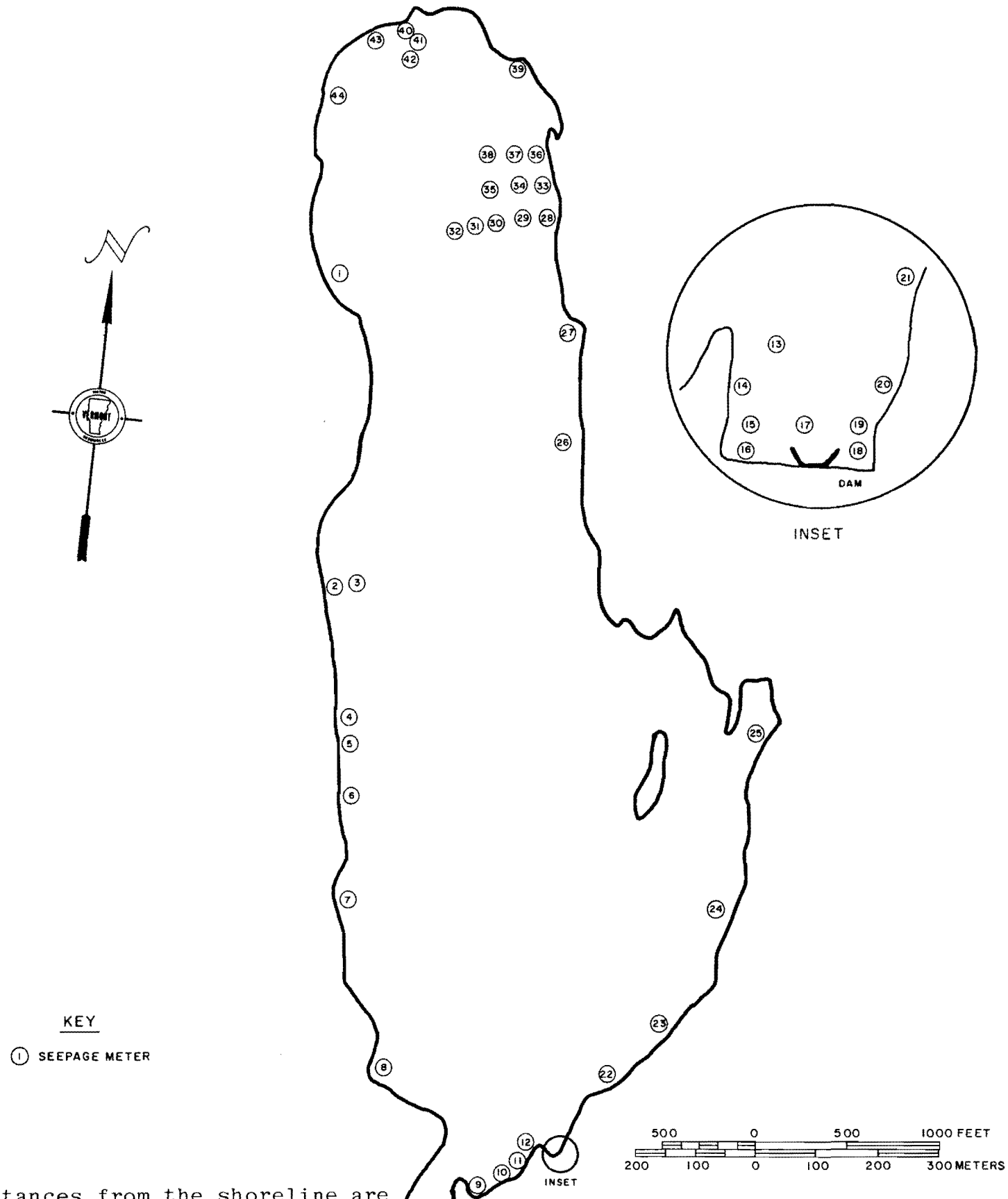
Figure 11. Piezometer locations.

and B wells. Piezometer location descriptions are given in Appendix E. The transparent piezometer casings were extended above the lake surface and were supported by wooden stakes. An additional section of casing tubing (0.4 m length) which extended above and below the lake surface was attached parallel to the main casing. This tube was used to provide a lake level reading. Piezometer head readings were made after extended periods of lake calm. Readings were made only twice during 1982 (see Appendix F) since plans to use these measurements in conjunction with seepage meter measurements to estimate ground water inflow were altered during the course of the study.

Seepage Meters

Seepage meters were operated at 44 sites in the lake during the summer of 1982 (Figure 12). Site descriptions are given in Appendix G. Seepage meters were constructed and operated as described by Lee (1977). Briefly, the seepage meters used were end sections cut from 55 gal steel drums (section ht. = 15 cm). The drum section was then placed at a site by pushing it part way into the lake bottom sediment in an inverted position. A one-hole rubber stopper bearing a short section of polyethylene tubing was inserted in a hole in the drum section face. A plastic bag containing a measured volume of water was attached to the polyethylene tube, and after a measured time period was removed and the water volume remeasured. This provided a measure of the rate of water movement out of or into the lake bottom within the area sealed by the drum section. Measurements made for individual seepage meters are given in Table 21. Three sets of seepage meters which were located along the northeast shore (meters 28-38, Figure 12) were installed to examine the width of the shoreline seepage zone and seepage variation with distance from shore.

Some additional seepage meter operation was conducted during 1983 along the lake's southeast shore to evaluate highly variable results obtained in this area during 1982. One concern was that



Distances from the shoreline are not true to scale.

Figure 12. Seepage meter locations.

coarser substrate materials along this shoreline might have prevented adequate sealing of the meters and resulted in variable responses due to lake water movement. Four meters (numbers 2-5, Table 22) were placed near the former site of meter 22 (Figure 12). Two of these were left resting on the bottom surface to demonstrate the behavior of an ineffectively sealed meter. A fifth (number 1) meter was placed about 50 m northeast of the site. All meters were placed 1-3 meters from the shoreline at depths of 0.3-0.4 meters. Measurements made for these meters are given in Table 22.

Groundwater Inflow Estimation

Initial plans to divide the shoreline into distinctive seepage zone segments and to determine seepage relationships to water depth or distance from shore within each segment were found to be impractical due to the high variability of seepage encountered. Large areas of substrate unsuitable for seepage meter operation also existed. Because of these problems a more simplified approach was taken. Mean seepage rates for individual seepage meters (numbers 1-12 and 22-44, Figure 12) were determined. All suspect measurements were rejected. The mean seepage rates from all meters were then averagéd to produce a mean rate.

The width of the seepage zone was estimated to be 10 m based on measurements made at three sets of seepage meters operated along the northeast shore, which were placed at varying distances from the shoreline. General observations indicated that this value was also suitable for most sections of the shoreline. The shoreline perimeter length was estimated by measuring a series of line segments around the lake, thus removing the effects of minor shoreline irregularities. The estimated area of the seepage zone was multiplied by the mean seepage rate to produce an average total groundwater inflow rate for the lake. A second estimate made by dividing the seepage zone into six sections and applying mean seepage rates from each section produced a similar value for

the average total groundwater inflow rate. A third estimate of groundwater inflow was made as a residual term in the annual water budget for 1983. Groundwater outflow was felt to be quite small and was assumed to be zero. The groundwater inflow estimate thus determined was fairly similar to that obtained from seepage meter measurements and was used in the water budget and phosphorus loading calculations for both years.

Groundwater Phosphorus Loading Estimation

Groundwater phosphorus concentration was determined from well samples for dissolved phosphorus. A phosphorus concentration for inflow areas not influenced by wetland seepage was based on the mean of results from eight wells (numbers A-1,2,3,6,8, B-32 and C-1,2). A phosphorus concentration for inflow areas influenced by wetland seepage was based on the mean of results from six wells (numbers A-5,9 and B-12,17,30,31). The percentage of total groundwater inflow represented by the wetland seepage influenced wells was estimated from the percent of surface drainage areas upgradient from these wells (10% of the total drainage area). A flow weighted mean phosphorus concentration for groundwater was then calculated from the two separate means.

The annual groundwater phosphorus loading estimate was based on the mean phosphorus concentration estimate and the annual groundwater inflow estimate determined from the 1983 water budget. This annual loading estimate was applied to both years. Groundwater phosphorus loading derived from septic systems has been evaluated separately. For the 1983 short-term phosphorus budget, groundwater loading was distributed on the basis of well head measurements. Monthly mean well head above lake level was determined (Appendix C) and monthly loadings were assumed to be proportional to these values. These head difference values were approximately proportional to the hydraulic gradients since seasonal differences in the distance between the well and shoreline were small. For 1982, well head measurements were

unavailable, so loading was assumed to occur at a constant rate over the course of the year.

Precipitation

During May-October, 1982, and June-November, 1983, daily precipitation measurements were made at sites within the Lake Iroquois watershed by local residents (1982 - 0.9 mi. north of the lake, 1983 - 300 yards west of the lake). Standard 20.3 cm diameter rain gages mounted on platforms 1.5 m above the ground were used. During the remaining portions of the two years, precipitation records from a recording precipitation gage located in Hinesburg approximately two miles southwest of the lake were applied (gage no. 3 operated by the University of Vermont as part of the LaPlatte River watershed study).

Areal phosphorus loading from bulk precipitation was estimated from literature values. Three references which seemed most applicable to Vermont are described below.

	Bulk Precipitation Areal P Loading (Kg P/ha/yr.)		
Reference	Mean	Range	Comments
Pearson and Fisher (1971)	0.15	0.10-0.49	Based on 9 sites in the northeastern U.S. during a single year.
Morgan *et al.* (1984)	0.18	--	Based on a 2 year sampling program at Lake Morey (Town of Fairlee, Vermont). This value is something less than bulk pptn. since cotton plugs were used in collector openings.
Scheider *et al.* (1979)	0.36	0.06-0.77	Based on 11 Ontario studies cited.

Other references reviewed showed similar ranges of values, but sampling locations and methods were less comparable. A bulk precipitation areal phosphorus loading rate of 0.30 kg P/ha/yr was selected and applied to Lake Iroquois for both 1982 and 1983. For short-term estimates, loading was distributed proportionally

to the percent of total annual precipitation which occurred during a given time interval.

Evaporation

Lake surface evaporation was estimated from Class A evaporation pan records collected at a NOAA observation station at Essex Junction, Vt., eight miles north of the lake. A pan to lake coefficient of 0.75 was applied to daily pan evaporation measurements for the May-October periods of record during 1982 and 1983 (Kohler, et al., 1959). Applying this annual coefficient (0.75) to daily values distorted the seasonal distribution of evaporation since the actual coefficient varies over the May-October period. However, this was felt to have a relatively minor effect on the water budget evaluations conducted. Evaporation during the remainder of each year was estimated to be 20% of the total May-October figures (Kohler, et al., 1959). This was assumed to occur at a constant rate for purposes of short-term distribution.

Water Sample Analysis

Sample collection and filtration procedures for lake, stream, and groundwater samples have been described in their respective methodology sections. Chemical analysis methods are listed in Table 17. Chlorophyll samples were filtered onto GF/A glass fiber filters and frozen prior to analysis. The filters were then ground, chlorophyll was extracted in 90% acetone, analyzed fluorometrically for chlorophyll-a, and corrected for the presence of pheophytin (American Public Health Association, 1981).

Samples for phytoplankton analysis were preserved in the field with Lugol's solution. Organisms were enumerated using the inverted microscope technique (Lund et al., 1958), and identified using a variety of standard taxonomic keys. Phytoplankton biovolume estimates were made according to American Public Health Association (1981).

Table 17. Chemical analysis methods.

Parameter	Analytical Method
Phosphorus	Autoclave digestion with acid persulfate followed by an automated colorimetric ascorbic acid method (U.S. E.P.A., 1979)
Ammonium, Nitrate plus Nitrite	Automated colorimetric methods (U.S. E.P.A., 1979)
Total Kjeldahl Nitrogen	Potentiometric method (U.S. E.P.A., 1979)
Dissolved Oxygen	Azide modification of the Winkler method (American Public Health Association, 1981)
Iron, Manganese, Calcium, Magnesium, Sodium, Potassium	Atomic absorption, direct aspiration method (U.S. E.P.A., 1979)
Total Hardness	Calculation method (American Public Health Association, 1981)
Sulfate	Automated colorimetric methylthymol blue method (U.S. E.P.A., 1979)
Chloride	Automated colorimetric ferricyanide method (U.S. E.P.A., 1979)
pH	Standardized pH electrode (American Public Health Association, 1981)
Alkalinity	Potentiometric titration with sulfuric acid (American Public Health Association, 1981)
Total Silica	Digestion with sodium carbonate according to a procedure modified from Paashe (1980) followed by an automated colorimetric method (Technicon, 1976)
Conductivity	Conductivity meter (American Public Health Association, 1981)

Sediment Core Collection and Analysis

An 80 cm lake sediment core was collected in December, 1982 at a site approximately 100 m west of sampling site 1 (Figure 9). The core was collected with a 10 cm diameter plexiglass coring tube by a SCUBA diver. The coring tube was pushed into the sediment and then sealed, top and bottom, with rubber stoppers. The tube was then withdrawn from the sediment and carried to the surface. The core was sectioned into 1 cm layers as it was extruded vertically upwards using a graduated plunger apparatus. Samples were placed in plastic bags and refrigerated in the dark until they were analyzed.

Subsamples of the 1 cm layers were composited over 4 cm intervals for dating purposes. The samples were dated by Dr. Richard Brugam of Southern Illinois University at Edwardsville through Lead-210 analysis (Brugam, 1983). Sediment accumulation rates were calculated using the constant-rate-of-supply method (Appleby and Oldfield, 1978).

Samples from 25 levels in the core were analyzed by Edward Swain of the University of Minnesota for the following parameters:

weight loss at 110°C and 500°C
authigenic iron and manganese
% native chlorophyll
chlorophyll derivatives
total carotenoids

The analytical methods used are described in Engstrom and Wright (1983) and in Swain (1985).

Aquatic Macrophyte Sampling

Species and Distribution Survey

A preliminary survey to determine macrophyte distribution and species present was conducted during mid to late July of 1982. A series of transects between identifiable shoreline

points were established at the north end of the lake, and in the area east of the island along the lake's east shore (Figure 13). Observation sites were spaced at equal intervals along individual transects. At each site, water depth, species present, and relative abundance of species were recorded (Appendix H).

Macrophyte species identification had been performed earlier in the summer in accordance with Fassett (1980), so most species were readily identifiable during the survey. Species present at shallow water sites were determined by observing the area visible off the sides of the boat with the use of a face mask. Species present at deep water sites were also determined partly by this same method, since exceptional water clarity existed at the time, but a bottom drag was also employed to confirm the existence or absence of low-growing species. The drag consisted of two garden rake heads bound back-to-back with tines in opposite directions, and tied to a length of rope. Three tosses of the drag were made at each deep site.

Relative abundance of species at a site was estimated visually, and so was a strongly subjective measurement. The following relative abundance categories were applied:

abundant - 4
common - 3
sparse - 2
extremely sparse -1

It was later noted during the biomass sampling (see following subsection) that the relative abundance estimates tended to overrate tall and large-leafed species, such as Potamogeton praelongus, and underrate low-growing forms, especially Elodea Nuttallii, when compared on a biomass/area basis.

Macrophyte coverage in the area of the lake not surveyed by the transects consists of a relatively narrow band of plants around the remaining perimeter of the lake. Observations of species present and relative abundance in this area (sites 31-48, Figure 13) were made by a skin diver swimming along a path perpendicular to the shoreline, from the shore to the outer limit of macrophyte growth. The distance to the outer limit of

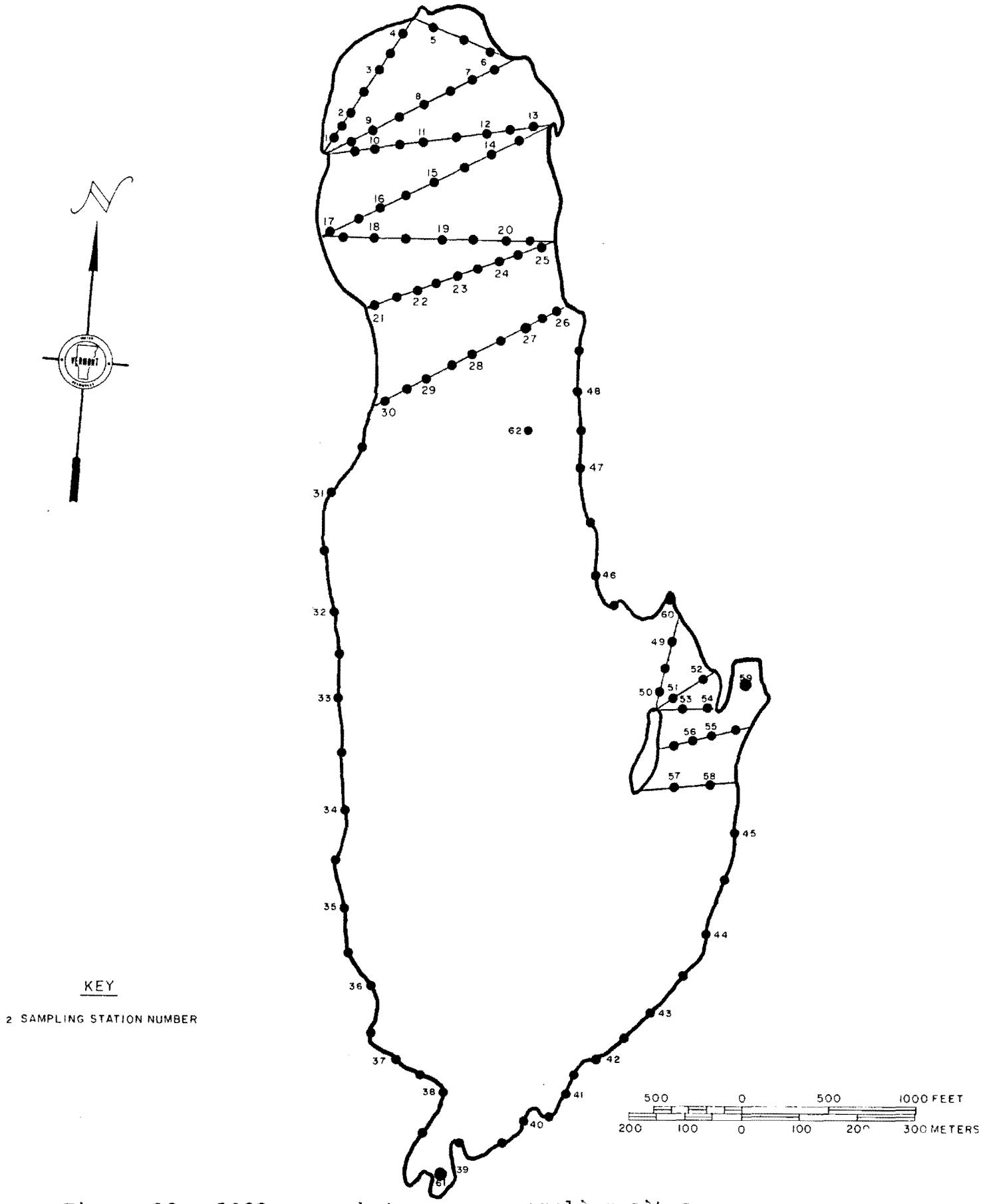


Figure 13. 1982 macrophyte survey sampling sites.

macrophyte growth was measured with a calibrated nylon rope, and the depth of water at this limit was also measured. At some sites, distinctive shallow and deep zones of macrophyte growth were evident, and widths and maximum depths were measured for both zones.

Biomass Survey

Macrophyte biomass sampling was conducted during late August and early September of 1982. The same transects and sites established during the species and distribution survey were used. However, additional sites were established mid-way between existing sites, thus doubling the sampling density. Biomass sampling was also conducted in the shallow zone (less than 2.3 m) in 1983. Sites sampled are shown in Figure 14.

A cylindrical plastic ring (dia. = 43.5 cm, ht. = 15 cm, area = 0.149 m^2) was used by a diver with SCUBA gear to collect the samples. Two random tosses of the ring were made at each site. The ring was weighted with lead, and maintained an air pocket under the upper rim, so it kept an upright position and descended quite vertically, thereby minimizing the tipping of tall macrophytes as it settled. The bottom edge of the ring also proved useful in allowing the lateral extensions of some macrophytes, Elodea Nuttallii especially, to be torn along the ring perimeter. Macrophytes within the ring were removed by hand with much of their root systems intact.

The plant samples were rinsed by agitation in lake water and then wrapped with a section of cloth mesh (sold as fruit tree protection in garden supply stores). Species composition of samples was not determined. A total of 194 samples was collected. Samples were air dried and later oven dried to a constant weight at 105°C, and weighed.

Areal coverage determinations were made using a 1:5000 scale air photo (Vermont base map series 5000 1978, sheet numbers 104204, 104208). Site locations and depths were plotted on the map, and boundaries for the transect sites were determined by

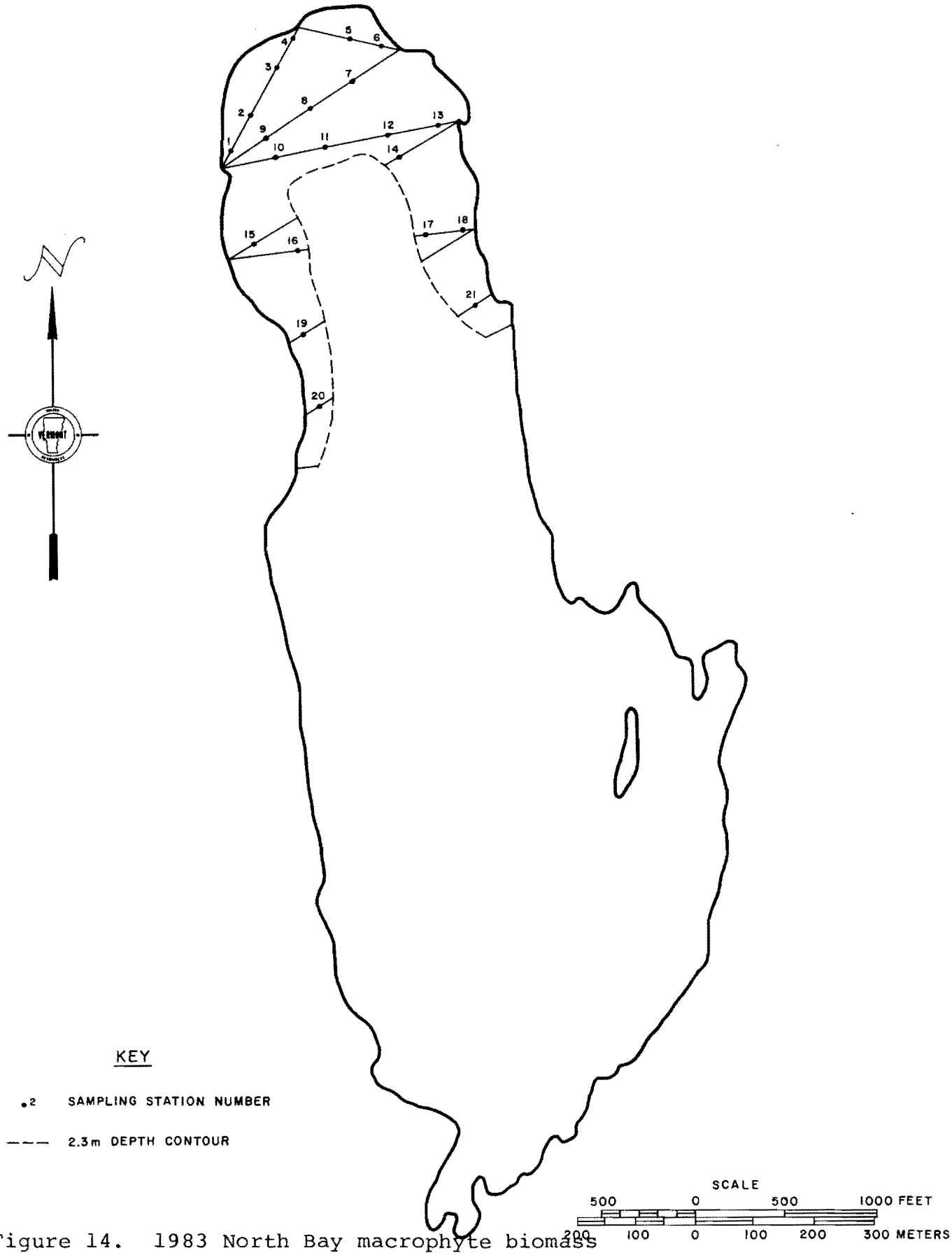


Figure 14. 1983 North Bay macrophyte biomass sampling sites.

interpolation between sites. Boundaries for the area covered by sites 31-48 (Figure 13) were based on direct measurements made during the preliminary survey. The area of coverage around the submerged island in the northeastern section of the lake was approximated by reference to the 1972 bathymetric map of the lake. Portions of the outer boundaries of coverage near the island on the east side of the lake were also partially based on this reference, combined with general observations. Areal measurements were made with a planimeter.

The area of macrophyte coverage was divided into three zones which appeared distinctive based on the ranges of macrophyte density found, as well as previous observations of depth-related zonal divisions. The mean of the square root of sample dry weights and the 95% confidence interval were determined for each zone. The square root transformation had the effect of normalizing the data distribution. The mean dry weight and 95% confidence interval for all three zones combined was computed using weighting factors corresponding to the proportion of the total area represented by each of the three zones.

Soil Sampling and Analysis

Soil samples were collected from 21 sites within the Lake Iroquois watershed (Figure 15). Many sites had bare or poorly vegetated soils which experienced high erosion rates (see Table 48 for site descriptions). Samples were oven dried at 105°C and dry sieved through #10 (2.0 mm) and #60 (0.25 mm) sieves. The coarse fraction, medium fraction, and fine fraction were weighed. Portions of the fine and medium fractions were oven dried a second time just prior to soil solution preparation.

Soil solutions were prepared by adding approximately 30 mg of the fine fractions to 200 ml of filtered (GF/A glass fiber filter) stream water collected from a Lake Iroquois tributary. Approximately 50 mg of the medium fractions were used for these sample solutions. Five duplicates and three stream water blanks were also prepared. Five samples were prepared in triplicate

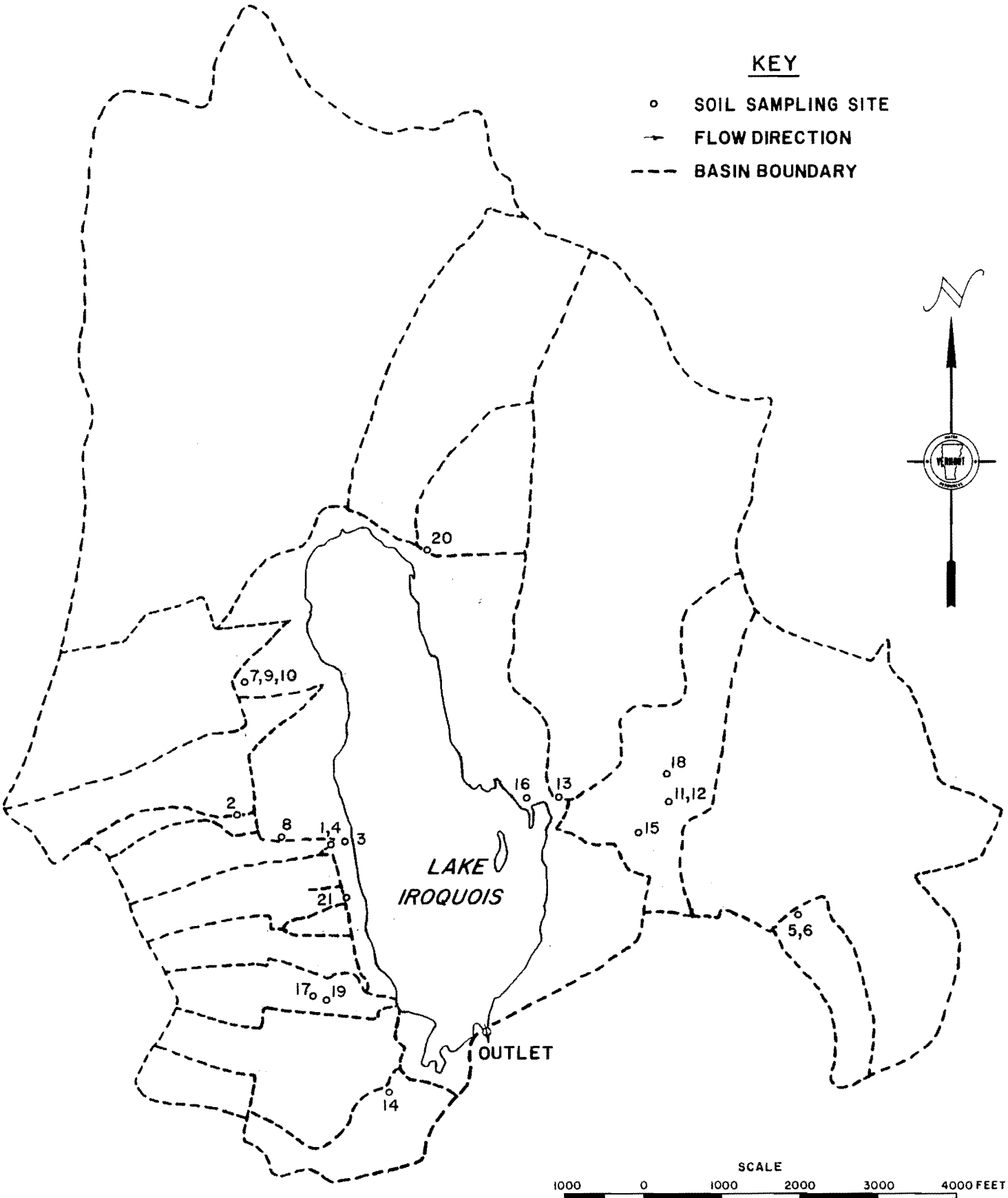


Figure 15. Soil sampling sites.

using different weights of soil to determine if soil concentration influenced analytical phosphorus yield from the solutions. The samples were analyzed for total phosphorus using the same method used for stream samples. (see Table 17).

Watershed Boundary and Land Use Determination

Watershed and sub-basin boundaries (Figure 3) were determined using U.S.G.S. topographic maps coupled with extensive field observations. A 1:5000 scale air photo (Vermont base map series 5000 1978, sheet numbers 104204 and 104208) was used in determining land use and land use boundaries, along with field verification during the summer of 1982. Basin and land use areas were determined by planimetry.

HYDROLOGIC AND PHOSPHORUS BUDGET

Surface Inflow

Surface inflow provided 67% and 76% of the lake's water supply during 1982 and 1983, respectively. The distribution of surface inflow during the two years is depicted in Figure 16. High flows during the spring of 1983 exceeded those in 1982 and were maintained over a longer period. Peak fall flows in 1983 were also far in excess of 1982 fall flows. Annual surface inflow in 1983 (604×10^4 m^3) was approximately double that of 1982 (294×10^4 m^3). This large difference between years may be due in part to the methods used to estimate the 1982 and 1983 flows. 1982 flows were estimated from a regression relationship based on flow measurements made at a nearby watershed (station no. 3 in the LaPlatte River basin study being conducted by the University of Vermont). Measurement of annual flow at this station indicated 1983 flow was about 100% greater than 1982 flow. Flow records from two U.S.G.S. gaging stations located within 25 miles of Lake Iroquois (Dog River at Northfield, Vt. - 25 miles SE; Otter Creek at Middlebury, Vt. - 25 miles SSW) showed a flow increase of only 40-50% in 1983 over 1982. Annual precipitation records for these two watersheds (Dog River - Northfield, Montpelier; Otter Creek - Salisbury, Chittenden, Rutland) showed increases of 52-60% between the two years. Annual precipitation at Burlington (7 miles NW of Lake Iroquois) showed a 55% increase between the two years. Annual precipitation for Lake Iroquois showed a 68% increase. This fact partially supports the validity of the exceptional increase in annual flow measured at station no. 3 in the LaPlatte River basin (4.5 mile W of the lake). However, the accuracy of station no. 3 flow measurement and its applicability to Lake Iroquois remains somewhat uncertain. Annual flows for Lake Iroquois may be underestimated for 1982.

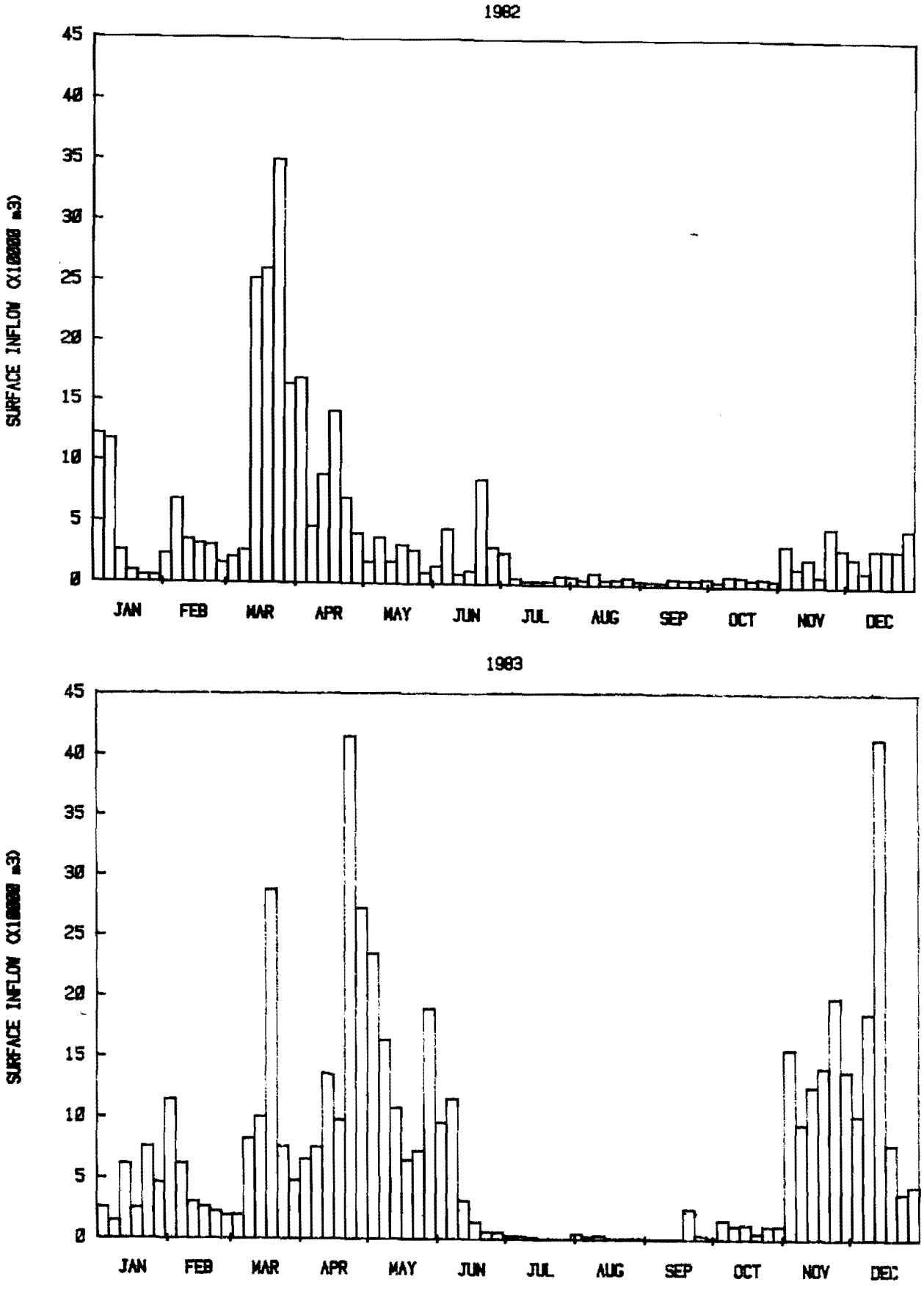


Figure 16. Surface inflow distribution (5-day intervals).

Inflow estimates for 1983 were based entirely on measurements made on Lake Iroquois watershed streams and are felt to be quite reliable. Total annual flows measured for the two streams equipped with continual flow recorders (E-10 and W-5/6) were found to be basin area proportional (within 1%).

The normalcy of total flows and monthly flow distribution for Lake Iroquois during the two years can only be roughly evaluated due to the flow differences noted for the nearest unregulated stream with long-term flow records available (Dog River). Total annual flow for the Dog River was 7% below normal in 1982 and 41% above normal in 1983. Judging from precipitation records, Iroquois flows were probably similarly below normal in 1982, but more above normal in 1983. During 1982, Dog River monthly flows fell shortest of normalcy during November and December. During 1983, Dog River spring flows were consistently above normal and November and December flows were far above normal. The December monthly total flow was the highest in 49 years of record, while the November flow was the second highest ever recorded for that month. Lake Iroquois precipitation records (Figure 20) indicate that similar variations from normal monthly flows were likely in the Lake Iroquois tributaries.

Surface inflow supplied 198 kg of phosphorus to the lake during 1982, and 358 kg during 1983. These values are 83% and 90%, respectively, of the total external phosphorus loadings for the two years. Phosphorus loadings and areal phosphorus yields from individual sub-basins are listed in Tables 18 and 19. Areal total phosphorus and dissolved phosphorus yields for watershed sub-basins are also shown in Figures 17 and 18.

Total phosphorus loading from streams during 1983 was 67% higher than in 1982. Although total streamflows were 105% greater in 1983 than 1982, the increased phosphorus loading is not directly related to increased streamflows. Two sub-basins, W-0 and W-1, showed major phosphorus loading increases during 1983, while total loadings from the remaining sub-basins were nearly equal during the two years.

Table 18. 1982 surface inflow phosphorus loadings.

SUB-BASIN	TOTAL PHOSPHORUS LOAD (Kg P/yr)	STD. DEV.	% OF TOTAL STREAM LOAD	AREAL TP YIELD (Kg P/ha/yr)	DISSOLVED PHOSPHORUS LOAD (Kg P/yr)	STD. DEV.	% OF TOTAL STREAM LOAD	AREAL DP YIELD (Kg P/ha/yr)
W-0	51.3	4.1	26.1	0.21	29.3	2.3	49.7	0.12
W-1	23.6	8.8	12.0	0.54	3.0	0.3	5.1	0.07
W-2	11.0	4.5	5.6	0.45	2.3	0.3	3.9	0.09
W-3	0.6	0.1	0.3	0.24	0.3	0.02	0.5	0.12
W-4	6.9	1.5	3.5	0.43	2.2	0.1	3.7	0.14
W-5/6	9.6	2.1	4.1	0.51	0.6	0.05	1.0	0.04
W-8	4.8	1.5	2.4	0.40	0.5	0.09	0.8	0.04
W-9	5.1	2.6	2.6	0.35	0.7	0.08	1.2	0.05
W-10	4.4	0.9	2.2	0.13	1.0	0.1	1.7	0.03
W-11	2.9	0.8	1.5	0.14	1.8	0.5	3.1	0.09
E-1	5.3	2.2	2.7	0.10	1.9	0.2	3.2	0.03
E-2	11.1	5.1	5.7	0.56	0.8	0.07	1.4	0.07
E-10	17.8	10.9	9.1	0.15	3.3	0.3	5.6	0.03
E-11	16.0	7.4	8.1	0.14	3.4	0.1	5.8	0.03
UNSAMPLED AREAS								
east shore	8.1	--	4.1	0.14	1.7	--	2.9	0.03
west shore	19.4	--	9.9	0.43	6.1	--	10.4	0.14
	197.9				58.9			

Table 19. 1983 surface inflow phosphorus loadings.

SUB-BASIN	TOTAL PHOSPHORUS LOAD (Kg P/yr)	STD. DEV.	% OF TOTAL STREAM LOAD	AREAL TP YIELD (Kg P/ha/yr)	DISSOLVED PHOSPHORUS LOAD (Kg P/yr)	STD. DEV.	% OF TOTAL STREAM LOAD	AREAL DP YIELD (Kg P/ha/yr)
W-0	118.4	13.0	33.6	0.49	70.2	7.6	66.2	0.29
W-1	111.3	29.2	30.1	2.43	3.3	0.3	3.1	0.08
W-2	10.5	1.7	3.0	0.43	2.7	0.3	2.5	0.11
W-3	1.0	0.2	0.3	0.40	0.5	0.06	0.5	0.20
W-4	11.4	3.1	3.2	0.72	2.1	0.2	2.0	0.13
W-5/6	6.6	1.5	1.9	0.42	0.9	0.07	0.8	0.06
W-8	3.2	0.8	0.9	0.27	0.7	0.13	0.7	0.06
W-9	3.5	1.0	1.0	0.24	0.6	0.04	0.6	0.04
W-10	2.1	0.3	0.6	0.06	1.1	0.1	1.0	0.03
W-11	3.4	0.3	1.0	0.17	1.9	0.2	1.8	0.09
E-1	4.2	0.3	1.2	0.08	2.6	0.2	2.5	0.05
E-2	6.2	1.9	1.8	0.31	0.9	0.07	0.8	0.05
E-10	14.2	3.6	4.0	0.12	4.2	0.3	4.0	0.03
E-11	20.2	5.1	5.7	0.18	5.9	0.4	5.6	0.05
UNSAMPLED AREAS								
east shore	9.6	--	2.7	0.17	2.7	--	2.5	0.05
west shore	31.8	--	9.0	0.71	5.8	--	5.5	0.13
	357.6				106.1			
W-O-A[1]	94.0	--	--	2.19	49.3	--	--	1.15

[1]This is a portion of sub-basin W-0. See Figures 17 and 18.

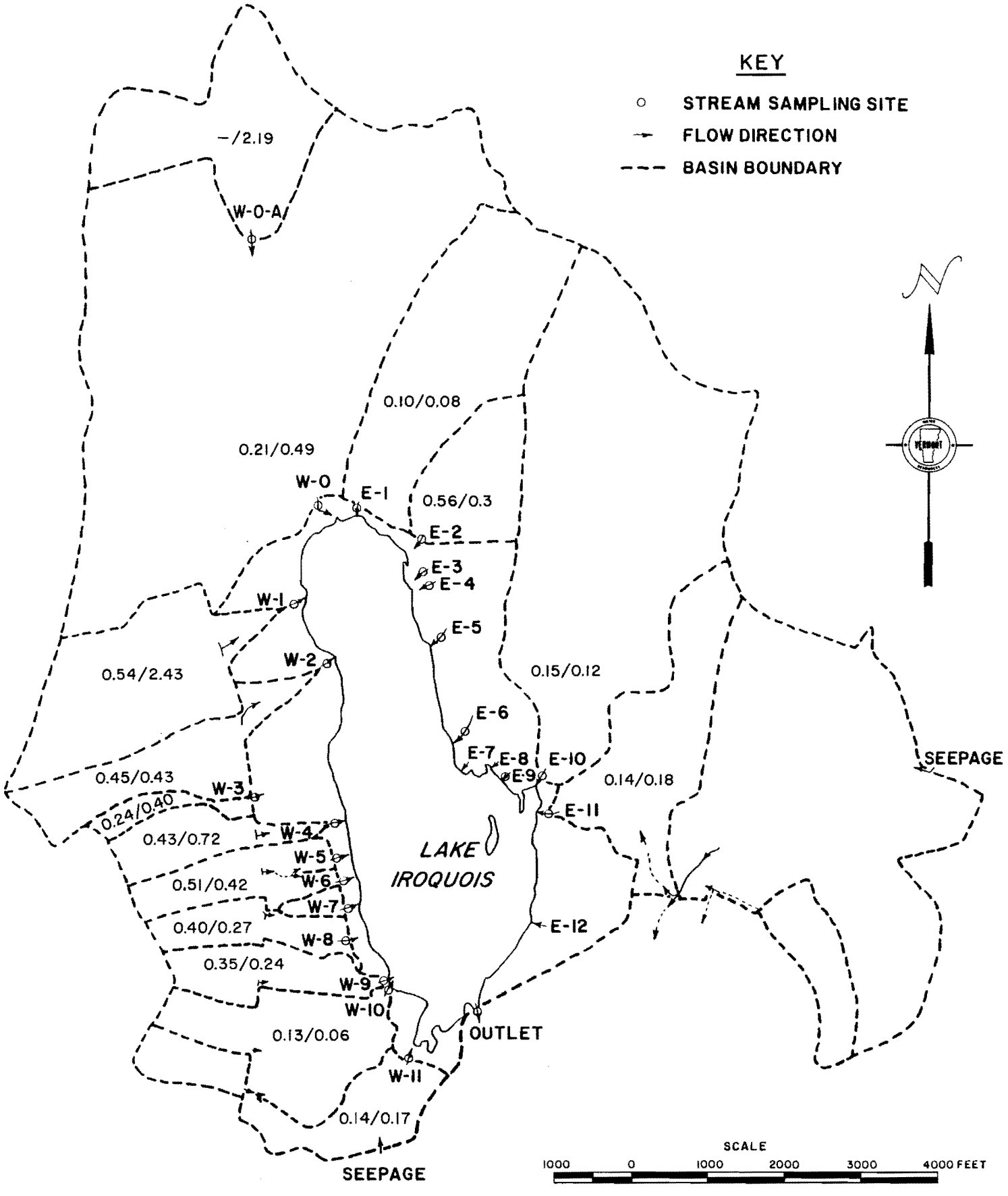


Figure 17. Total phosphorus yields during 1982/1983 (kg P/ha/yr).

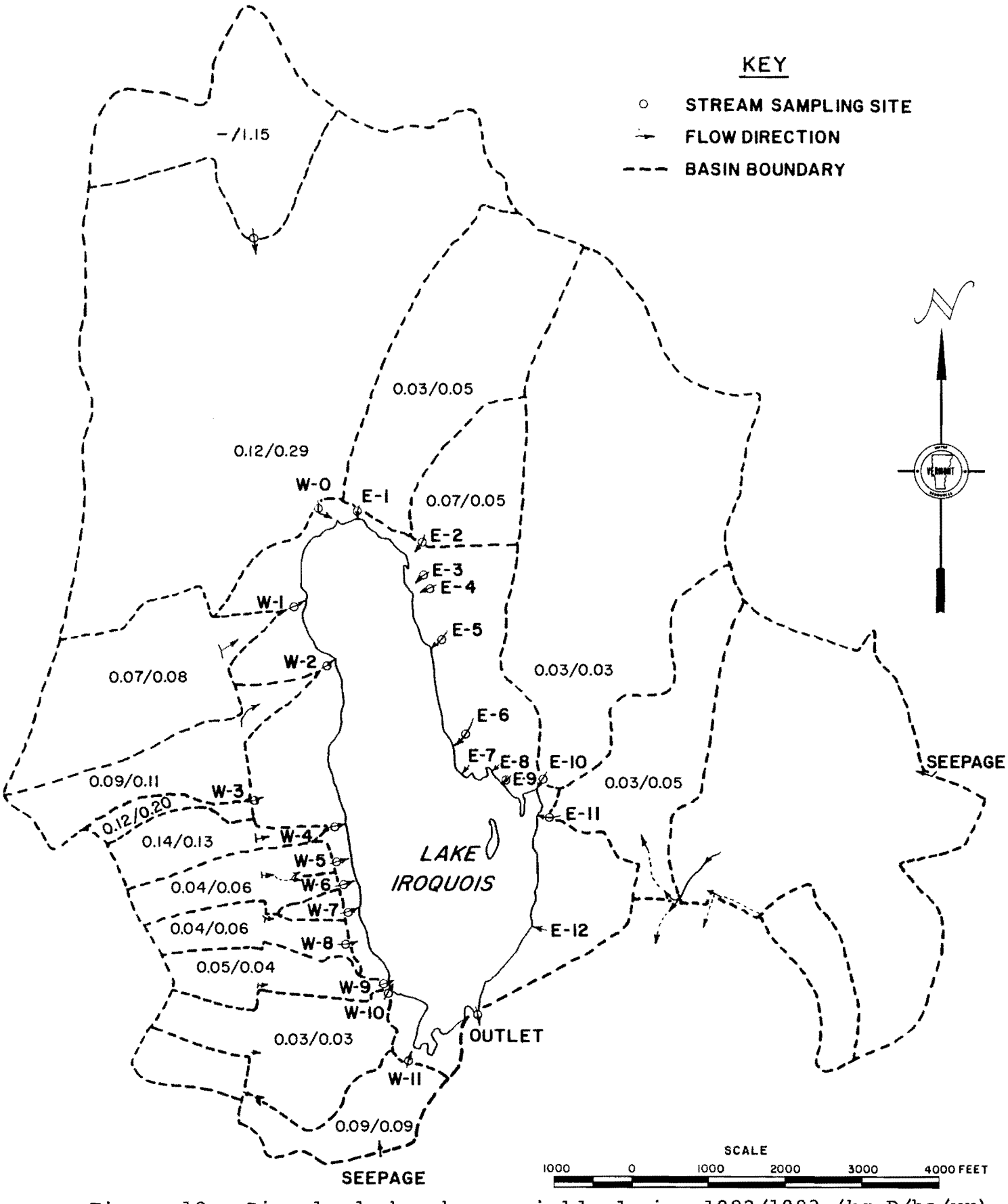


Figure 18. Dissolved phosphorus yields during 1982/1983 (kg P/ha/yr).

Areal total phosphorus yields ranged from 0.06 to 2.43 kg/ha/yr. Lowest yields were associated with sub-basins which were predominantly wooded with minor areas of development (W-10, W-11, E-1, E-10, E-11). Areal total phosphorus yields from these sub-basins averaged 0.13 kg/ha/yr during the two years measured. Yields from the remaining west shore sub-basins (W-0 through W-9) were higher and averaged 0.53 kg/ha/yr. These sub-basins have undergone a much greater degree of agricultural and residential development. A discussion of sub-basin characteristics which contribute to increased phosphorus yields is presented in a later section of this report.

Surface Outflow

Surface outflow accounts for the great majority of the lake's water loss, 87-92% during the two years of measurement. Evaporation losses contribute the remainder. Groundwater outflow appeared to be insignificant (see following section). However, a small portion of the surface outflow was provided by seepage through lake bed material in the immediate vicinity of the outlet dam. This seepage passed through and around the outlet dam and largely became a component of the measured surface flow immediately below the dam.

Phosphorus losses through outflow were 59 and 137 kg during 1982 and 1983, respectively. These values were only 25-34% of the total phosphorus inputs which occurred during the two years. Thus, annual phosphorus retention by the lake ranged from 66-75%. The measured phosphorus retention agreed well with predicted values derived from empirical models (Table 20), indicating that the lake behaves normally in this respect. However, an empirical phosphorus retention model derived from a group of "anoxic" lakes (Reckhow, 1979) predicts much lower retention. Lakes having dissolved oxygen concentrations of 0 mg/l occurring in their hypolimnion were defined as anoxic. Lake Iroquois conforms to this definition, and thus, for an anoxic lake, exhibits above normal phosphorus retention.

Table 20. Phosphorus retention model predictions for Lake Iroquois*.

Reference	Predictive Equation	R_p Estimates 1982	1983
Kirchner and Dillon (1975)	$1-R_p = Q_s/(Q_s+13.2)$	0.75	0.62
Ostrofsky (1978)	$1-R_p = 1-.201e^{-.0425Q_s}-.574e^{-.00949Q_s}$	0.72	0.67
Reckhow (1979)			
"General"	$1-R_p = Q_s/(11.6+1.2Q_s)$	0.74	0.62
"Anoxic"	$1-R_p = Q_s/(.17Z+1.13Q_s)$	0.26	0.20
	Measured Retentions	0.75	0.66

	1982	1983
$*Q_s$ = areal water loading (m/yr) =	4.425	8.071
Z = mean depth (m) =	5.9	5.9

Outflow phosphorus concentrations were compared to lake surface water phosphorus concentrations samples from 0 and 1 m depths at sampling site number one (Figure 9) to determine whether lake samples alone would have been adequate for estimating phosphorus losses through outflow. Flow-weighted mean phosphorus concentrations based on actual outlet samples and lake samples were computed for dates of common samples during 1982 and 1983 (Appendix I). For 1982, the flow weighted mean outlet phosphorus concentration was 13-14% lower than the mean based on lake phosphorus concentrations at 0 and 1 meters. This difference, however, was not statistically significant at the 95% confidence level. For 1983, outlet sample means were again lower than means based on lake samples. The differences were smaller, 2% and 13% for the 0 and 1 meter samples, respectively, and again were not statistically significant. Although a significant difference is not apparent within the sample sets examined, outlet phosphorus concentration estimates based on lake samples were consistently higher than those based on actual outlet samples. Use of lake samples to estimate outlet phosphorus could have increased lake loss estimates by as much as 14%.

Groundwater

Flow Patterns

Groundwater flow patterns were examined through the use of seepage meters, wells, and piezometers placed around the lake's perimeter. Seepage meter measurements showed that groundwater inflow occurred around the entire shoreline of the lake. Measurements from individual meters are listed in Table 21 (see Figure 12 for meter locations). Evaluation of the reliability of individual measurements required some subjectivity, since meters were sensitive to a variety of operational and site specific conditions which could result in erroneous readings. For example, seepage meter 1 provided consistently negative readings indicating groundwater outflow in an area of shoreline dominated

Table 21. 1982 seepage rates (um/sec).

Seepage Meter	DATE 6/10	6/17	6/25	6/28	7/1	7/6	7/9	7/15	7/29	7/30	8/2	8/6	8/9
1	--	-0.028	-0.002[1,2]	--	-0.036	--	--	-0.043	--	--	--	--	--
2	--	--	--	--	--	--	--	0.035	0.028	0.027	-0.003[2]	--	--
3	--	--	--	--	--	--	--	0.004[2]	0.002[2]	-0.013	0.003[2]	--	--
4	--	0.594	1.251	--	--	--	--	0.416	--	--	--	--	--
5	--	-0.009[3]	0.404	--	--	--	--	0.144	0.133	0.181	--	--	--
6	--	0.346	0.639[1]	--	--	--	--	--	0.330	0.319	0.279	--	0.286
7	--	0.028	0.045	--	0.039	--	--	0.023	0.063	0.034	0.043	--	0.215
8	--	0.643	0.243	--	≥0.791	--	--	0.144	≥1.640[4]	≥1.565[4]	1.674[4]	≥1.815[4]	--
9	--	--	--	--	--	--	--	0.019	0.705	-0.013	0.095	0.214	--
10	--	--	-0.115[3]	--	--	--	--	--	--	0.347	0.381	--	≥1.383
11	--	--	--	--	--	--	--	--	-0.009	0.006[2]	0.040	0.000[2]	--
12	--	--	--	--	--	--	--	--	0.000	-0.137	0.035	0.056	--
13	--	--	--	--	--	--	--	--	-0.283	-0.305	-0.163	--	-0.192
14	--	--	--	--	--	--	--	--	0.330	0.532	0.005	--	-0.002[2]
15	--	--	--	--	--	--	--	0.179	-0.013	0.065	0.133	--	0.026
16	--	-0.194	-0.158	--	-0.338	--	--	--	0.180	-0.410	-0.428	--	-0.382
17	--	--	--	--	--	--	--	--	-0.059[3]	0.000[2]	-0.003[2]	--	-0.153
18	--	--	--	--	--	--	--	--	-0.106	-0.228	-0.225	--	-0.175
19	--	--	--	--	--	--	--	--	-0.063	-0.030	-0.037	--	-0.039
20	--	--	--	--	--	--	--	--	0.102	-0.035	-0.073	--	0.111
21													
22	--	--	--	--	--	--	--	-0.175	0.908	--	-0.005	--	-0.156
23	--	--	--	--	--	--	--	0.082[1]	1.318	0.023	0.087	--	0.916[1]
24	--	-0.030	-0.009	--	0.027	--	--	-0.022[1]	0.180	0.010	0.000[2]	--	0.002[2]
25	--	0.022	-0.046	--	-0.022	--	--	--	--	--	--	--	--
26	0.051	0.041	0.139	--	0.094	--	--	0.044	0.220	0.082	-0.029	--	0.026[1]

Table 21 (continued).

Seepage Meter	DATE 6/10	6/17	6/25	6/28	7/1	7/6	7/9	7/15	7/29	7/30	8/2	8/6	8/9
27	≥1.186[1]	≥3.61	8.38	--	7.468	--	--	--	--	--	--	--	--
28	--	6.32	9.31	0.157	0.798	1.176	0.916	--	--	--	--	--	--
						2.379							
29	--	--	--	0.021	0.075	0.214	0.112	--	--	0.326	--	--	--
30	--	--	--	0.154	0.282	0.399	0.107	0.110	0.083	--	0.091	--	0.037
31	--	--	--	0.010	0.012	-0.004[2]	--	--	--	--	--	--	--
32	--	--	--	-0.002	0.007	-0.023	--	--	--	--	--	--	--
33	--	--	--	0.419	0.483	0.482	--	--	--	--	--	--	--
34	--	--	--	≥0.978[1]	≥1.297	4.028[1]	--	--	--		--	--	--
					4.112								
35	--	--	--	0.005	0.005	0.000[2]	--	--	--	--	--	--	--
36	0.486[1]	0.526[1]	0.696	0.500	0.538	0.647	--	0.325	0.313	0.362	0.229	--	--
37	--	--	--	0.225	0.246	--	--	--	--	--	--	--	--
38	--	--	--	0.026	0.024	0.020	--	--	--	--	--	--	--
39	-0.016	-0.014	0.006	--	0.003	--	--	--	--	--	--	--	--
40	0.003[1,2]	0.002[2]	0.033	--	0.005	--	--	--	--	--	--	--	--
41	0.003[1,2]	0.004[2]	0.043	--	0.001	--	--	--	--	--	--	--	--
42	0.003[1,2]	0.004[2]	0.009	--	0.007	--	--	--	--	--	--	--	--
43	0.025	0.012	-0.001[1,2]	--	0.006	--	--	--	--	--	--	--	--
44	0.010[1]	0.030	0.010	--	0.009	--	--	--	--	--	--	--	--

[1] leak in bag; results suspect.
[2] less than 15 ml volume change; could be accounted for by sampling procedure alone.
[3] seepage meter recently emplaced; may not have been fully stabilized.
[4] meter 8 was relocated several feet from original position, due to dropping lake levels.

by groundwater inflow, as evidenced by well and piezometer data. The cause of this discrepancy was not identified. Seepage meter 10 was placed in a gravel substrate adjacent to a shoreline area with no significant source of groundwater. This meter showed relatively high positive readings which were probably due to lake water movement or pressure effects transmitted through the gravel rather than actual groundwater movement. Footnotes in Table 21 identify other individual measurements which may be questionable. Most seepage meters, however, were believed to provide reliable seepage measurements.

Low seepage rates (0.001 - 0.040 um/sec, Table 21) were found at the lake's north end (meters 39-44, Figure 12). Less permeable bottom materials and a less sloping water table could account for this. Water uptake by phreatophytes in this area might also be a factor. Moderate seepage rates (0.020 - 2.00 um/sec) were found at most east and west shore sites, with the highest rates (up to 9.00 um/sec) at sites near stream mouths (meters 4, 8, 27, Figure 12).

Three sets of seepage meters located along the northeast shore (meters 28-38, Figure 12) were installed to examine the width of the shoreline seepage zone and seepage variation with distance from shore. Measurements made at these meters are illustrated in Figure 19. The three sets of meters were all located within a 50 m section of shoreline. Seepage was negligible beyond 10 m from shore, where fine sediment began to blanket the bottom. Seepage rates showed a general decline with increasing distance from the shore. Mathematical groundwater models (McBride and Pfannkuch, 1975) and various lake studies (Fellows and Brezonik 1980; Brock et al., 1982; Lee, 1977) have shown that this is commonly the case. However, the great deal of variation exhibited at these sets of meters (as well as at other sites) made it apparent that accurate seepage rate vs. distance from shore relationships for various shoreline segments could not be readily established.

Negative seepage measurements, which indicate groundwater outflow, did occur at some seepage meter sites along the

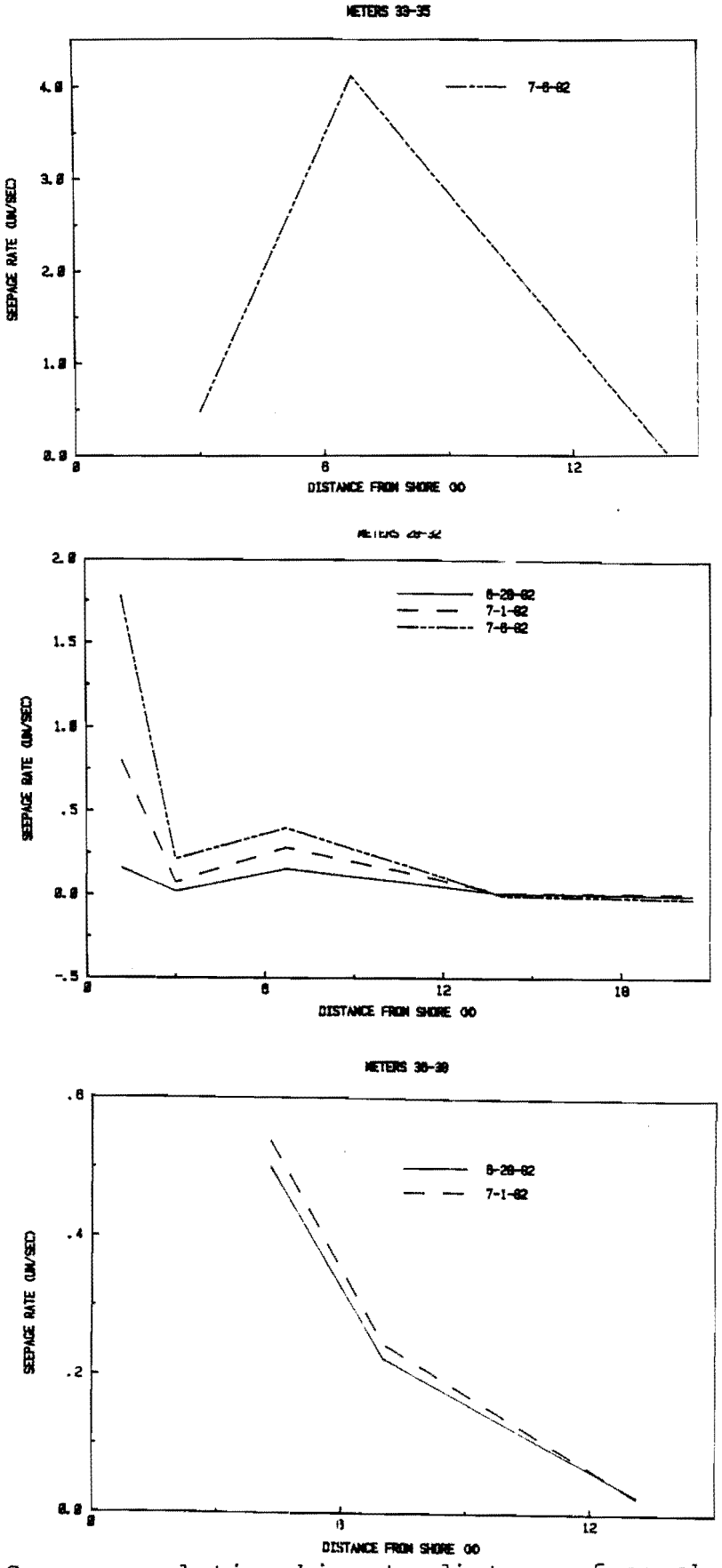


Figure 19. Seepage relationships to distance from shore.

southeast shoreline and at the dam area. Seepage meters 22 and 24, located along the southeast shoreline, fluctuated between positive and negative measurements, while meter 23, located between these two, consistently showed positive measurements (Figure 12 and Table 21). Some additional seepage meter measurements were made in this area during September, 1983 in an attempt to determine if the negative measurements were meaningful. Measurements from five meters located near the former site of meter 22 are given in Table 22. A mix of positive and negative values was again obtained. One meter (meter 1) gave consistently negative readings. Two others (meters 2 and 3) gave both positive and negative readings. Two meters (meters 4 and 5) which had been left resting loose on the bottom surface to demonstrate the response of a poorly sealed meter showed both positive and negative values with magnitudes similar to the sealed meters.

Groundwater flow patterns in this area appeared to be complex. The potential for groundwater outflow exists due to the drop in land elevation to the south of the lake. Any outflow would pass through bedrock fractures in the ridge separating the lake from the lower area to the south. Due to relatively low net negative seepage and the small area involved, the magnitude of this outflow has been assumed to be negligible in the overall hydrologic budget for the lake.

Negative seepage was more obvious in the small bay area at the lake's outlet. Seepage meters in this area (meters 13-20, Figure 12) provided relatively consistent negative measurements of moderate magnitude. However, most of this negative seepage apparently collected as leakage at various points in the dam face, and so was measured as a component of surface outflow. Several flow measurements of the outflow stream approximately 250 m downstream from the dam during low flows (less than 1 cfs) showed only slight increases over measurements made immediately below the dam. It was therefore assumed that there was very little groundwater outflow around the dam that was not measured as surface outflow.

Well sample analyses provided additional evidence of the dominance of groundwater inflow around the lake's perimeter. The chloride concentration of the lake water averaged 9.7 mg/l during 1982. A shoreline well in an area dominated by groundwater outflow would have a similar concentration. An examination of well chloride concentrations (Appendix D) showed a wide range of chloride concentrations that were clearly influenced by inflow of either uncontaminated or contaminated groundwater. Wells which did have chloride concentrations similar to lake water generally had distinguishing concentrations of other parameters which verified that groundwater outflow was not occurring at the site. Wells located along the southeast shoreline (wells B-13-16, Figure 10), where seepage meters indicated that some groundwater outflow might have occurred, had chloride concentrations which indicated groundwater inflow dominated at these sites.

Piezometer measurements proved to be of limited value since insufficient head existed at most sites for adequate readings. Also, preliminary plans to use these measurements in conjunction with seepage meter measurements to estimate groundwater inflow were found to be impractical due to limitations in the operation of both shallow piezometers and seepage meters. However, piezometer measurements (Appendix F) did support the general groundwater findings based on seepage meters and well samples.

Table 22. 1983 southeast shore seepage rates (um/sec).

	Seepage Meter Number				
Date	1	2	3	4[1]	5[1]
9-14-83	-.039	+.027	+.013	+.056	+.057
9-15-83	-.030	+.015	+.020	-.005	+.028
9-16-83	-.029	-.013	-.005	-.016	+.025

[1]meters 4 and 5 were unsealed.

Annual Inflow

Annual groundwater inflow during 1982 was estimated to be 81 x 10^4 m^3, based on the mean measured seepage rate applied to a 10 meter wide perimeter band. The mean measured seepage rate used was 0.532 um/sec, which had a large standard deviation of 1.468. A second groundwater inflow estimate was made by dividing the seepage zone into six sections and applying mean seepage rates from each section. This produced a value of 66 x 10^4 m^3.

For 1983, groundwater inflow was estimated as the residual term in the annual water budget. A value of 70 x 10^4 m^3 was obtained. A groundwater estimate produced as a water budget residual will be influenced by the cumulative errors associated with the measured terms, and so may contain a considerable degree of error. However, the good agreement between the estimates produced from seepage meter measurements and from the water budget lends support to the adequacy of either method. The groundwater inflow value of 70 x 10^4 m^3 was used for hydrologic and phosphorus budget calculations for both 1982 and 1983. Since groundwater contributed only a small fraction of the total external phosphorus loading (see below), any errors associated with this groundwater inflow term were relatively inconsequential.

Phosphorus Concentration

Groundwater phosphorus concentrations are commonly low since both particulate and dissolved forms of phosphorus are effectively retained by soils under most conditions. Mean phosphorus concentrations in eight shoreline wells not influenced by septic tank effluent or wetland areas (wells A-1,2,3,6,8, B-32 and C-1,2, Figure 10) ranged from 4 to 12 ug/l, with an overall mean of 8 ug/l. Phosphorus concentrations in wells downgradient from some wetland areas were considerably higher. Means from six wells influenced by wetland areas (wells A-5,9 and B-12,17,30,31) ranged from 22 to 89 ug/l, with an overall mean of 50 ug/l.

Reducing conditions which develop in wetland soils can result in the conversion of ferric iron to the soluble ferrous form and the associated release of iron-bound phosphorus. The wetland areas upgradient from these wells have been created or expanded in recent years. Thus, the present elevated groundwater phosphorus concentrations may be due to inundation of previously aerated soils and may not represent long-term equilibrium conditions.

Phosphorus Loading

The annual groundwater phosphorus loading to the lake was estimated to be 8 kg. This value was only 2-3% of the total external phosphorus loading. Additional phosphorus loading carried by groundwater but derived from septic systems was estimated to be about 3 kg. This estimate was based on results of a well sampling program discussed in a later section of this report.

Precipitation and Evaporation

Precipitation totaled 29.16 inches during 1982 and 49.10 inches during 1983. Annual precipitation averages 32.75 inches at the Burlington weather station, approximately seven miles northeast of the lake. The 1983 precipitation total for the Lake Iroquois watershed exceeded all totals for the 40 previous years of record at the Burlington weather station. The monthly distribution of precipitation for the watershed during 1982 and 1983 and for long-term means at Burlington are shown in Figure 20. During 1983, nine out of twelve months had precipitation above normal with the months of April, May, November, and December being most notable. These months, which have a relatively high potential for runoff, received nearly double or more of their normal precipitation. Water inputs directly to the lake surface via precipitation amounted to 73×10^4 m^3 in 1982, and 123×10^4 m^3 in 1983, accounting for 17% and 15% of the total inputs during 1982 and 1983, respectively.

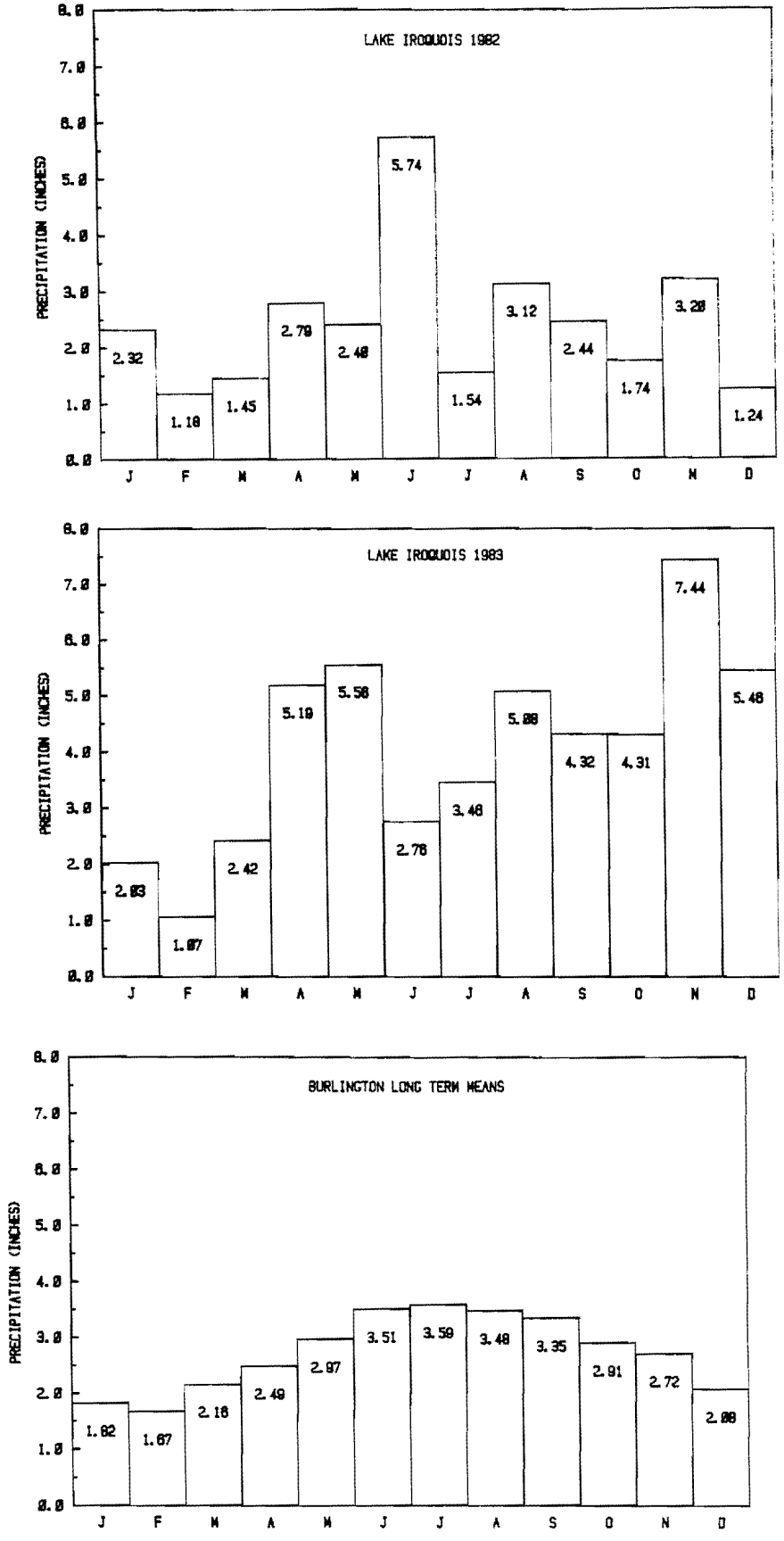


Figure 20. Monthly precipitation distribution.

Phosphorus loading to the lake from atmospheric inputs was estimated to be 30 kg/yr based on literature values. This phosphorus was delivered in both dissolved and particulate forms in wet precipitation and dry fallout. The loading from these sources provided 8-13% of the total external phosphorus loading during the two years.

Water losses through evaporation were estimated to be 62 x 10^4 m^3 and 63 x 10^4 m^3 during 1982 and 1983, respectively. This was the equivalent of approximately 25 inches of water lost from the lake's surface each year. Evaporative losses accounted for 8-13% of the total water losses during the two years.

Lake Storage

Fluctuations in lake level during 1982 and 1983 are shown in Figure 21. Individual lake level measurements are listed in Appendix A. Levels varied over ranges of 2.91 feet during 1982, 2.21 feet during 1983, and 3.57 feet during the two year period. Lake levels were generally lower in 1982 due to less water input and greater releases through the outlet dam. The lake experienced a net volume loss of 47 x 10^4 m^3 during 1982, and a net volume gain of 42 x 10^4 m^3 during 1983. Fluctuations in lake phosphorus content are presented in a later report section.

Summarized Hydrologic and Phosphorus Budgets

Hydrologic budgets were developed for the 1982 and 1983 calendar years. Table 23 lists the budget term estimates made for the two years.

Surface inflow and outflow dominated the budget. Surface inflow provided 67% and 76% of total water input, and surface outflow provided 87% and 92% of total water output during 1982 and 1983, respectively. Groundwater outflow was assumed to be negligible.

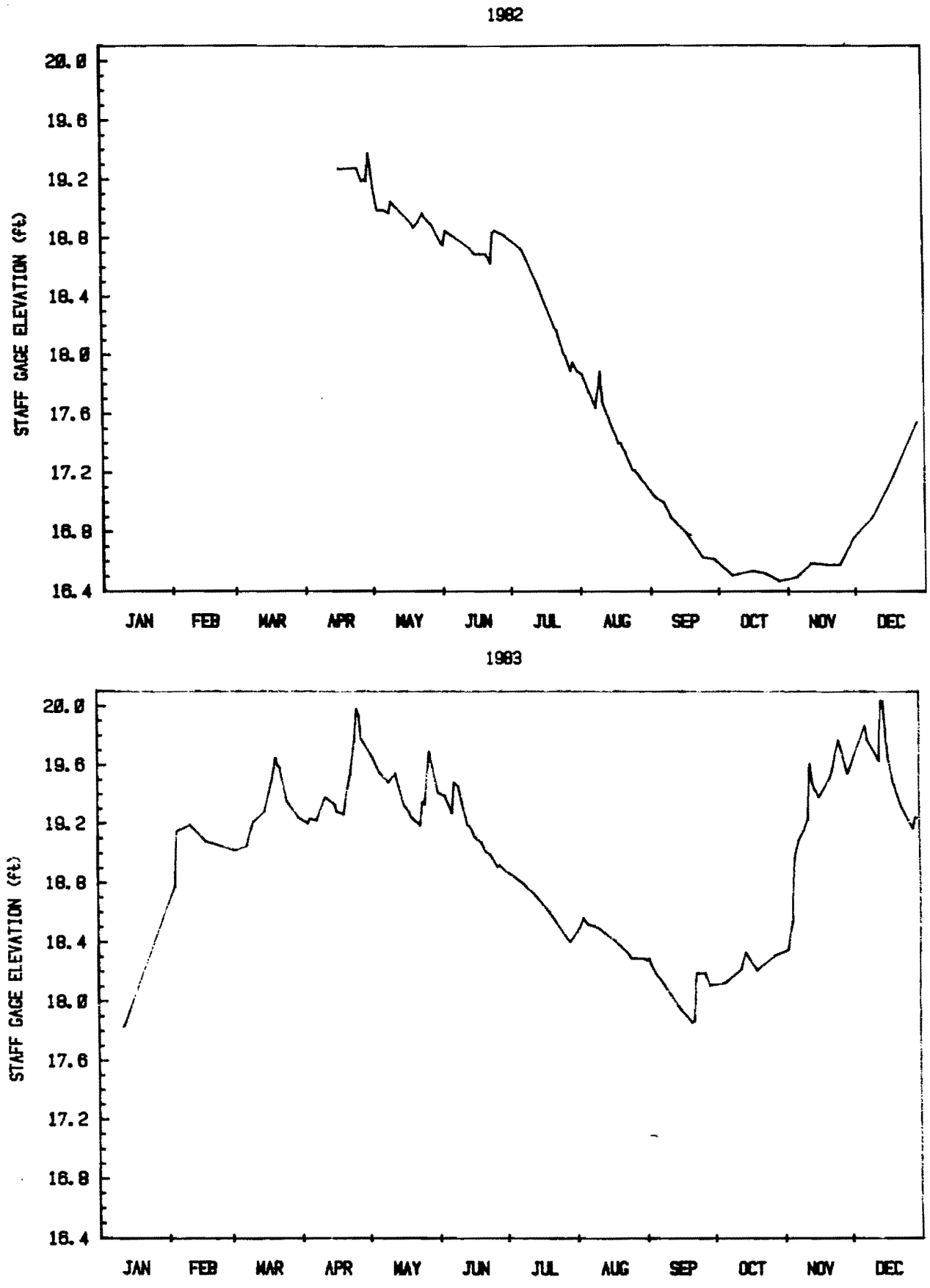


Figure 21. Lake level fluctuations.

Table 24 lists the phosphorus budget terms for the two years. Surface inflow was the dominant source of phosphorus input, accounting for 83% and 90% of the total during 1982 and 1983, respectively. Figure 22 depicts the distribution of total phosphorus inputs and outputs for both years. The phosphorus distribution pattern generally reflected the pattern of surface inflow (Figure 16), with peak inputs occurring during the peak flow periods of spring and fall.

Table 23. Annual hydrologic budget terms.

		Water Volume ($10^4 m^3$)	
		1982	1983
Inputs			
Surface Inflow		294	604
Groundwater Inflow		70	70
Direct Precipitation		73	123
	TOTAL	437	797
Outputs			
Surface Outflow		422	692
Evaporation		62	63
	TOTAL	484	755
Change in Lake Storage		-47	+42

Table 24. Annual phosphorus budget terms.

		Total Phosphorus (kg)	
		1982	1983
Inputs			
Surface Inflow		198	358
Groundwater Inflow		8	8
Atmospheric Deposition		30	30
Septic Systems		3	3
	TOTAL	239	399
Outputs			
Surface Outflow		59	137

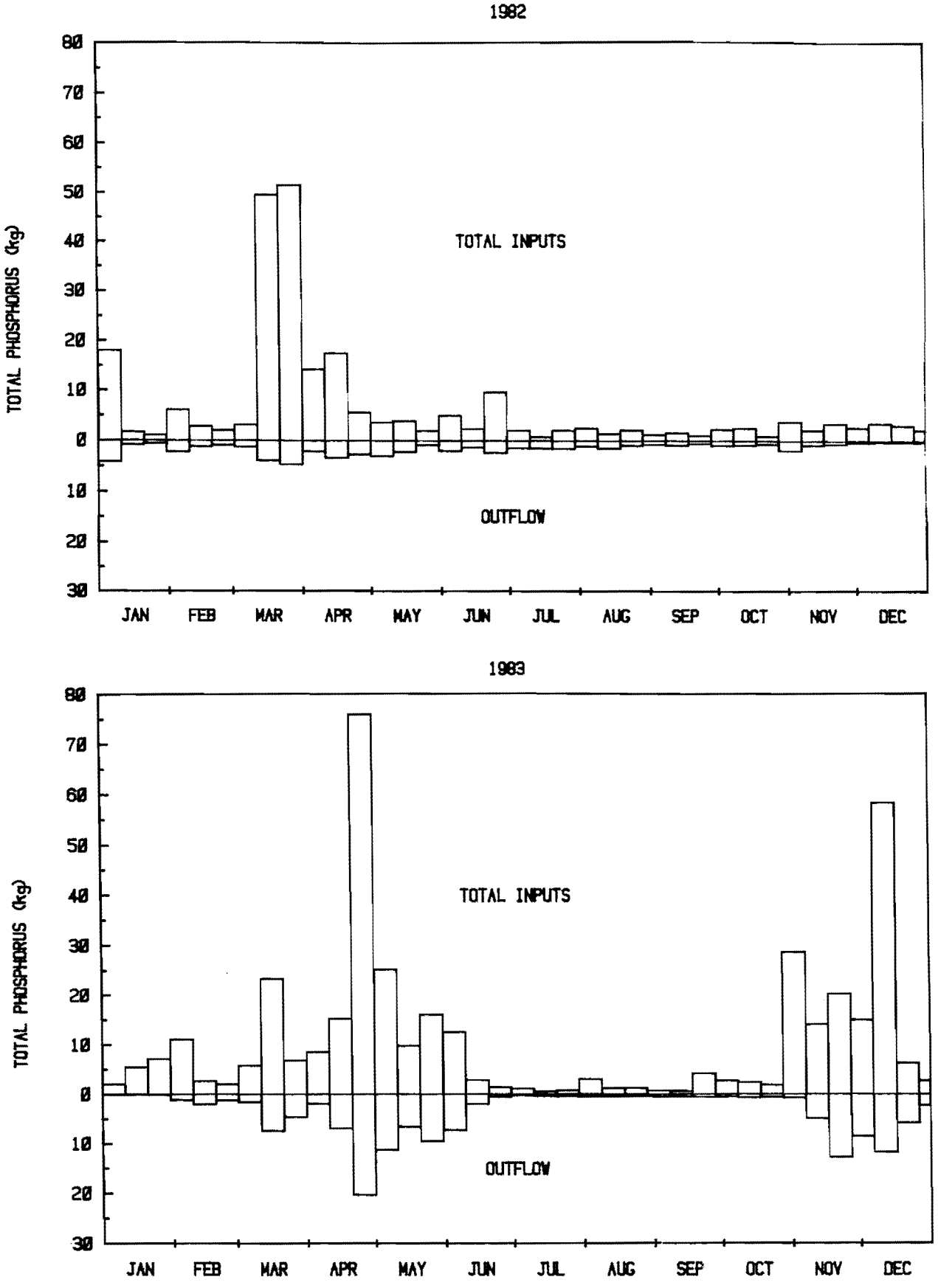


Figure 22. Total phosphorus inptus and outputs (10-day intervals).

FACTORS AFFECTING EXTERNAL PHOSPHORUS LOADING

Phosphorus Transport in the Environment

Phosphorus is widely present in the environment and is an essential nutrient for both plant and animal life. Phosphorus is a component of nearly all forms of organic matter, both living and non-living. It is also present in soils. Phosphorus (as phosphate) is readily adsorbed to surfaces of soil particles. Organically bound phosphorus and phosphorus bearing minerals are also present in soils. Organic matter and soils are major phosphorus pools and their rate of delivery to a lake has a major influence on the lake's phosphorus supply. A substantial, although minor, portion of the lake's phosphorus supply is delivered in an unbound form, largely as ionic phosphate. The supply and delivery of this phosphorus is often controlled by factors closely related to those controlling particulate phosphorus.

Phosphorus is the nutrient controlling productivity in Lake Iroquois. The lake's phosphorus supply is largely controlled by the rate at which phosphorus is lost from the watershed and delivered to the lake. The rate at which phosphorus is lost from a land area is greatly influenced by the manner in which the area is developed and utilized.

An important factor influencing phosphorus loss is the quantity and intensity of surface runoff. Surface runoff can readily transport both particulate and dissolved forms of phosphorus. In contrast, water which infiltrates into the ground before draining transports very little phosphorus. Land development generally increases runoff and reduces infiltration. A second related influential factor is the efficiency with which runoff-transported phosphorus is delivered to the lake. This is determined by the pathway runoff follows on its way to the lake and the existing opportunities for phosphorus retention through sedimentation or biological uptake.

Areas of undisturbed forest produce runoff which is extremely low in phosphorus. In fact, a forested area commonly yields less phosphorus in its drainage than it receives from bulk precipitation. The surface layer of organic material on the forest floor is extremely important in this regard. This layer protects the soil from raindrop impact. It has the capacity to absorb and retain water, slow its movement, and filter the water which passes through. This organic layer, along with the improved underlying soil structure, promotes infiltration and minimizes surface runoff. The transport of dissolved phosphorus is reduced through efficient biological uptake and soil adsorption. The forest floor is commonly pitted by treefall which has occurred over the centuries. These depressions increase the water storage capacity of the area and further promote infiltration. Existing stream channels are highly stable.

As agricultural and residential development occurs, the hydrologic and phosphorus retention properties of an area are altered. The protective vegetation and organic surface layer are removed or greatly changed. Soil surfaces are exposed temporarily and in some cases permanently. Pitted and irregular surfaces are smoothed. Wet areas, which once provided phosphorus depositional zones, are drained. Surface runoff and its unimpeded delivery to the lake is greatly promoted. Ditches are constructed to accelerate surface drainage. Channelization of existing water courses occurs. Straight channels allow higher streamflow velocities and have higher sediment delivery efficiencies. As the areal density of drainageways increases, the average travel distance between any point in a basin and the nearest drainageway is reduced, which also increases the efficiency of delivery. Constructed ditches and channels are frequently steep-banked and highly erodable. Increased surface runoff results in increased storm flow peaks which destabilize and erode natural stream channels.

Erosion increases associated with development are also closely tied to increased phosphorus losses since soil contains phosphorus. Increased loss of organically bound phosphorus also

commonly coincides with increased soil erosion. For a given area, erosion rates increase as vegetation and surface organic matter is reduced. Exposed soil areas are subject to extremely high erosion rates. Erosion rates from disturbed soil sites are typically several hundred times greater than from wooded sites. Land slope also strongly influences erosion. Development of a steeply sloping site results in a much greater increase in the quantity of soil eroded than similar development on a gently sloping site. Channel erosion in unstable drainageways can also provide large quantities of soil phosphorus to a lake.

Other sources of an increased phosphorus supply to a lake which are not directly related to land surface and drainage modification include animal wastes, sewage, and fertilizer. Animal wastes appear to be by far the most significant of these for Lake Iroquois. The manure produced by a single dairy cow has the phosphorus equivalent of the sewage produced by 16-50 people (Porcella et al., 1974; Millar and Turk, 1955; Ellis and Childs, 1973). Thus, 200-250 cows (the approximate number present in the Lake Iroquois watershed) produce the waste phosphorus equivalent of a human population of 3,200-12,500. The manner in which this manure is handled can have a great influence on a lake's phosphorus supply. Winter spreading of manure in the Lake Iroquois watershed was practiced prior to 1981. A Vermont study (Midgely and Dunklee, 1945) found that 5-10% of the phosphorus in winter spread manure was lost with spring runoff. This practice may have contributed substantial quantities of phosphorus to the lake in the past. The installation of manure pits which allow winter manure storage has probably greatly reduced this problem. Uncontrolled drainage from sites of manure accumulation (barnyards, manure piles, manure pits) can also supply very significant quantities of phosphorus.

Residential sewage in the Lake Iroquois watershed is handled by on-site disposal. Shoreline septic systems were found to achieve high levels of phosphorus retention. Two partially failing systems were observed in the watershed, although no specific attempt to examine systems not on the shoreline was

made. Sewage appears to make a relatively small contribution to the lake's phosphorus supply, although it does provide yet another addition above background conditions.

Application of phosphorus fertilizer to cropland and lawn areas also has the potential for contributing phosphorus to the lake, although this source is also probably small relative to others. Phosphorus fertilizer applied to cropland is rapidly bound by the soil, and losses which occur result largely from erosion of the phosphorus enriched surface soil (Taylor, 1967). On the average, less than 5% of the phosphorus fertilizer applied to lawn areas is lost through runoff (Windham Regional Planning Agency, 1982). Higher losses can result if heavy rains occur shortly after fertilizer application. Fertilization of Lake Iroquois shoreline lawn areas was found to be quite limited.

Although almost any development of a lake watershed will result in some increase in the lake's phosphorus supply, it should not be concluded that the problem is uncontrollable. The magnitude of the impact of land development can be substantially reduced by applying appropriate practices to existing problems and by guiding future development to minimize its effects.

Sub-basin Characteristics Influencing Phosphorus Yields

Areal total phosphorus yields from the sub-basins of the Lake Iroquois watershed varied widely, from 0.06 to 2.43 kg P/ha/yr (Table 25 and Figure 17). The sub-basins are ranked on the basis of their total phosphorus yields during 1982 and 1983 in Table 25. The five sub-basins with the lowest phosphorus yields (E-1,10,11, and W-10,11) all have 65-100% of their areas wooded (see Table 17). Of the nine remaining sub-basins, only two (E-2 and W-5/6) have more than 65% of their areas wooded. Generally, the sub-basins with a greater degree of residential and agricultural development had greater phosphorus yields. However, very localized problem sites within a sub-basin and specific characteristics of certain areas sometimes had a major

Table 25. Ranked sub-basin total phosphorus yields (kg P/ha/yr).

1982		1983	
Sub-basin	TP Yield	Sub-basin	TP Yield
E-2	0.56	W-1	2.43
W-1	0.54	W-4	0.72
W-5/6	0.51	W-0	0.49
W-2	0.45	W-2	0.43
W-4	0.43	W-5/6	0.42
W-8	0.40	W-3	0.40
W-9	0.35	E-2	0.31
W-3	0.24	W-8	0.27
W-0	0.21	W-9	0.24
E-10	0.15	E-11	0.18
E-11	0.14	W-11	0.17
W-11	0.14	E-10	0.12
W-10	0.13	E-1	0.08
E-1	0.10	W-10	0.06

influence on the entire sub-basin's phosphorus yield. Notable features of sub-basins which influenced their phosphorus yields are presented in the following paragraphs.

Sub-basin W-0

This is the largest sub-basin (244 ha), comprising 29% of the watershed's land area. The sub-basin contains the watershed's two largest dairy farms and its largest wetland area. The sub-basins's areal total phosphorus yield was fairly low in 1982 (0.21 kg P/ha/yr), but considerably higher in 1983 (0.49 kg P/ha/yr). Due mostly to its size, it contributed more phosphorus to the lake than any other sub-basin (Table 18 and 19).

During 1983, stream sampling was conducted at a site below the largest dairy farm (sub-basin section W-O-A, Figure 3) to evaluate the contribution of dairy farms to the total sub-basin phosphorus yield. The areal total phosphorus yield from this section of the sub-basin, which comprises 17.6% of the sub-basin area (5.1% of the total watershed area), was 2.19 kg P/ha/yr. Fifty-nine percent of the total phosphorus load from this section was in the dissolved form. Since this flow passes through a wetland area before becoming a component of total sub-basin flow, its contribution cannot be clearly quantified. However, assuming 100% of the dissolved phosphorus passes through the wetland, this section provided 70% of the dissolved phosphorus load from the sub-basin and 46% of the dissolved phosphorus streamload from the entire watershed during 1983. The assumption of 100% dissolved phosphorus passage through the wetland was not verified. However, most loading passed during the high flows of early spring and late fall at a time of relatively low biological activity. Additionally, any dissolved phosphorus retention occurring within the wetland may be balanced by conversion of particulate phosphorus to the dissolved form. Considerable retention of particulate phosphorus by the wetland is likely.

Additional sampling was conducted during the spring of 1984 to more precisely identify the major sources of the high phosphorus yield found from this section of the sub-basin. Sampling sites and results of analyses are shown in Figure 23 and Tables 26 and 27. Barnyard area drainage provided the majority of the phosphorus present in these streamflows on the dates sampled. Phosphorus concentration increases between sites 1 and 2 (Figure 23) appeared to be largely due to manure pit leakage and paved barnyard drainage. Erosion from exposed soil areas probably also made smaller contributions. Gradual release of channel edge manure deposits appeared significant on some dates. Phosphorus concentration increases between sites 5 and 4 were due to manure pit leakage. This leakage declined substantially on later sampling dates. The high total phosphorus concentration at site 5 on 4/19/84 was probably due to upstream channel disturbance by pastured cows. Phosphorus concentration increases between sites 6 and 7 on 4/13/84 and 4/19/84 may have been due to a poorly operating residential septic system located just upstream from site 7. Release of phosphorus from channel storage might also have been possible. The milkhouse waste system, located just below site 6, did not appear to be making significant contributions to the stream at the times of sampling. The very high phosphorus concentrations at site 8 were due largely to the existing manure pit drainage system. This system was installed to control groundwater problems encountered on the site during pit installation. Unfortunately this system is also effectively collecting manure pit seepage and conducting it into the adjacent road ditch. Drainage from the remaining barnyard area (site 11) was also high in phosphorus. Flows in the road ditch at site 8 were far below those at site 7, while phosphorus concentrations at site 8 were far higher. This resulted in a phosphorus loading at site 8 which ranged from 21-99% of the site 7 loading (Table 27). The phosphorus derived from barnyard area drainage at these areas is clearly a large source of the entire basin's phosphorus yield.

Table 26. 1984 barnyard stream phosphorus sampling results (ug/l).

TP = Total Phosphorus
DP = Dissolved Phosphorus

Date and Analysis	SITE* 1	2	3	4	5	6	7	8	9	10	11
3-28-84 TP	41	182	130	353	31	177	118	2900			
3-28-84 DP	17	66	63	215	14	79	67	2590			
4-05-84 TP	43	190	96	143	20	144	145	3460			
4-05-84 DP	18	114	44	75	13	102	81	3170			
4-13-84 TP	24	61	35	20	16	48	131	15500			
4-13-84 DP	14	37	16	11	10	31	92	12500			
4-16-84 TP	135	652	64	120	50	410	352	748	202	9360	2500
4-16-84 DP	31	322	32	57	20	220	208	604			
4-19-84 TP	29	231	94	55	70	125	196	4980			
4-19-84 DP	22	183	27	26	19	94	161	4780			

*Site locations are shown in Figure 23.

Table 27. Miscellaneous barnyard stream sampling data.

Sample Date	Site 7 FLow (cfs)	Site 8 Flow (cfs)	Phosphorus loading at site 8 as % of Site 7 loading	% of Phosphorus Concentration Appearing Between: Site 1 & Site 2		Site 5 & Site 3	
				TP	DP	TP	DP
3-28-84	0.43	0.009	51	77	74	76	78
4-05-84	0.70	0.015	51	77	84	79	70
4-13-84	0.24	0.002	99	61	62	54	38
4-16-84	1.61	0.156	21	79	90	22	38
4-19-84	0.51	0.010	50	87	88	26	30

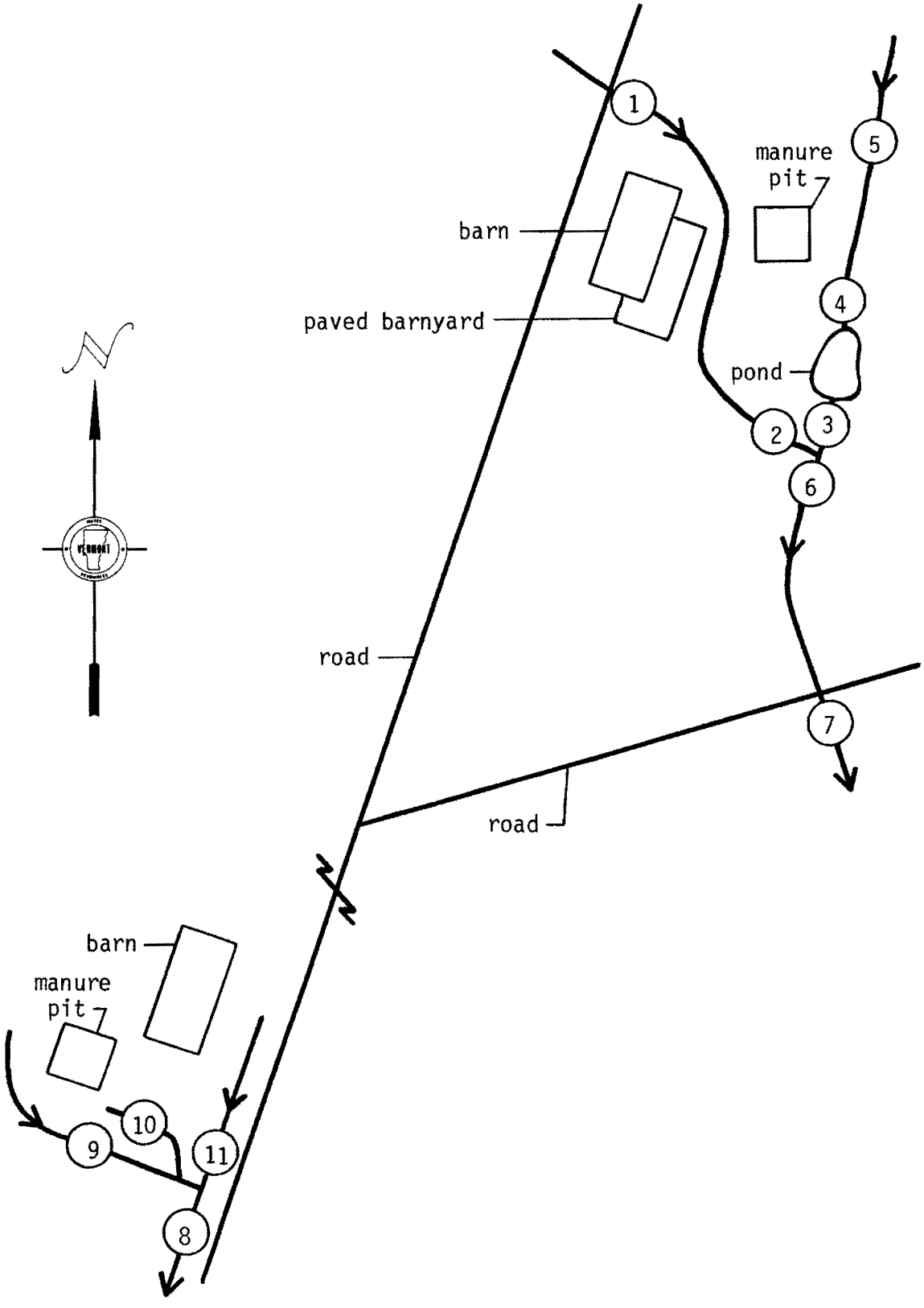


Figure 23. 1984 barnyard stream phosphorus sampling sites (not to scale).

The wetland in the W-0 sub-basin was created as a result of beaver activity about 10 years ago. It was not possible to accurately evaluate the phosphorus retention capability of the wetland during this study due to the extensive additional sampling effort which would have been required. Most wetland areas examined elsewhere have been found to provide net retention of phosphorus. In a review of 16 wetland studies by Van der Valk et al. (1979), most were found to accumulate phosphorus on an annual basis. Five additional wetland studies cited in another review (New Hampshire Water Supply and Pollution Control Commission, 1983) all showed that net phosphorus retention occurred. It is likely that the wetland in the W-0 sub-basin also provided net phosphorus retention, especially since a series of beaver ponds was present. Van der Valk et al. (1979) suggest the construction of ponds within wetlands to improve phosphorus removal efficiencies by enhancing sedimentation.

The total phosphorus loading from the sub-basin section containing the large dairy farm (W-O-A) was 79% of the total phosphorus loading from the entire sub-basin in 1983. If no wetland phosphorus retention was occurring, the remaining 201 hectares of the sub-basin would have had an areal phosphorus yield of only 0.12 kg P/ha/yr. This is unreasonably low, especially when the relative phosphorus contributions from the second farm and from residential development activities in the sub-basin are considered. Thus, the loadings measured also indicate that the existing wetland achieves a substantial degree of phosphorus retention.

Sub-basin W-1

Events which occurred in this sub-basin during the course of the study provide an excellent example of the impact that can result from a very small area of land clearing and stream channel disturbance. In the early fall of 1982, a small area of grassland (0.4 ha) bordering the road was bulldozed. The adjacent road ditch was also graded and a segment of drainageway

perpendicular to the road was reshaped. The total phosphorus load for the sub-basin during 1982 was 23.6 kg (0.54 kg/ha/yr). Application of the flow-weighted mean phosphorus concentration measured before the disturbance indicated the loading would have been only 13.8 kg (0.32 kg/ha/yr) if the disturbance had not occurred. Below average flows during the fall of 1982 probably helped minimize the impact that year.

In 1983 the total phosphorus load for this sub-basin was 111.3 kg (2.43 kg/ha/yr). Massive erosion occurred at the disturbed site during the spring high flow period. Soil was washed from both the bulldozed area and the disturbed channels. Gravel and rock washed from the site scoured and eroded the channel immediately downstream from the site which further increased the quantity of soil carried to the lake. On one day of sample collection, a band of turbid water which extended half the length of the lake was observed originating at the mouth of this stream.

Other west shore sub-basins (W-2 - W-9) did not show significant increases in phosphorus yields in 1983. Assuming the 1982 undisturbed loading estimate for sub-basin W-1 (13.8 kg P) would have been applicable to 1983, this small disturbed area resulted in the contribution of an additional 98 kg of phosphorus to the lake, 27% of the total phosphorus streamload that year. The land disturbance involved only 1% of the sub-basin area and resulted in a 759% increase in the sub-basin's total phosphorus yield.

Sub-basins W-2, 3, 4, 5/6, 8, 9

Residential land use comprises a large percentage of the areas of most of these six sub-basins (see Table 12). Areal total phosphorus yields were moderately high (0.24 - 0.72 kg/ha/yr). Ditching and drainage modifications were fairly extensive. Numerous exposed soil sites were present, including unpaved road surfaces, unstable drainageway banks, and various small construction and land clearing projects.

Sub-basins W-3 and W-4 were found to have relatively high yields of dissolved phosphorus (see Tables 18 and 19). Failing septic systems and/or animal wastes are possible sources. The property examination required to verify this was not conducted.

Total phosphorus yields for sub-basin W-4 were considerably higher in 1983 than in 1982. This probably reflects the fact that samples taken early in 1982 (during peak flows) were collected at a site upstream from the lowest segment of the drainageway. This segment was channelized in 1981 and was probably eroding more rapidly in the spring of 1982. Samples for the remainder of 1982 and all of 1983 were collected below this channelized segment. Total phosphorus loading measured during 1982 would probably have been higher than that measured in 1983 if the phosphorus contributions resulting from erosion of this channel segment during 1982 peak spring flows had been represented.

NOTE: The sub-basin divide between W-3 and W-4 as shown in Figures 3, 17, and 18 is probably not accurate. Sub-basin flow measurement analysis indicated that 2.4 hectares of the W-3 sub-basin area shown is actually part of the W-4 sub-basin. All flow, loading and areal estimates listed elsewhere in this report (with the exception of land use percentages) have been adjusted accordingly.

Sub-basin W-10

This sub-basin had the second lowest total phosphorus yield of any sub-basin during 1982 (0.13 kg/ha/yr) and the lowest during 1983 (0.06 kg/ha/yr). It is 90% wooded. Other characteristics contributing to low phosphorus yields include more moderate slopes, small areas of internal drainage, and a stable stream channel. Lower sections of the stream channel are wide and very well vegetated, with small wetland pool areas which probably act as effective sediment traps.

Sub-basin W-11

This sub-basin also had low total phosphorus yields (0.14 kg/ha/yr in 1982, 0.17 kg/ha/yr in 1983). It is 76% wooded. A wetland area in the lower section of the sub-basin maintains a pool of standing water which promotes sedimentation of particulate phosphorus. The sub-basin does have a relatively high dissolved phosphorus yield (0.09 kg/ha/yr), considering the low magnitude of its total phosphorus yield. This is probably a result of processing of phosphorus within the wetland. High concentrations of soluble organic phosphorus have been found in wetland drainage elsewhere (Lee et al., 1975).

Sub-basin E-1

This 96% wooded sub-basin had very low total phosphorus yields (0.10 kg/ha/yr in 1982, 0.08 kg/ha/yr in 1983).

Sub-basin E-2

This 100% wooded sub-basin had unusually high total phosphorus yields during both 1982 and 1983 (0.56 and 0.31 kg/ha/yr). Most of the area appears to have been agricultural land at one time. The ground cover is very poorly developed with areas of nearly bare soil present. This probably accounted for the high phosphorus yields. High concentrations of organic-rich particulates (processed leaf litter and soil aggregates) were observed during high flow periods.

Sub-basin E-10

This nearly 100% wooded sub-basin had low total phosphorus yields (0.15 kg/ha/yr in 1982, 0.12 kg/ha/yr in 1983). The steep slopes of this sub-basin and the lack of any significant depositional areas along the mainstream channels probably account for the phosphorus yields being somewhat above the lowest phosphorus yields measured in the watershed.

Sub-basin E-11

Total phosphorus yields from this sub-basin were relatively low (0.14 kg/ha/yr in 1982, 0.18 kg/ha/yr in 1983) considering the amount of existing developed land. Two sections of the sub-basin (E-11-A and E-11-B) are delineated in Figure 3. Portions of the surface flow derived from these sections is conducted out of the watershed at two points of divergent streamflow. Based on an areal flow yield comparison with sub-basin E-10, it was estimated that only 59% of the combined surface flow produced by these sections was delivered to the lake. (Areal phosphorus yields have been adjusted accordingly.) This partially limits the impact of the area of development on phosphorus loading by reducing the percentage of water it contributes. Thus, about 70% of the sub-basin's surface flow delivered to the lake was derived from wooded land.

Another factor which may contribute to this sub-basin's relatively low phosphorus yield is the presence of three small wetland areas along the major drainageways. These areas become submerged during high flows and probably provide for sedimentation of particulate forms of phosphorus.

INTERNAL PHOSPHORUS CYCLING

Sediment Phosphorus Release

In a phosphorus limited lake, the rate at which phosphorus is delivered through streamflow, groundwater inflow, and atmospheric deposition controls the level of productivity which will occur. A second major factor controlling productivity is the efficiency with which external phosphorus inputs are recycled within a lake. Sedimentation of particulate bound phosphorus results in continual losses of phosphorus from the water column. To maintain significant phosphorus concentrations in the water column, these losses must be countered by continued external supplies or sediment derived phosphorus releases.

A major means of returning sedimented phosphorus to the water is through the release of iron-bound phosphorus from hypolimnetic sediments. Non-calcareous sediments have been shown to release negligible quantities of phosphorus under aerobic conditions (Holdren and Armstrong, 1980; Sonzogni *et al.*, 1977; Larsen *et al.*, 1980; Mortimer, 1941.) However, much greater release rates commonly occur under anoxic conditions. This is largely due to the reduction and dissolution of hydrous ferric oxide precipitates. Under aerobic conditions, these precipitates can effectively adsorb large quantities of phosphorus. The development of anoxic conditions (low redox potential) causes these precipitates to dissolve, which releases both ferrous iron and phosphorus into the overlying water.

The depth and morphometry of Lake Iroquois make it prone to the development of an anoxic hypolimnion. The lake exhibits stable thermal stratification during the summer period which restricts the transport of oxygen to the hypolimnion. The hypolimnion is relatively thin, with a mean depth of 1.5 m, and so has a relatively low initial supply of oxygen at the onset of stratification. Thus, the lake is sensitive to productivity increases which result in a greater supply of oxygen-demanding organic matter being delivered to the hypolimnion.

The depth at which the hypolimnion was formed and maintained was similar in 1981 and 1982, but was substantially different in 1983 (Figure 24). In that year, a consistently thicker hypolimnion was observed. This was largely due to the unusually rapid warming of the surface water during June. Below normal wind speeds during June of that year also contributed to a reduced transfer of heat inputs to deeper water. Surface temperatures climbed approximately 10°C during the first two weeks of the month and reached a temperature of 25°C by mid-June (Figure 25). Similar surface temperatures were not reached until mid-July in 1981 and 1982.

The depletion of oxygen in the hypolimnion during 1981-1983 is shown in Figure 26 and Table 28 (a constant hypolimnion depth of 8.5 m was applied). Initial losses were partly due to temperature increases which reduced oxygen solubility. Depletion was rapid during May, and hypolimnetic oxygen content approached zero during June of each year. Calculated depletion rates were dependent on initial and final dates selected. May 10-12th were used as initial dates since depletion was rapid afterwards and a similar oxygen mass was present on these dates during all three years. The first date on which dissolved oxygen concentrations fell below 1 mg/l was used as the end point (6-19-81, 6-28-82, 6-15-83). Volumetric hypolimnetic oxygen depletion rates ranged from 0.18 mg/l/day in 1982 to 0.27 mg/l/day in 1983.

Table 28. 1981-1983 Hypolimnetic oxygen depletion rates.

Year	Areal H.O.D. ($g/m^2/day$)	Volumetric H.O.D. ($g/m^3/day$)	Dates of Measurement
1981	0.35	0.24	5/12 - 6/19
1982	0.26	0.18	5/10 - 6/28
1983	0.40	0.27	5/12 - 6/15

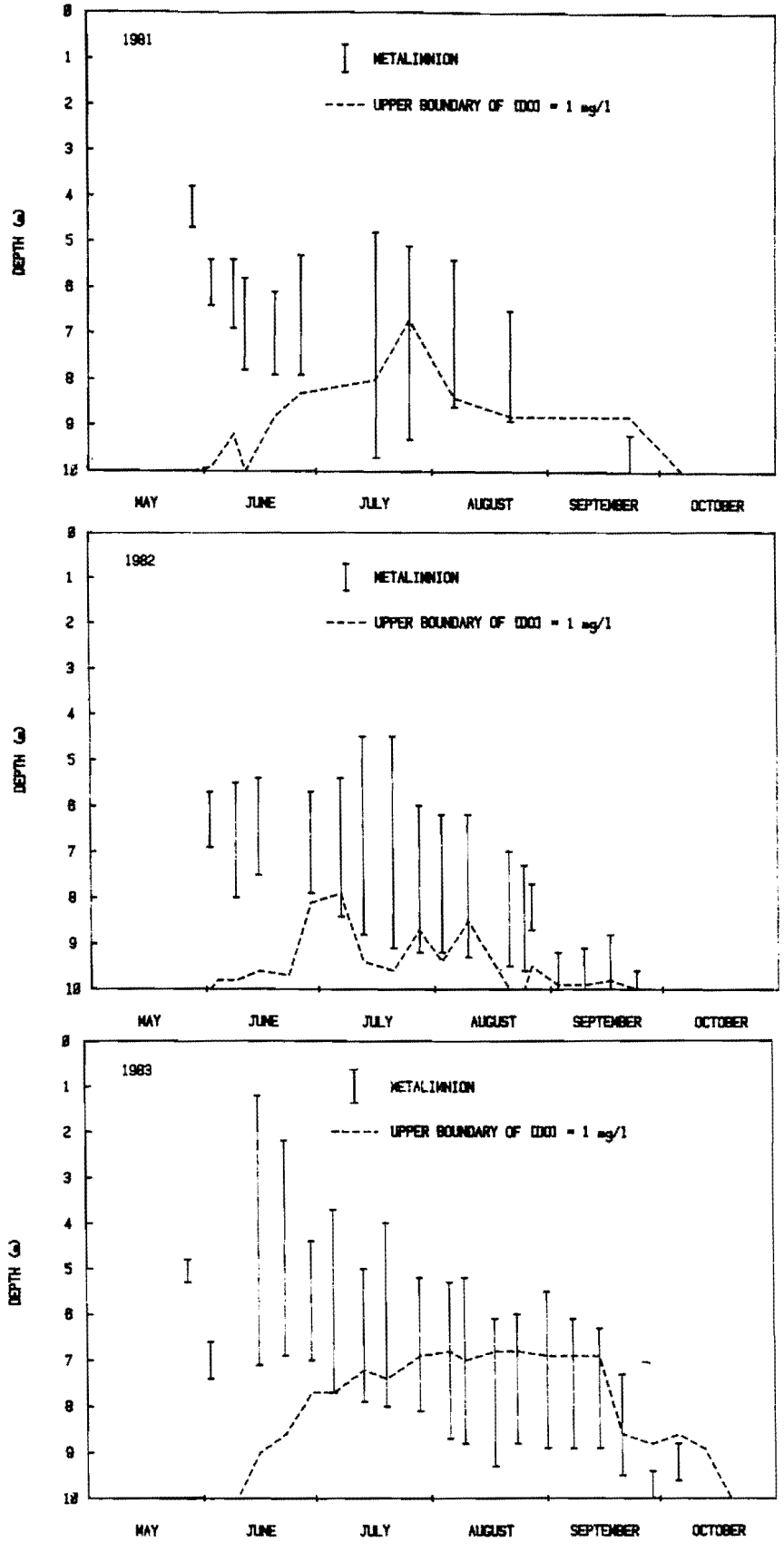


Figure 24. Metalimnetic boundaries and extent of critical dissolved oxygen concentrations (less than 1.0 mg/l).

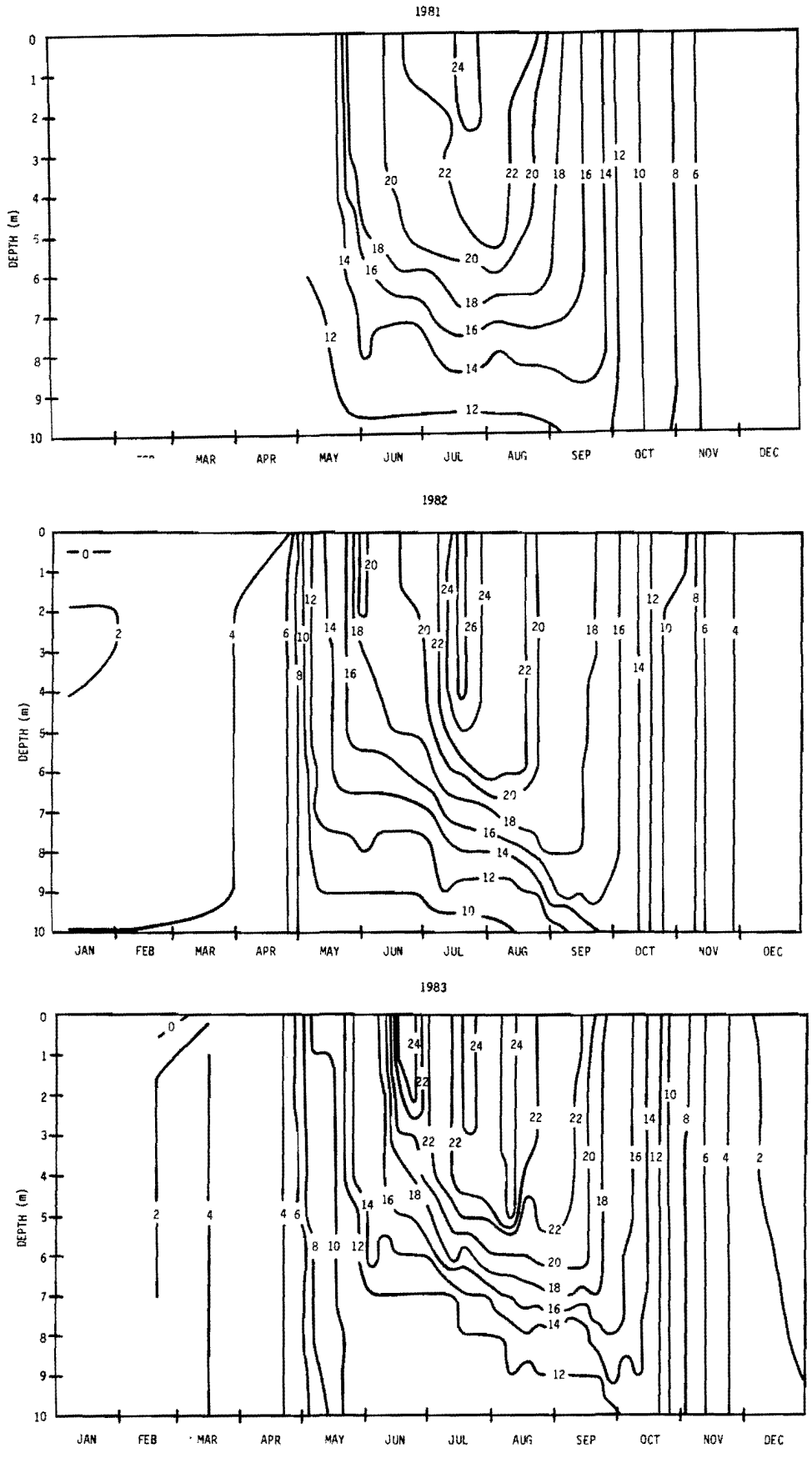


Figure 25. Depth-time isopleths for temperature (°C).

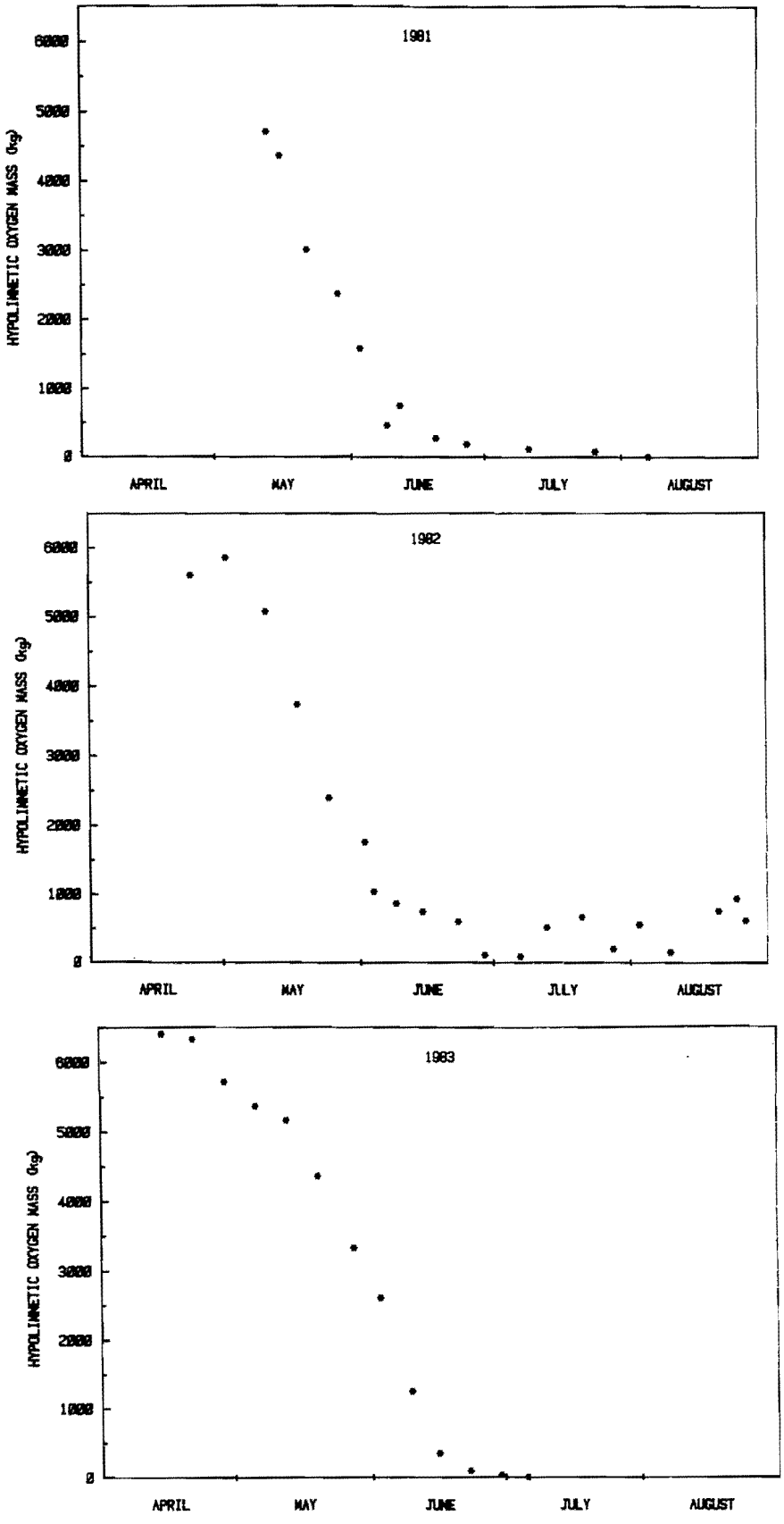


Figure 26. Hypolimnetic oxygen depletion.

The severity of depletion after June also varied during the three years. Hypolimnetic oxygen concentrations were highest during 1982 when a concentration of 0 mg/l was found on only one sampling date at a depth of 10 m during early August (Figure 27). In 1981, dissolved oxygen concentrations at the 10 m depth remained at 0 mg/l from mid-July through early September. Concentrations of 0 mg/l were also found at 9 m during August. Hypolimnetic anoxia was most extensive during 1983 when the 10 m depth remained anoxic from late June to mid-October. Concentrations of 0 mg/l were found as shallow as 7 m during August. Differences in the degree of hypolimnetic oxygen depletion during the three years are also depicted by the 1 mg/l dissolved oxygen boundary lines shown in Figure 24.

As hypolimnetic oxygen concentrations approached zero, sediment phosphorus release was initiated, elevating the hypolimnetic phosphorus concentrations. Phosphorus release preceded the development of complete anoxia at the 10 m depth since lower oxygen concentrations probably existed at the sediment surface. Sedimentation of epilimnetic phosphorus would also have contributed to elevated hypolimnetic phosphorus concentrations, although this was probably a relatively minor factor in comparison to sediment phosphorus release.

Phosphorus concentrations in the hypolimnion showed similar peak values in 1981 and 1982 (126 and 128 ug/l, respectively), although in 1981 near peak values (75 ug/l) occurred over a much longer time period (Figure 28). In 1983, relatively massive releases of sediment phosphorus occurred. Hypolimnetic phosphorus concentrations as high as 884 ug/l were found. This resulted in a net gain of 325 kg of phosphorus by the water column between June and September (Figure 29). This was nearly a four-fold increase in the total lake phosphorus mass.

Hypolimnetic iron concentrations showed increases which roughly paralleled those of phosphorus. A peak concentration of 8370 ug/l was found during October of 1983 (Table 29). Hypolimnetic manganese concentrations also showed similar increases (Table 30).

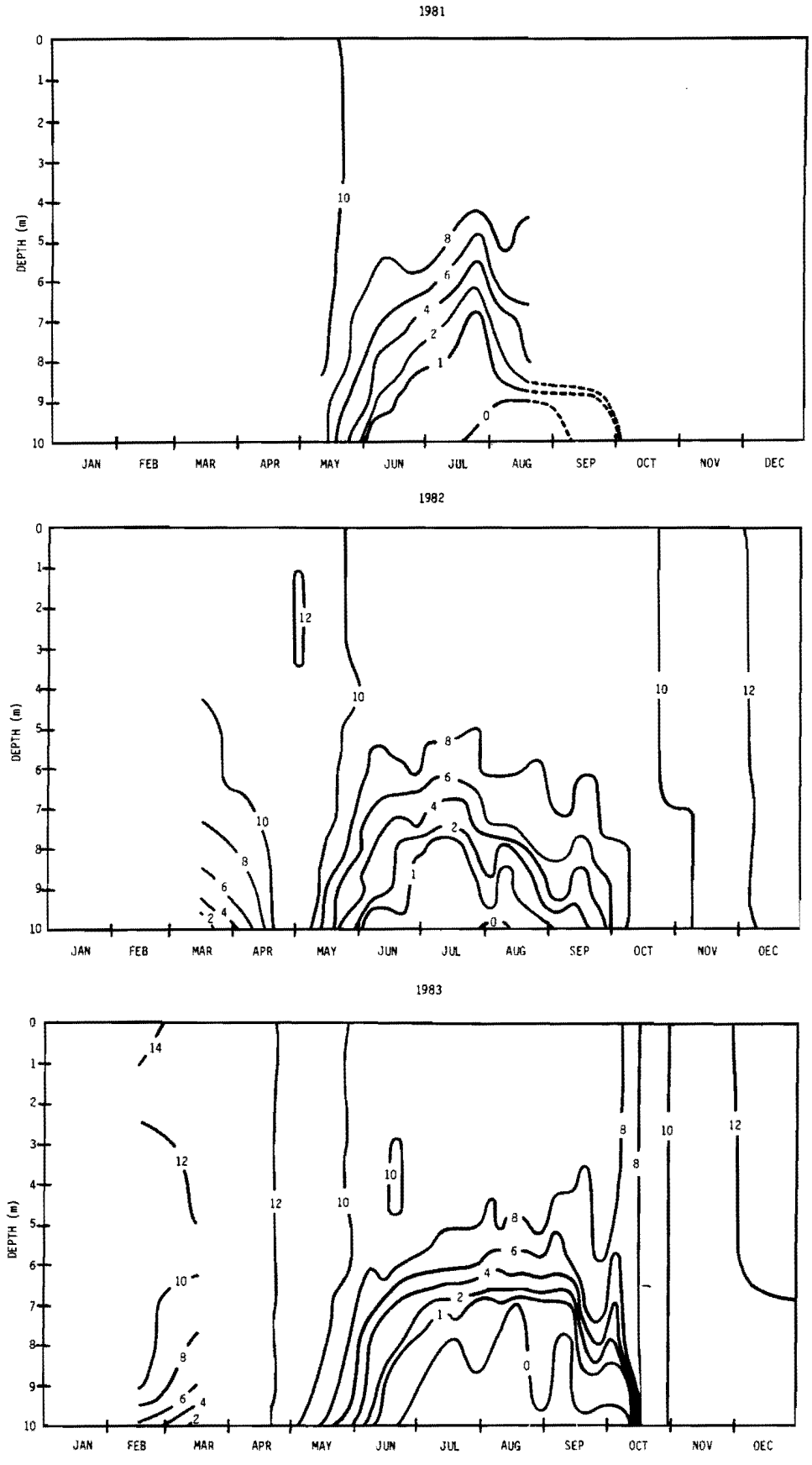


Figure 27. Depth-time isopleths for dissolved oxygen (mg/l).

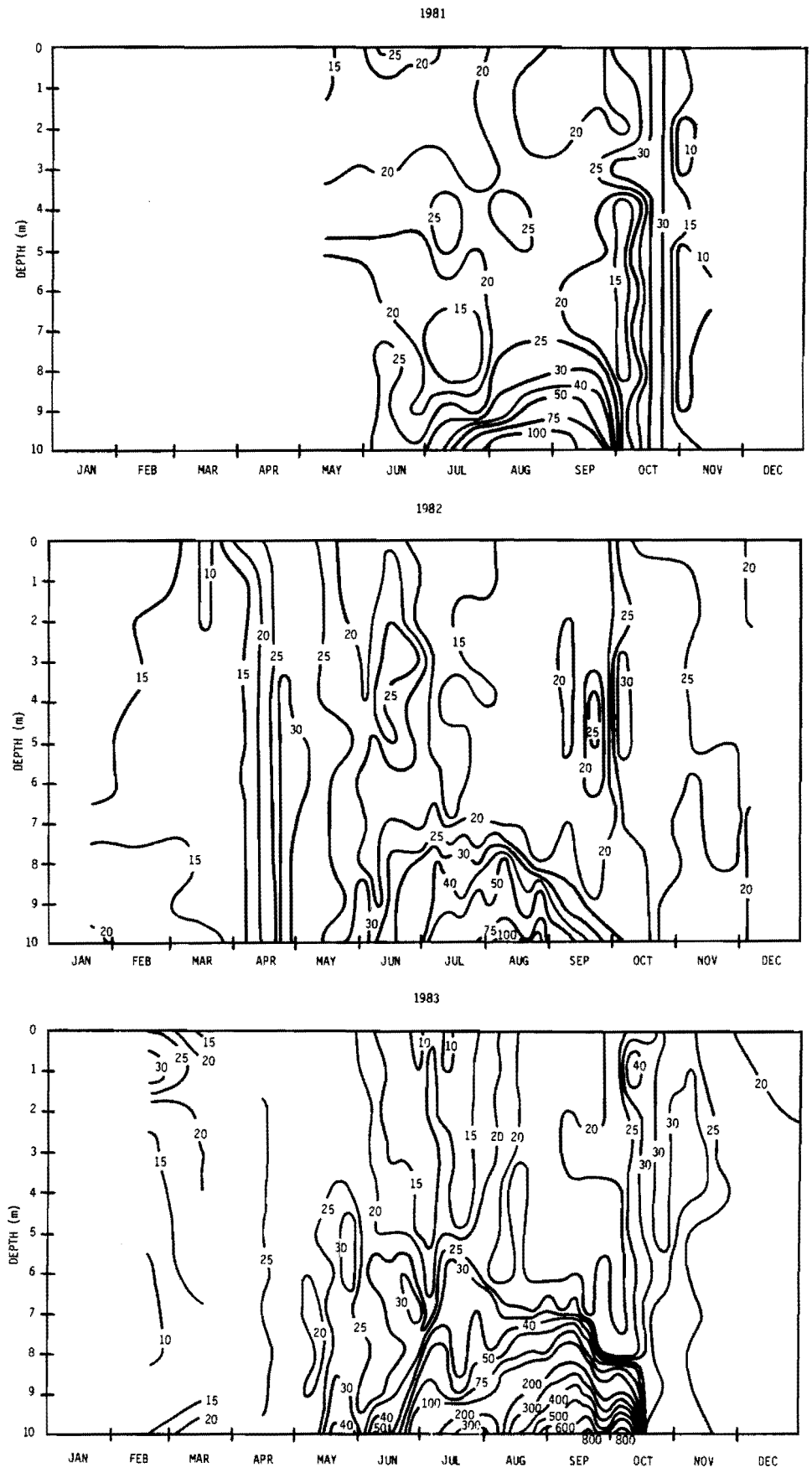


Figure 28. Depth-time isopleths for total phosphorus concentrations (ug/l).

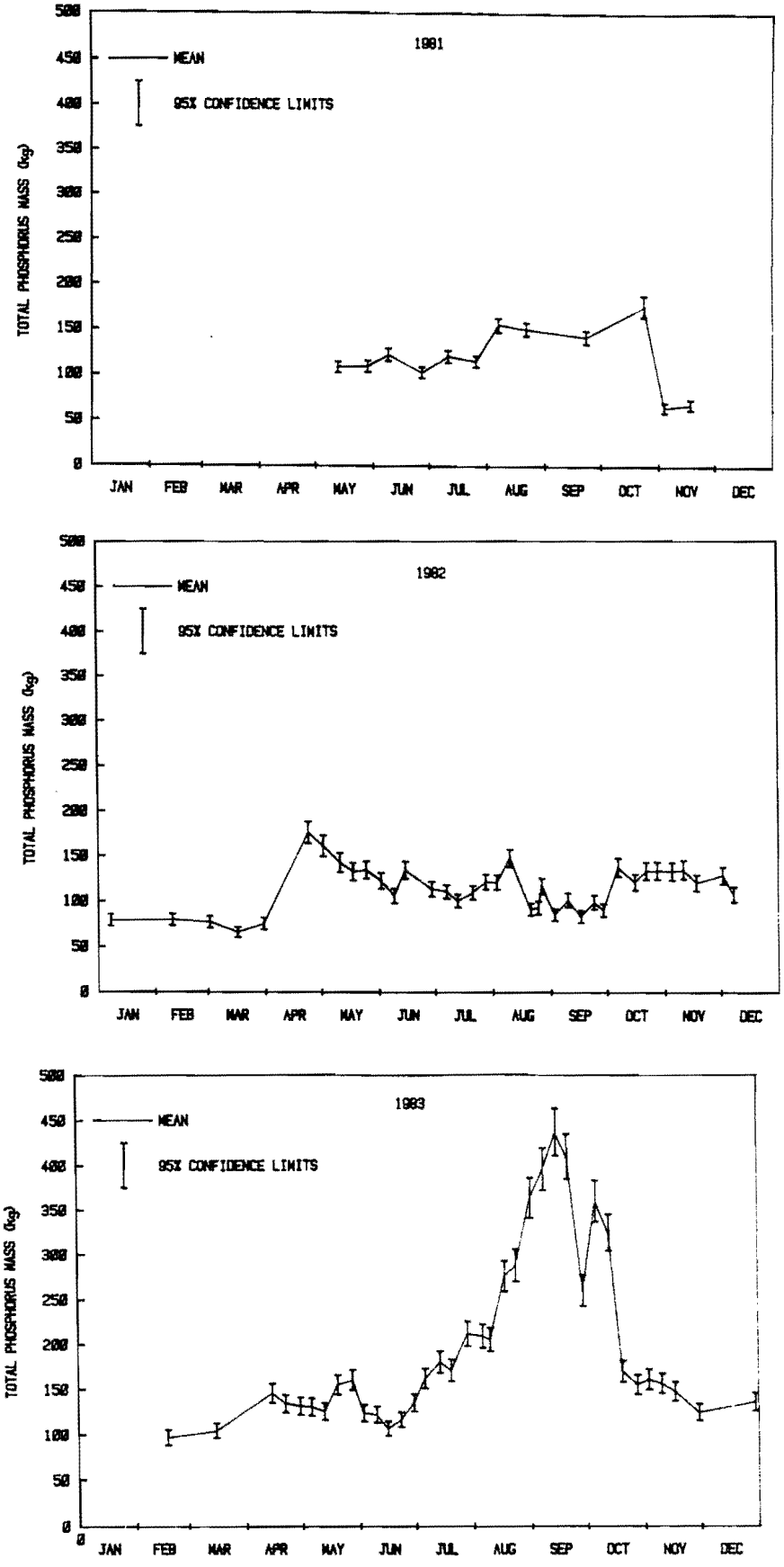


Figure 29. Fluctuations in lake phosphorus mass.

Table 29. 1983 lake iron concentrations (ug/l). (Dissolved iron values are in parentheses.)

Depth (m)	Date									
	May 19	Jun 15	Jul 19	Aug 17	Sep 20	Oct 5	Oct 12	Oct 19	Oct 27	Nov 9
1										
2	92 (14)	262 (21)	30 (17)	46 (8)	78 (20)	68 (7)				
3							135 (23)	139 (11)	135 (7)	105 (15)
4	103 (27)	289 (27)	32 (18)	105 (8)	74 (14)	68 (7)				
5										
6	129 (27)	125 (21)	321 (5)	13 (9)	61 (14)	53 (7)	137 (11)	133 (8)	142 (4)	109 (12)
7	98 (9)	121 (21)	186 (13)	110 (25)	71 (27)					
8	161 (27)	121 (18)	220 (18)	784 (650)	1270 (1110)	131 (23)				
9	188 (31)	357 (50)	955 (607)	1960 (1880)	3820 (2020)	2140 (1840)		131 (4)	139 (7)	113 (12)
10	430 (58)	593 (129)	3820 (3480)	3900 (3820)	5840 (3240)	8370 (8030)	6010 (4280)			

Table 30. 1983 lake manganese concentrations (ug/l). (Dissolved manganese values are in parentheses.)

Depth (m)	Date									
	May 19	Jun 15	Jul 19	Aug 17	Sep 20	Oct 5	Oct 12	Oct 19	Oct 27	Nov 9
1										
2	23 (2)	22 (4)	12 (2)	14 (1)	70 (2)	38 (5)				
3							148 (2)	124 (3)	80 (2)	40 (13)
4	27 (4)	31 (2)	18 (2)	20 (1)	66 (1)	36 (2)				
5										
6	25 (24)	36 (3)	31 (4)	38 (1)	86 (2)	56 (2)	162 (2)	135 (3)	80 (2)	43 (10)
7	25 (2)	72 (4)	348 (295)	800 (633)	96 (3)					
8	52 (16)	110 (14)	818 (742)	1520 (1455)	2180 (1240)	334 (21)				
9	66 (41)	435 (418)	1350 (1310)	1730 (1630)	2160 (1920)	372 (372)		134 (2)	80 (5)	46 (12)
10	165 (132)	590 (540)	1590 (1560)	1990 (1890)	2280 (2060)	4140 (4140)	4940 (3880)			

Although extremely high concentrations of phosphorus developed in the hypolimnion, the response of the epilimnion to this build-up was dependent on the delivery of this phosphorus to the epilimnion. The thermal stratification which limited the replenishment of hypolimnetic oxygen also limited the upward transport of hypolimnetic phosphorus. However, thermal stratification was not a static condition. The hypolimnetic boundary generally showed a progressive deepening throughout the summer (see Figure 24). Periods of high winds induced partial mixing of the epilimnion and hypolimnion and caused rapid downward movement of the epilimnion boundary. This resulted in pulses of phosphorus transport to the epilimnion.

A vertical transport model described by Stauffer and Lee (1974) was applied to Lake Iroquois to quantify phosphorus loading to the epilimnion from the metalimnion and hypolimnion during 1982 and 1983. This model used temperature profile data from two successive dates and the total phosphorus profile from the initial date to estimate phosphorus transport. Turbulent exchange coefficients for phosphorus were assumed to be the same as those for heat (see Methods section for a more detailed explanation).

Calculated epilimnetic phosphorus gains due to vertical transport are listed in Table 31. Total vertical phosphorus transport during 1982 and 1983 was estimated to be 84 and 489 kg, respectively. The much greater transport during 1983 reflected the greater duration and magnitude of sediment phosphorus release which occurred.

Some of the vertical transport was not directly derived from sediment phosphorus release. A portion was accounted for by simple epilimnetic volume expansion. Some of the total transport was also due to sedimenting phosphorus, still in suspension, which was returned to the epilimnion. These factors were of greater significance during 1982 when estimated total transport was relatively low. Another consideration was that substantial algal productivity occurred in the metalimnion and hypolimnion

Table 31. 1982 and 1983 epilimnetic phosphorus gains due to vertical transport.

1982

$Date_0$	$Date_1$	$*De_0$	$*De_1$	Epilimnetic Phosphorus Gain (kg)
7-12	7-20	4.3	4.3	3.9
7-20	7-27	4.5	6.0	16.9
7-27	8-2	6.0	6.2	2.0
8-9	8-20	6.2	7.1	23.1
8-20	8-24	7.1	7.3	3.5
8-24	8-26	7.3	7.7	7.1
8-26	9-2	7.7	9.2	22.2
9-16	9-23	8.8	9.6	4.8
			TOTAL:	83.5

1983

$Date_0$	$Date_1$	$*De_0$	$*De_1$	Epilimnetic Phosphorus Gain (kg)
6-22	6-29	2.2	4.4	22.0
7-5	7-13	3.7	5.0	10.5
7-19	7-28	4.0	5.2	19.0
7-28	8-5	5.2	5.3	4.5
8-9	8-17	5.2	6.1	8.1
8-31	9-7	5.5	6.1	4.9
9-7	9-14	6.1	6.3	7.8
9-14	9-20	6.3	7.3	29.1
9-20	9-28	7.3	9.4	150.6
10-5	10-12	8.8	9.0	52.0
10-12	10-19	9.0	11.3	180.3
			TOTAL:	488.8

$*De_0$ = depth of epilimnion (m) at $Date_0$
$*De_1$ = depth of epilimnion (m) at $Date_1$

during 1983, and probably also during 1982. Thus, while phosphorus transport to the epilimnion controlled the quality of epilimnetic water, it did not completely control productivity in the lake.

Phosphorus sedimentation due to reformation of iron precipitates greatly limits the impact of vertically transported phosphorus. The ferrous iron which accompanies the phosphorus released from the sediments forms a hydrous ferric oxide precipitate upon contact with oxygenated water and immediately binds the phosphorus. This process occurs in a matter of minutes. A longer period (days) is required for this initially collodial ferric bound phosphorus to form a settleable precipitate (Stauffer 1981). Stauffer states that if the atomic Fe/P ratio exceeds 1.8, phosphorus is nearly quantitatively precipitated as ferric-phosphate when ferrous iron is oxidized by dissolved oxygen during diffusion upward through the overlying water column. As a consequence, phosphate released during stagnation into the anoxic bottom waters of deep non-calcareous lakes plays no subsequent role in the phosphorus economy of the trophogenic layer. Atomic Fe/P ratios at the 8-10 m depths in Lake Iroquois during the stratified summer period in 1983 generally were far in excess of 1.8 (Table 32). Recapture of phosphorus by iron would, therefore, be expected to be highly effective. However, contrary to Stauffer's statement, sediment phosphorus releases did appear to make a very significant contribution to epilimnetic phosphorus concentrations and resultant increases in epilimnetic algal populations in Lake Iroquois. Figure 30 shows that epilimnetic phosphorus concentrations made gains between late June and late September of all three years, especially 1983 when greatest sediment phosphorus releases occurred. External phosphorus loading at this time of the year was insignificant. Epilimnetic chlorophyll-a concentrations also showed a dramatic increase during this period in 1983 (Figure 34). Epilimnetic chlorophyll-a concentrations were not measured in 1981 and 1982 when composite samples were taken to twice the Secchi disc depth.

Table 32. Fe/P ratios (concentrations in ug/l).

	Depth (m)								
	8			9			10		
Date	Diss. Fe	Diss. P	Fe/P	Diss. Fe	Diss. P	Fe/P	Diss. Fe	Diss. P.	Fe/P
6-15-83	18	9	2.0	50	10	5.0	129	4	32.2
7-19-83	18	3	6.0	607	8	75.9	3480	252	13.8
8-17-83	650	38	17.1	1880	142	13.2	3820	352	10.8
9-20-83	1110	29	38.3	2020	224	9.0	3240	416	7.8
10-5-83	-	-	-	1840	185	9.9	8030	832	9.6
10-12-83	-	-	-	-	-	-	4280	378	11.3

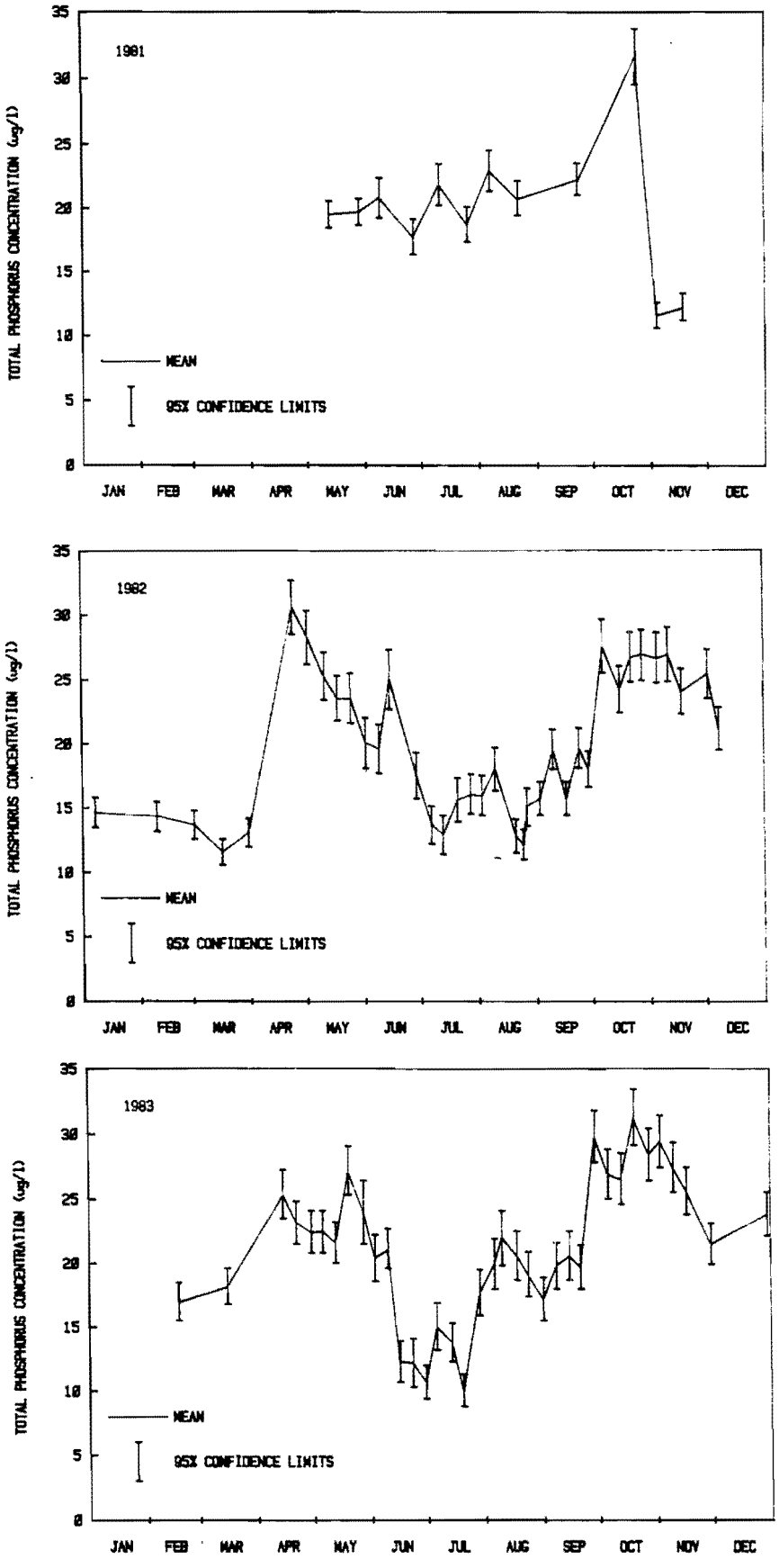


Figure 30. Fluctuations in mean epilimnetic phosphorus concentrations.

However, chlorophyll-a concentrations from these composite samples did show a similar but smaller increase during the June-September period of 1982. A somewhat different pattern of chlorophyll-a concentrations appeared in 1981 when peaks occurred in July and October and relatively low values were found in August and September. The limited sampling conducted during this year did not allow an evaluation of the internal phosphorus cycling patterns which might have accounted for this variation.

The apparent response of epilimnetic algae to the delivery of phosphorus which in theory was iron bound may indicate that algae are capable of obtaining a portion of this phosphorus. A second possibility is that vertically transported particulate organic phosphorus may be a significant source of phosphorus for epilimnetic algae. Particulate phosphorus ranged from 15%-82% (mean = 41%) of total phosphorus in anoxic hypolimnetic samples during the summer of 1983 (Table 33). Despite anoxic conditions, particulate iron ranged from 2%-45% (mean = 19%) of total iron in these same samples. The form and phosphorus binding capabilities of this iron under anoxic conditions is uncertain and so a quantitative estimate of organically bound phosphorus was not made. However, algae, bacteria, and detritus were observed in the hypolimnion and must account for some portion of the particulate phosphorus measured. Algae produced in the hypolimnion and metalimnion, where phosphorus was more readily available, and then transported to the epilimnion may have supplied phosphorus to epilimnetic algae. Hypolimnetic algae could be relatively phosphorus rich due to luxury uptake of phosphorus. The highest chlorophyll-a concentration found during 1983 (120 ug/l) occurred at the 9 m depth at the end of June (Figure 4). Maximum chlorophyll-a concentrations were found at progressively shallower depths after this date. Hypolimnetic phosphorus contained in suspended bacteria and detritus might also have become partially available to epilimnetic algae following vertical transport events. Purple-pink bacteria were regularly noted on filters used in sampling anoxic hypolimnetic water.

Table 33. Particulate phosphorus and iron in anoxic hypolimnetic samples.

Date	Depth (m)	Total P (ug/l)	Particulate P (Total P-Diss. P) (ug/l)	Particulate P (% of Total P)	Total Fe (ug/l)	Particulate Fe (Total Fe-Diss. Fe) (ug/l)	Particulate Fe (% of Total Fe)
7-19-83	9	44	36	82	955	348	36
7-19-83	10	326	74	23	3820	340	9
8-17-83	8	70	34	49	785	135	17
8-17-83	9	196	54	28	1960	80	4
8-17-83	10	415	63	15	3900	80	2
9-20-83	10	816	400	49	5840	2600	45

Macrophyte Phosphorus Release

The phosphorus contained in aquatic macrophyte tissue is derived largely from the sediment. A portion of this phosphorus can potentially be released to the water column during the period of macrophyte senescence and decay. The potential for phosphorus release in Lake Iroquois is discussed in the Aquatic Macrophytes section of this report. The total phosphorus content of macrophytes during the period of peak biomass in 1982 was estimated to be approximately 70 kg. The response of the lake to any release of this phosphorus could not be easily isolated due to other phosphorus releases and losses occurring simultaneously. Most macrophyte phosphorus release would be expected to occur in the late summer and fall.

Turbulent Resuspension of Sediment Phosphorus

Turbulent resuspension of sediment phosphorus by wave action and wind induced currents has been associated with increases in water column phosphorus concentrations (Bannerman *et al.*, 1975, Lam and Jaquet, 1976; DeGroot, 1981). Both suspension of particulate phosphorus and release of dissolved interstitial phosphorus may occur. The shallow northern end of Lake Iroquois is susceptible to such releases, especially since the prevailing wind direction is from the south and the lake's maximum fetch is approximately along a north-south axis. Highly turbid conditions were noted at the lake's north end on days with strong southerly winds during the fall of 1982.

Turbulent resuspension of deep water sediment probably also occurs to a lesser degree, especially during the spring and fall periods of complete lake circulation. The spring and fall periods of whole lake circulation follow periods of prolonged stratification and also have the highest mean wind velocities of the open water season. Davis (1973) found large peaks in rates of pollen redeposition during the spring and fall in a seasonally stratified Michigan lake similar in depth to Lake Iroquois. This

was attributed to resuspension of sediment throughout the lake including deep water areas where at least the upper millimeter of sediment was estimated to be resuspended by turbulent water movement. The impact of turbulent resuspension of sediment was not closely evaluated for Lake Iroquois. However, the potential influence of this phosphorus source (as well as macrophyte phosphorus release) has been given consideration in interpretation of lake phosphorus mass fluctuations in the Lake Response to External and Internal Phosphorus Loading section of this report.

LAKE RESPONSE TO EXTERNAL AND INTERNAL PHOSPHORUS LOADING

Fluctuations in Lake Phosphorus Content

Figure 29 shows fluctuations in the total lake phosphorus mass during 1981-1983. The changes measured were the net result of a variety of simultaneously occurring gains and losses. The probable dominant factors controlling the observed fluctuations are discussed in the following paragraphs.

Increases in phosphorus mass occurred in the spring (April and May) of 1982 and 1983 (data was not collected during the early months of 1981). This was largely a response to the high rate of external phosphorus loading which occurred at this time (Figure 22). Runoff from snowmelt and spring rains accounted for most of this loading. Net external phosphorus loading (total phosphorus inputs minus outflow losses) during March - April of 1982 and March - May of 1983 were similar, yet lake phosphorus mass for these months during 1982 and 1983 showed maximum net gains of about 100 and 50 kg, respectively. This difference was probably due to the longer time period over which high phosphorus inputs occurred in 1983, which allowed for more sedimentation to occur prior to attainment of peak phosphorus mass. The sharp increase in phosphorus mass during 1982 was probably also influenced by the unusually high wind velocities during April which would have caused sediment resuspension and limited sedimentation of phosphorus inputs. The mean wind velocity that month was 13.3 mph, compared to a normal April mean of 9.4 mph, based on National Weather Service records.

Lake phosphorus mass losses became dominant during May and June of 1982 and June of 1983 due to rapidly declining inputs and effective sedimentation, partially from diatom blooms (see Tables 34 and 35). During 1981, phosphorus mass showed little significant variation from May-June. The limited data collected that year made interpretation difficult.

Lake phosphorus mass gains again became dominant in late June or July of all three years in response to hypolimnetic sediment phosphorus release which resulted from the development

of anoxia at the sediment surface (see Figure 24). In 1981, a maximum net gain of about 50 kg occurred between June and early August. A slight decline occurred by September, prior to turnover. In 1982, a similar maximum net gain of about 50 kg occurred between mid-July and early August. However, the 1982 August peak was followed by a sharp drop, after which phosphorus mass generally remained below the minimum value reached in July. The late summer phosphorus mass declines in 1981 and 1982 resulted from partial recoveries of hypolimnetic oxygen concentrations which allowed ferric phosphate precipitation and restricted continued sediment release. Recovery of hypolimnetic oxygen concentrations was more pronounced and occurred earlier in 1982, accounting for the much greater declines in phosphorus mass that occurred.

In 1983, summer increases in lake phosphorus mass far exceeded those in 1981 and 1982 due to the much more extensive and prolonged hypolimnetic anoxia which occurred. A net gain of about 325 kg was observed between mid-June and early September. A sharp drop in phosphorus mass during late September was associated with a substantial downward shift in the epilimnetic boundary (Figure 24) which resulted in oxygen delivery to the hypolimnion and ferric phosphate precipitation. A partial recovery of phosphorus mass followed, due to hypolimnetic restabilization.

Phosphorus mass changes during and after fall turnover varied considerably among the three years. Turnover during the three years occurred sometime between the dates listed below.

1981: September 22 - October 6

1982: September 28 - October 6

1983: October 12 - October 19

In 1981, an increase in phosphorus mass occurred in mid-October after turnover. This may have been a result of turbulent sediment resuspension. Strong southerly winds occurred on three of the five days preceding this sampling date. One of these days had a mean wind velocity of 15.7 mph which exceeded all prior daily means since mid-April. Release of phosphorus by

aquatic macrophytes might also have played a role in this increase. The mid-October samples were taken at least 18 days after turnover, so the lake's behavior immediately after turnover was uncertain. A rapid decline in phosphorus mass occurred by early November, possibly due to a rapid sedimentation of previously resuspended particulate phosphorus and diatom bound phosphorus (chlorophyll-a concentrations shown in Figure 34 suggest a decline of a diatom bloom at this time). Mean monthly wind speeds for October and November were slightly below normal and water temperature at the beginning of November was slightly below that found in 1982 and 1983.

Immediately following fall overturn in 1982 a gain in phosphorus mass of about 50 kg was observed. The hypolimnion was fairly well oxygenated prior to turnover, so phosphorus losses through ferric phosphate precipitation would not have occurred to significantly counter any gains. Turbulent resuspension of sediment phosphorus was the probable cause of the observed gain. Phosphorus release by aquatic macrophytes may also have contributed. This gain in phosphorus mass was maintained through October and November with the first significant decline occurring in early December. The mean monthly wind speed during October was slightly below normal, while the November mean was considerably above normal. High turbidity was noted at the lake's north end on dates in November with strong south winds.

During turnover in 1983, a large decline in phosphorus mass occurred due to sedimentation of ferric phosphate precipitates formed during oxygenation of the hypolimnion which had remained partially anoxic until this time. A gradual decline in phosphorus mass continued through October and November. Mean wind speed was above normal in October and below normal in November. High rates of external phosphorus loading occurred during November and December providing a net external loading (inputs minus outflow losses) of about 70 kg. This loading probably reduced the net phosphorus mass loss which occurred, and accounted for the slight gain which occurred between late November and late December.

Fluctuations in mean epilimnetic phosphorus concentrations during 1981-1983 are shown in Figure 30. During periods of complete lake circulation, epilimnetic phosphorus concentrations reflected lake phosphorus mass since the whole lake was considered to represent the epilimnion at this time. Generally, during the stratified summer period, increases in epilimnetic phosphorus concentrations were much less than increases in lake phosphorus mass since a great deal of phosphorus was confined to the metalimnion and hypolimnion. This was especially noticeable in 1983. Gains and losses in the lake phosphorus mass and epilimnetic phosphorus concentrations during the stratified period did not correspond well, since factors contributing to epilimnetic losses contributed to hypolimnetic gains. For example, a decline in mean epilimnetic phosphorus concentration occurred in August of 1983, at which time lake phosphorus mass showed a sharp increase. A single small vertical transport event during this decline (Figure 34) suggested a low mixing rate. The lack of vertical mixing apparently promoted sedimentation of epilimnetic phosphorus and limited phosphorus replenishment through vertical transport. An algal bloom in progress at the time (Figure 34) also contributed to phosphorus sedimentation. This resulted in an increased delivery of oxygen demanding organic material to the hypolimnion at a time when oxygen replenishment through mixing was reduced. Release of sediment phosphorus was promoted by the resultant oxygen depletion and caused large increases in hypolimnetic phosphorus concentrations and total lake phosphorus mass. In late September of 1983, a large increase in epilimnetic phosphorus concentration resulted from a major vertical transport event. This greatly reduced hypolimnetic phosphorus concentrations through ferric phosphate precipitation and produced a large net loss in lake phosphorus mass.

Between mid-August and late September of 1982, lake phosphorus mass showed no significant net gain while mean epilimnetic phosphorus concentrations climbed from about 12 ug/l to 19 ug/l (Figures 29 and 30). Lake volume losses at this time

were substantial (greater than 6%), and could account for a small portion of the difference observed (Figure 21). Hypolimnetic phosphorus concentrations declined during this period (Figure 28). Four vertical transport events occurred (Figure 34), and could have accounted for most, if not all, of the epilimnetic phosphorus concentration increase observed. Phosphorus release by aquatic macrophytes may also have contributed. Substantial macrophyte decay was noted during September of 1982.

Short-term Phosphorus Budget Terms

Short-term phosphorus budget terms provided an alternate means of evaluating lake response to external and internal phosphorus loading. Tables 34-37 list short-term phosphorus budget terms on both an interval and a cumulative basis for 1982 and 1983. A change in lake phosphorus mass term is also included in these tables, which allows the calculation of a net phosphorus sedimentation term. The data in these tables is graphically presented in Figures 31 and 32. These figures are only briefly discussed here, since much of the discussion in the preceding section is applicable.

Figure 31 shows net sedimentation during ten day intervals. Positive values indicate net phosphorus deposition, while negative values indicate net phosphorus release. Winter sedimentation rates were generally positive, but quite low. Deposition increased in spring in response to inputs of particulate phosphorus by streamflow and increased sedimentation resulting from spring algal blooms. Net releases became frequent during the summer period of thermal stratification, especially in 1983, due to sediment phosphorus release. Periods of greatest net deposition during both years followed extended periods of sediment phosphorus release and resulted largely from sedimentation of ferric phosphate precipitates. Net deposition following turnover in 1983 exceeded that in 1982 due to sedimentation of much greater streamflow inputs in 1983 and the greater importance of sediment resuspension in 1982.

Table 34. 1982 short-term P budget terms (kg P).

INTERVAL (Julian Days)	INFLOW	PRECIPITATION	GROUNDWATER	SEPTIC SYSTEMS	TOTAL INPUTS	OUTFLOW	LAKE	NET SEDIMENTATION
1-30	18.2	1.64	0.69	0.12	20.65	5.69	+1.8	+13.2
31-60	8.1	1.91	0.69	0.12	10.82	4.50	-2.5	+8.8
61-90	101.8	1.46	0.69	0.12	104.07	9.94	+2.5	+91.6
91-110	28.7	2.27	0.46	0.08	31.51	5.58	+83.8	-13.52
111-120	4.7	0.57	0.23	0.04	5.54	2.79	-0.2	+3.0
121-130	2.4	0.93	0.23	0.04	3.60	3.10	-20.2	+20.7
131-140	2.1	1.45	0.23	0.04	3.82	2.29	-9.3	+10.8
141-150	1.6	0.05	0.23	0.04	1.92	0.87	-8.1	+9.2
151-160	2.6	2.09	0.23	0.04	4.96	2.08	-14.8	+17.7
161-170	0.8	1.26	0.23	0.04	2.33	1.37	+16.2	-15.2
171-180	6.9	2.46	0.23	0.04	9.63	2.47	-14.0	+21.2
181-190	1.3	0.27	0.23	0.18	1.98	1.49	-7.6	+8.1
191-200	0.2	0.09	0.23	0.18	0.70	1.54	+3.0	-3.8
201-210	0.4	1.20	0.23	0.18	2.01	1.61	+12.7	-12.3
211-220	0.5	1.52	0.23	0.18	2.43	1.16	+22.8	-21.5
221-230	0.6	0.33	0.23	0.18	1.34	1.60	-42.5	+42.2
231-240	0.5	1.19	0.23	0.18	2.10	0.98	+6.2	-5.1
241-250	0.2	0.56	0.23	0.18	1.17	0.77	-11.0	+11.4
251-260	0.3	0.81	0.23	0.18	1.52	0.96	-11.4	+12.0
261-270	0.4	0.22	0.23	0.18	1.03	0.62	+6.6	-6.2
271-280	0.5	1.29	0.23	0.18	2.20	1.02	+43.8	-42.6
281-290	0.6	1.47	0.23	0.18	2.48	0.91	-10.5	+12.1
291-300	0.5	0.00	0.23	0.18	0.91	0.68	+8.5	-8.3
301-310	1.6	1.89	0.23	0.04	3.76	1.94	-0.2	+2.0
311-320	1.5	0.30	0.23	0.04	2.07	0.87	-11.0	+12.2
321-330	2.3	0.82	0.23	0.04	3.39	0.68	+3.2	-0.5
331-340	1.9	0.39	0.23	0.04	2.56	0.35	-14.6	+16.8
341-365	6.8	1.11	0.58	0.10	8.59	0.80	-7.0	+14.8
TOTALS	198.0	29.55	8.40	3.14	239.09	58.66	+26.2	+154.4

Table 35. 1983 short-term P budget terms (kg P).

INTERVAL (Julian Days)	INFLOW	PRECIPITATION	GROUNDWATER	SEPTIC SYSTEMS	TOTAL INPUTS	OUTFLOW	LAKE	NET SEDIMENTATION
1-30	12.19	1.11	1.20	0.12	14.62	0.43	-4.3	+18.5
31-60	13.90	0.77	1.01	0.12	15.80	4.46	+0.9	+10.4
61-90	33.61	2.22	0.72	0.12	36.67	13.58	+26.1	-3.0
91-110	21.08	2.24	0.44	0.08	23.84	8.80	+9.4	+5.6
111-120	74.76	0.89	0.22	0.04	75.91	20.29	-4.5	+60.1
121-130	23.77	1.17	0.22	0.04	25.20	11.24	-4.4	+18.4
131-140	9.27	0.29	0.22	0.04	9.82	6.51	+28.8	-25.5
141-150	13.96	1.80	0.22	0.04	16.02	9.50	-13.7	+20.2
151-160	11.44	0.88	0.22	0.04	12.58	7.22	-19.9	+25.3
161-170	2.28	0.33	0.22	0.04	2.87	1.98	-9.6	+10.5
171-180	0.53	0.55	0.22	0.04	1.34	0.55	+22.7	-21.9
181-190	0.25	0.51	0.15	0.18	1.09	0.26	+35.7	-34.9
191-200	0.10	0.07	0.14	0.18	0.49	0.39	0	+0.1
201-210	0.014	0.37	0.14	0.18	0.704	0.47	+40.3	-40.1
211-220	0.34	2.43	0.06	0.18	3.01	0.47	-5.2	+7.7
221-230	0.22	0.72	0.04	0.18	1.16	0.49	+71.5	-70.8
231-240	0.09	0.88	0.04	0.18	1.19	0.50	+57.1	-56.4
241-250	0.07	0.29	0.15	0.18	0.69	0.51	+60.8	-60.6
251-260	0.03	0.10	0.20	0.18	0.51	0.55	+27.8	-27.8
261-270	1.42	2.37	0.20	0.18	4.17	0.57	-145.4	+149.0
271-280	1.05	1.20	0.30	0.18	2.73	0.53	+77.5	-75.3
281-290	1.19	0.70	0.34	0.18	2.41	0.68	-135.8	+137.5
291-300	0.75	0.55	0.34	0.18	1.82	0.61	-64.6	+65.8
301-310	26.78	1.49	0.27	0.04	28.58	0.74	+3.1	+24.7
311-320	12.56	1.25	0.22	0.04	14.07	4.89	-10.7	+19.9
321-330	18.35	1.57	0.22	0.04	20.18	12.75	-17.5	+24.9
331-340	77.68	3.59	0.65	0.14	82.06	28.23	+11.2	+42.6
TOTALS	357.68	30.34	8.37	3.14	399.53	137.20	+37.3	+224.9

Table 36. 1982 short-term cumulative P budget terms (kg P).

INTERVAL (Julian Days)	TOTAL INPUTS	OUTFLOW	LAKE	NET SEDIMENTATION
1-30	20.65	5.69	+ 1.8	+13.2
31-60	31.47	10.19	- 0.7	+22.0
61-90	135.54	20.13	+51.1	113.6
91-110	167.05	25.71	-85.6	+55.7
111-120	172.59	28.50	+85.4	+58.7
121-130	176.19	31.60	+65.2	+79.4
131-140	180.01	33.89	+55.9	+90.2
141-150	181.93	34.76	+47.8	+99.4
151-160	186.90	36.84	+33.0	+117.1
161-170	189.22	38.21	+49.2	+101.9
171-180	198.85	40.68	+35.2	+123.1
181-190	200.83	42.17	+27.6	+131.2
191-200	201.53	43.71	+30.6	+127.4
201-210	203.54	45.32	+43.3	+115.1
211-220	205.97	46.48	+66.1	+93.6
221-230	207.31	48.08	+23.6	+135.8
231-240	209.41	49.06	+29.8	+130.7
241-250	210.58	49.83	+18.8	+142.1
251-260	212.10	50.79	+7.4	+154.1
261-270	213.13	51.41	+14.0	+147.9
271-280	215.33	52.43	+57.8	+105.3
281-290	217.81	53.34	+47.3	+117.4
291-300	218.72	54.02	+55.8	+109.1
301-310	222.48	55.96	+55.6	+111.1
311-320	224.55	56.83	+44.6	+123.3
321-330	227.94	57.51	+47.8	+122.8
331-340	230.50	57.86	+33.2	+139.6
341-365	239.09	58.66	+26.2	+154.4

Table 37. 1983 short-term cumulative P budget terms (kg P).

INTERVAL (Julian Days)	TOTAL INPUTS	OUTFLOW	LAKE	NET SEDIMENTATION
1-30	14.62	0.43	-4.3	+18.5
31-60	30.42	4.89	-3.4	+28.9
61-90	67.09	18.47	+22.7	+25.9
91-110	90.93	27.27	+32.1	+31.5
111-120	166.84	47.56	+27.6	+91.6
121-130	192.04	58.80	+23.2	+110.0
131-140	201.86	65.31	+52.0	+84.5
141-150	217.88	74.81	+38.3	+104.7
151-160	230.46	82.03	+18.4	+130.0
161-170	233.33	84.01	+8.8	+140.5
171-180	234.67	84.56	+31.5	+118.6
181-190	235.76	84.82	+67.2	+83.7
191-200	236.25	85.21	+67.2	+83.8
201-210	236.95	85.68	+107.5	+43.7
211-220	239.96	86.15	+102.3	+51.4
221-230	241.12	86.64	+173.8	-19.8
231-240	242.31	87.14	+230.9	-76.2
241-250	243.00	87.65	+291.7	-136.8
251-260	243.51	88.20	+319.5	-164.6
261-270	247.68	88.77	+174.1	-15.6
271-280	250.41	89.30	+251.6	-90.9
281-290	252.82	89.98	+115.8	+47.0
291-300	254.64	90.59	+51.2	+112.8
301-310	283.22	91.33	+54.3	+137.5
311-320	297.29	96.22	+43.6	+157.4
321-330	317.47	108.97	+26.1	+182.3
331-365	399.53	137.20	+37.3	+224.9

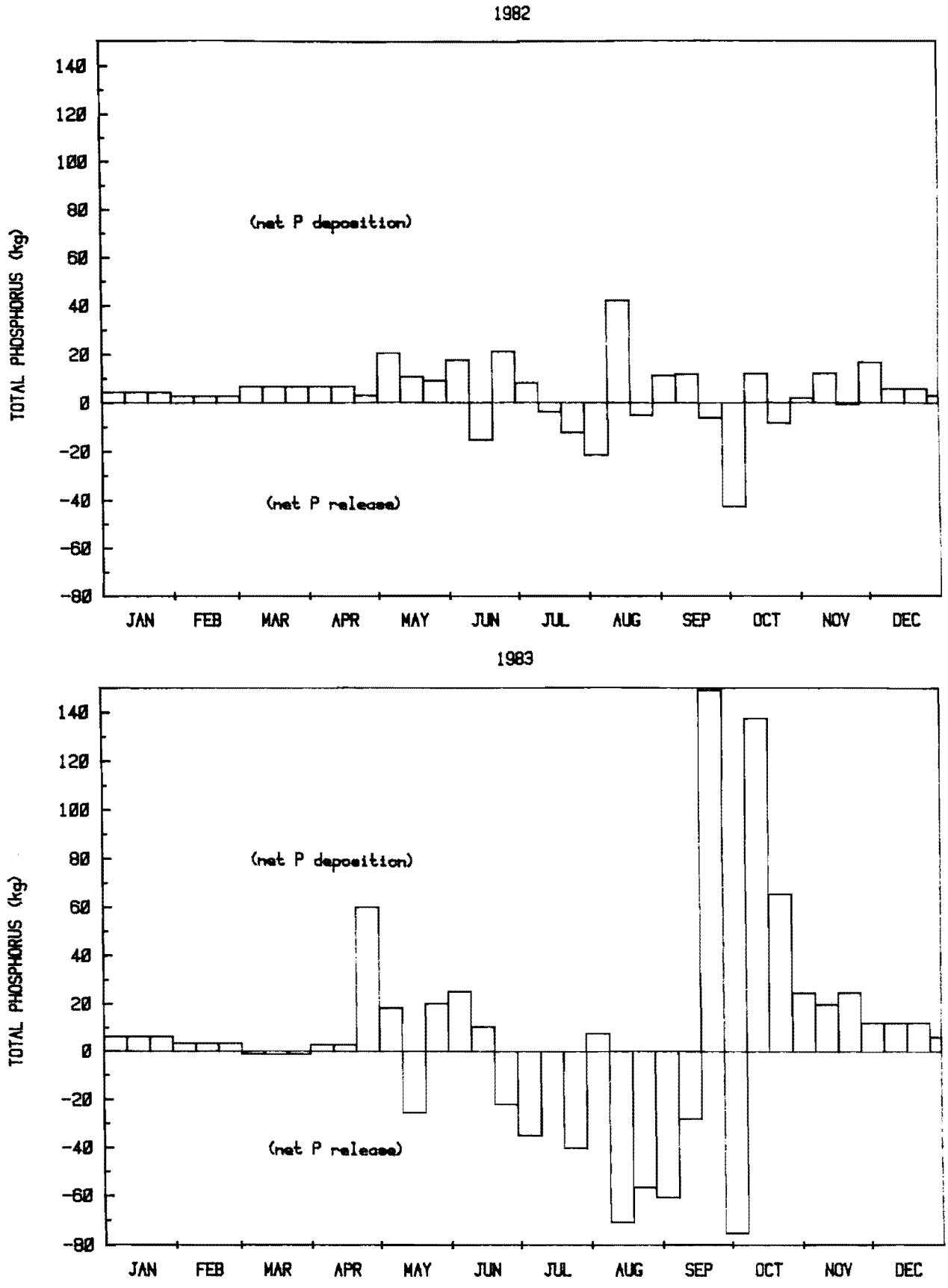


Figure 31. Net sedimentation plots (10-day intervals).

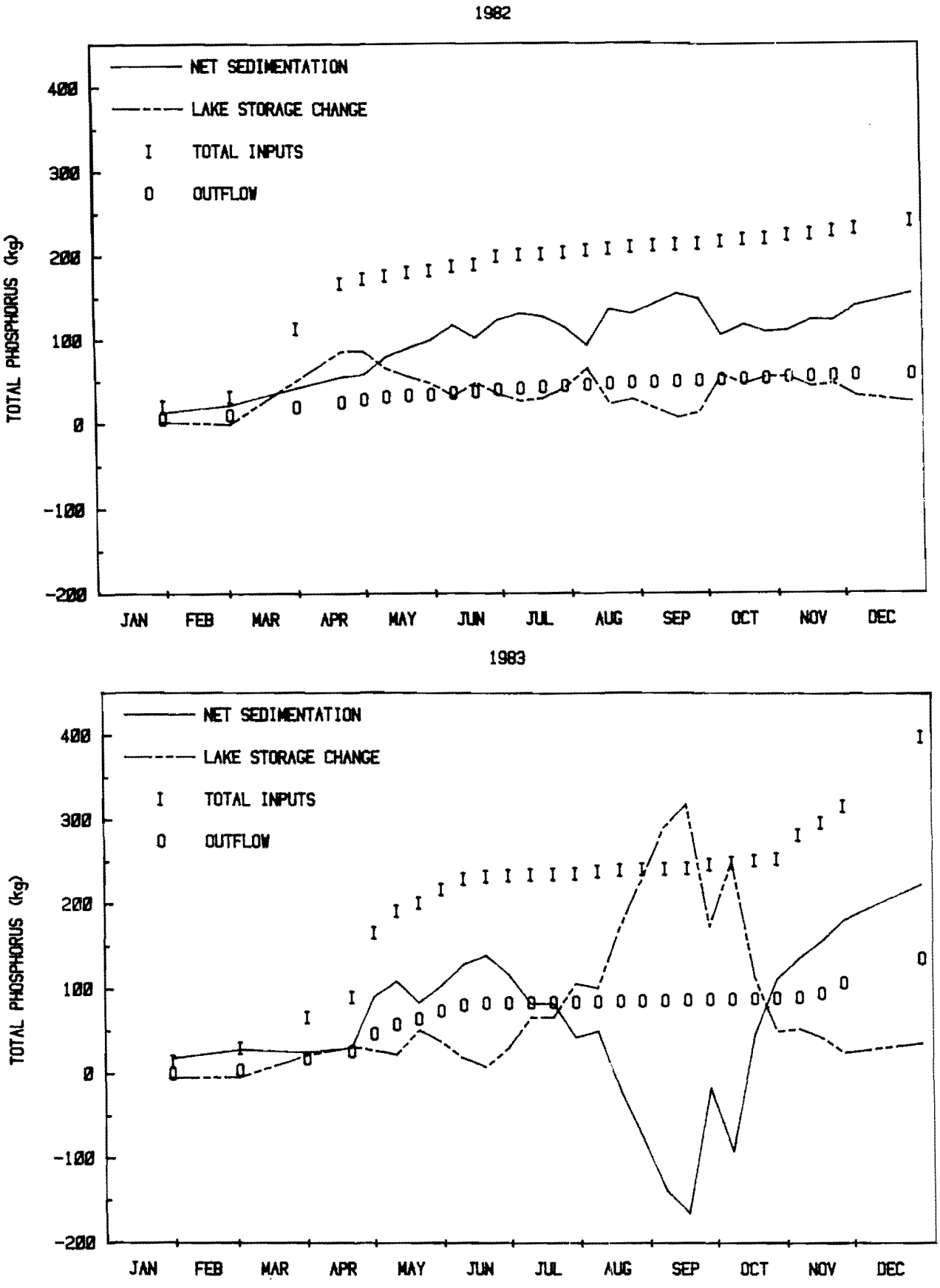


Figure 32. Cumulative lake phosphorus budget plots.

Figure 32 depicts the cumulative budget terms. The dominance of internal phosphorus cycling in the control of summer lake phosphorus fluctuations is evident.

Phosphorus - Chlorophyll-a Relationships

Phosphorus is widely agreed to be the nutrient most frequently limiting phytoplankton productivity in temperate zone lakes (Golterman, 1980). Nitrogen has less frequently been determined to be limiting. A total nitrogen to total phosphorus ratio greater than 12 (Dillon and Rigler, 1974) is generally indicative of phosphorus limitation. Epilimnetic nitrogen to phosphorus ratios for Lake Iroquois equaled or exceeded this value on all dates sampled, with an average value of 23 (Table 38). Lowest ratios (12-13) were found in April and May of 1983.

A regression plot of the logarithm of mean epilimnetic phosphorus concentrations vs. the logarithm of chlorophyll-a concentrations during 1983 is shown in Figure 33. The regression gave an R^2 value of 0.48. A high degree of correlation would not be expected due to numerous other factors influencing algal populations (temperature, light availability, circulation patterns, phosphorus availability, other temporary nutrient limitation, epilimnetic total nitrogen to total phosphorus ratios, etc.). Chlorophyll-a concentrations during 1981 and 1982 were based on composite samples taken to twice the Secchi disc depth rather than discrete samples from each meter, so phosphorus - chlorophyll relationships could not be well evaluated by regressions for these years.

Figure 34 shows epilimnetic phosphorus concentrations, chlorophyll-a concentrations, and vertical transport of phosphorus. Chlorophyll-a concentrations for 1982 were based on composite samples taken to twice the Secchi disc depth, while 1983 chlorophyll a concentrations were based on volume-weighted mean epilimnetic concentrations. A general correspondence between major peaks and troughs for phosphorus and chlorophyll-a concentrations is seen for both years. In 1982, chlorophyll-a fluctuations appeared to lag behind phosphorus fluctuations by

Table 38. Epilimnetic total nitrogen to total phosphorus ratios.

Date	N:P Ratio
7-01-82	20
7-29-82	19
9-16-82	28
10-21-82	21
12-01-82	19
4-14-83	12
5-19-83	13
6-15-83	26
7-19-83	43
8-17-83	17
9-20-83	28
10-19-83	29

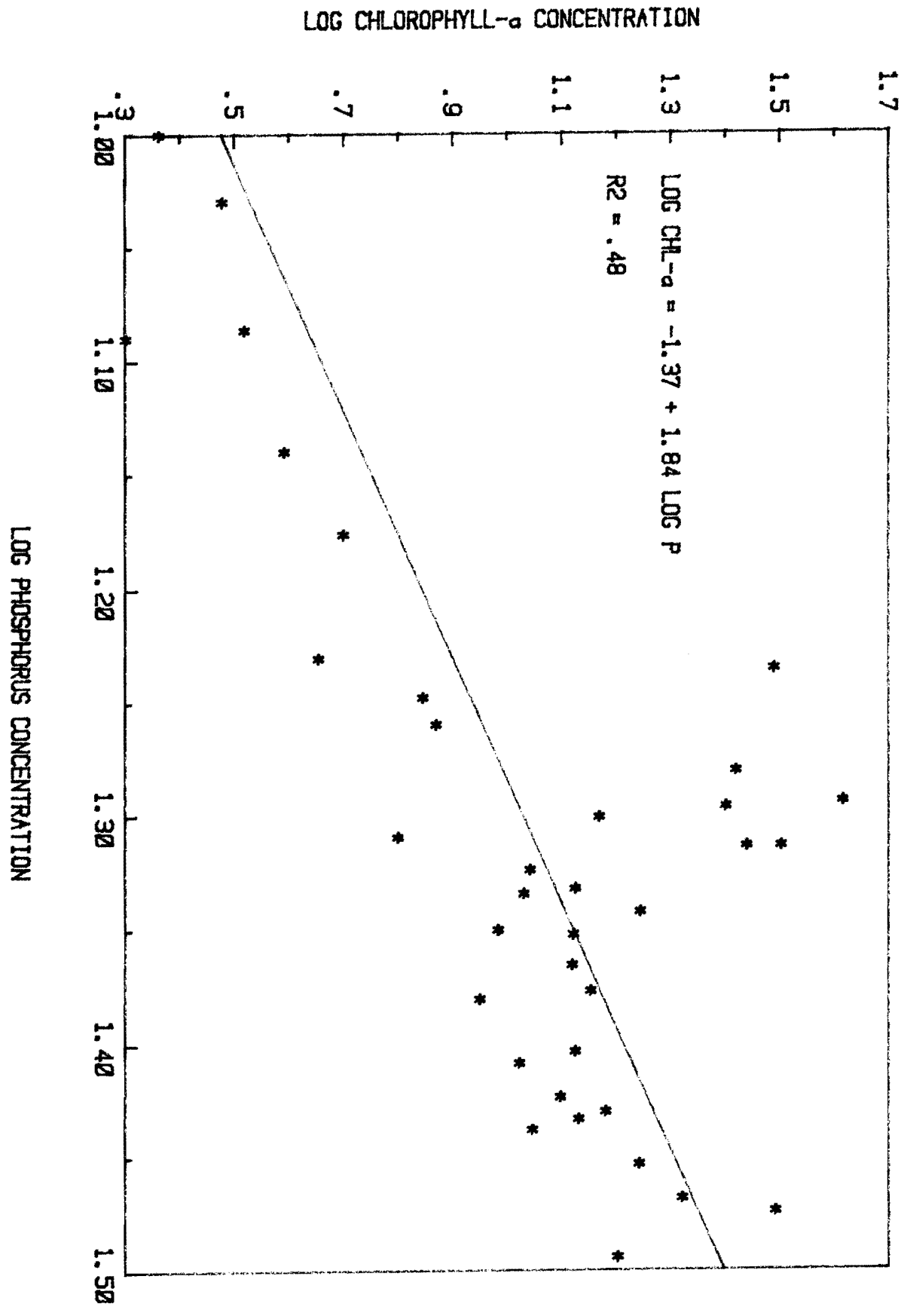


Figure 33. Regression of chlorophyll vs. phosphorus concentrations (ug/l) during 1983.

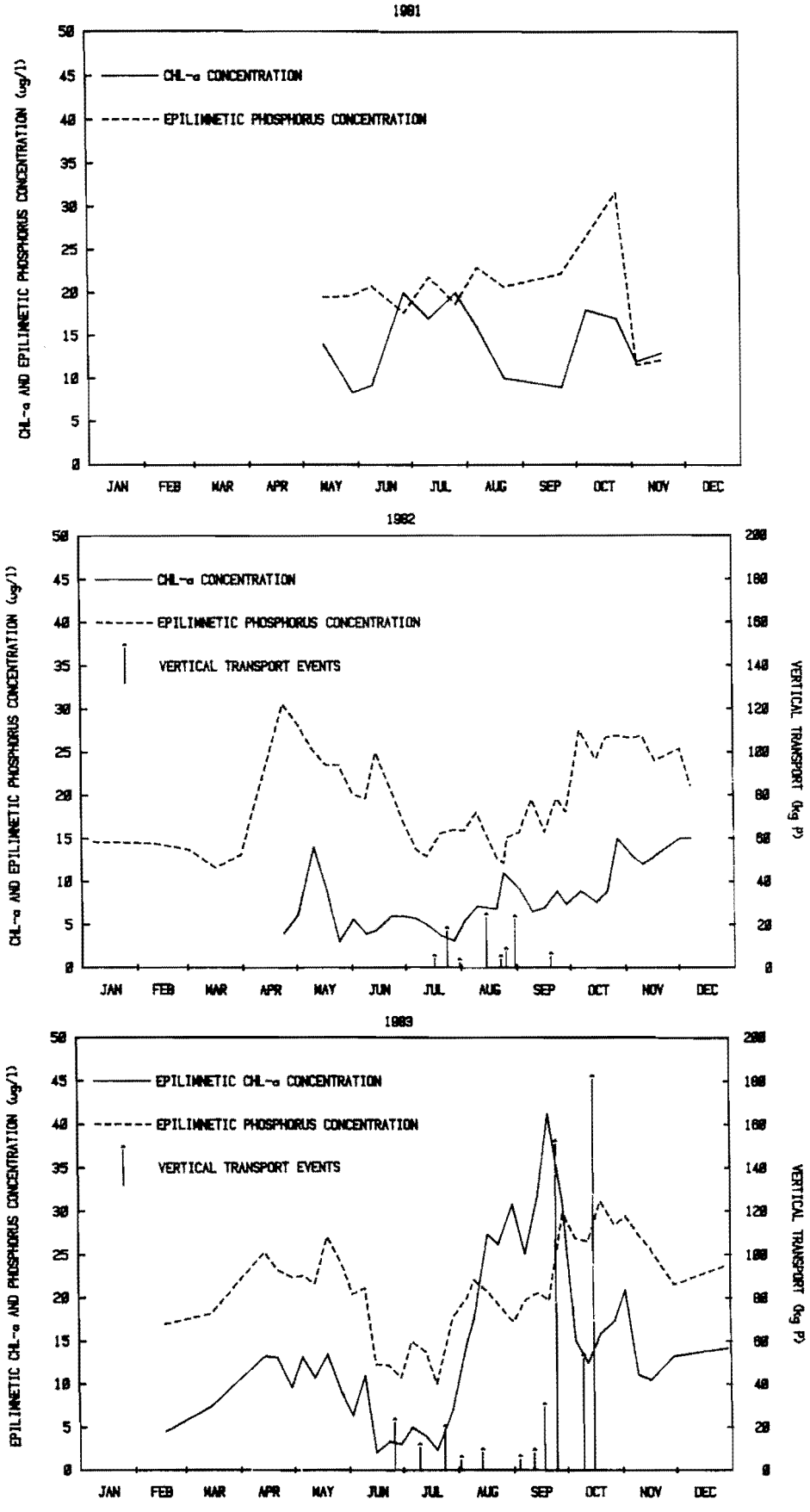


Figure 34. Chlorophyll-a concentrations, epilimnetic phosphorus concentrations, and vertical transport events.

about two weeks. A similar lag period was not apparent in 1983. This may partially reflect the difference in chlorophyll-a sampling depths between the two years.

A much broader chlorophyll-a peak appeared in the spring of 1983 than in the spring of 1982. The greater algal production associated with this peak in 1983 was probably a major cause of the more rapid and extensive hypolimnetic oxygen depletion which occurred that year. This depletion promoted hypolimnetic sediment phosphorus release which allowed the development of a relatively massive algal bloom in August and September.

Vertical transport events were generally associated with periods of increasing epilimnetic phosphorus and chlorophyll-a concentrations. This relationship is partially obscured by other factors such as ferric phosphate and algal phosphorus sedimentation, and algal response to influences other than phosphorus availability.

WATER QUALITY MODELING

Proposed management strategies for improving the water quality in Lake Iroquois include mechanisms for reducing both external and internal phosphorus loading. Lake water quality models can be useful in evaluating the magnitude of a lake's response to proposed management strategies. Such evaluation can provide insight into the relative benefits and cost effectiveness of various management strategies. The major drawback of using a modeling procedure to predict lake response is that most empirical models have been developed from a cross-section of various types of lakes rather than from actual observed water quality responses to management changes in individual lakes. This results in fairly large error statistics for most empirical models associated with predicting water quality parameter changes in one lake. This section attempts to use a method outlined by Reckhow and Chapra (1983) for modeling incremental changes that reduces prediction error through utilization of both observed data variability and model error.

Phosphorus Models

Reckhow (1977) developed a model to predict lake phosphorus concentrations as a function of areal phosphorus loading (L), mean depth (z), and hydraulic detention time (T_w). This model was developed from a data set of measurements from 95 lakes in northern temperate regions of North America and Europe. The parameter statistics for the lakes in this data set and for Lake Iroquois are compared in Table 39.

Figure 35 depicts predicted lake phosphorus concentrations for various levels of watershed phosphorus loading. The mean 1982-1983 watershed loading was 278 kg, with a total load of 319 kg when atmospheric deposition, groundwater and septic system loading are included (see Table 24). Using Reckhow's model, this level of loading predicts a lake phosphorus concentration of 28 ug/l, which is not far from the seven year (1977-1983)

Table 39. Parameter statistics for the data set used by Reckhow (1977) in development of the phosphorus model.

Variable (units)	Minimum	Geometric Mean	Maximum	Iroquois
z (m)	0.78	7.8	100	5.938
T_w (yr)	0.003	0.47	57.6	1.163[b]
z/T_w (m/yr)	0.53	16.5	575	5.106
$L\ T_w/z$ (mg/l)	0.0075	0.066	1.179	0.063[c]
P (mg/l)	0.004	0.033	0.958	0.031[d]
R_p	-0.08	0.38[a]	0.95	0.70[e]

[a]Arithmetic mean.

[b]Based on 1982 runoff rates plus 10% which probably most closely approximates the long term average.

[c]Based on mean 1982-1983 phosphorus loading.

[d]Seven year (1977-1983) mean spring phosphorus concentration.

[e]1982-1983 mean as measured by outflow concentrations.

where: P = lake phosphorus concentration (mg/l)

L = annual areal phosphorus loading ($mg/m^2/yr$)

z = mean depth (m)

T_w = hydraulic detention time (yrs)

R_p = phosphorus retention coefficient

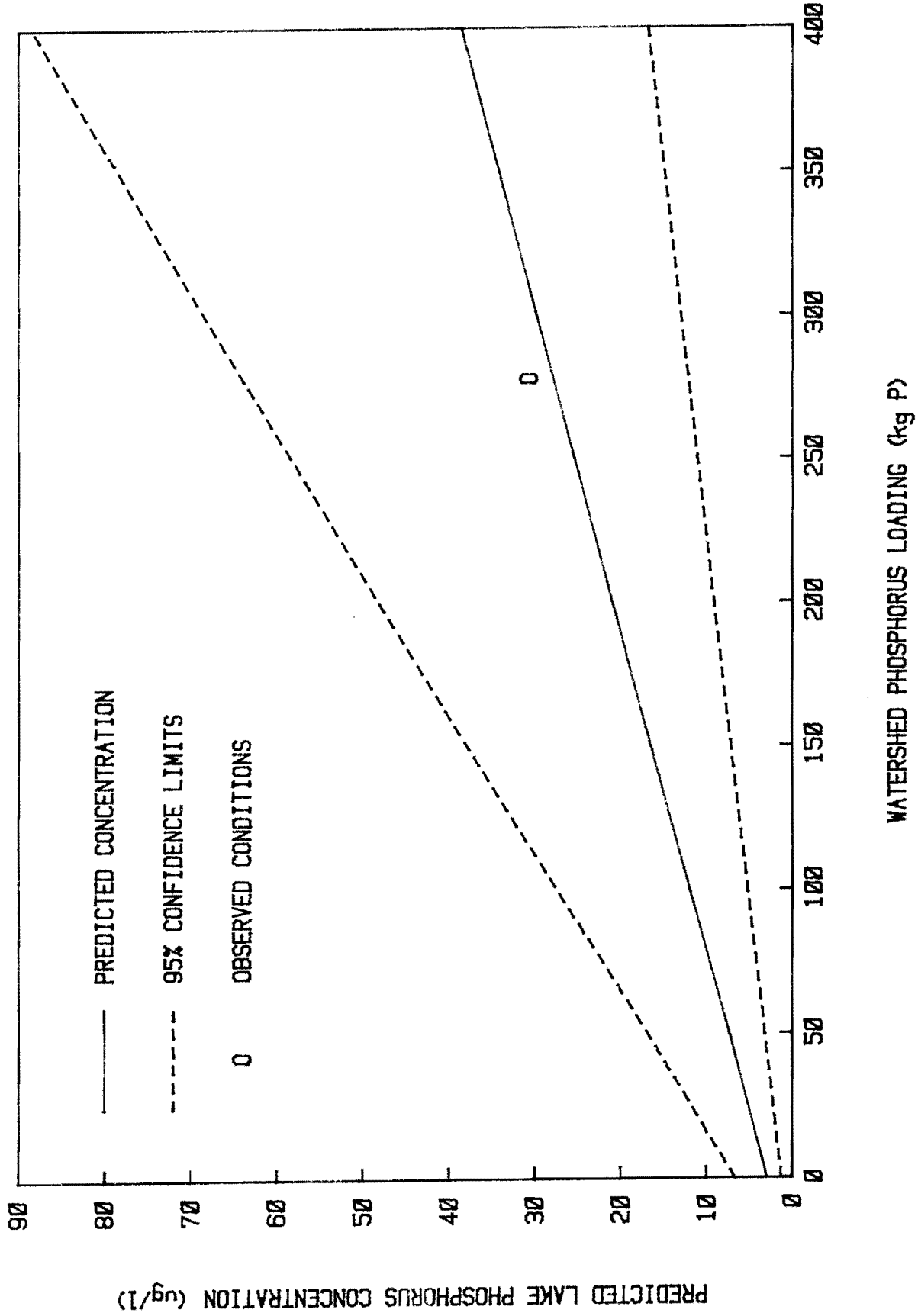


Figure 35. Predicted lake phosphorus concentration for various levels of watershed loading, based on Reckhow (1977).

observed mean spring phosphorus concentration of 31 ug/l. The figure portrays a 95% confidence interval on this prediction which is based on the model standard error of the estimate (.184, based on a logarithmic transformation of terms) and ranges from 17 ug/l to 88 ug/l for current loading conditions. Such a wide confidence range might seem to render the model inapplicable. However, these confidence limits probably greatly overestimate the true prediction error since they reflect the variability in a cross-section of lakes rather than the variability associated with changes occurring in one lake.

Reckhow and Chapra (1983) have attempted to address this problem of applying more realistic prediction intervals to empirical models through utilizing a combination of variability in observed lake conditions and model error estimates. Briefly, the procedure entails using available lake data to assess the prediction interval if the loading regime is to remain unchanged. The model error estimate is then used to place a prediction error interval about any increments in loading changes rather than about future total loading regimes. Thus, the prediction error for a future loading regime is calculated as:

$$(\text{prediction error})^2 = \pm(\text{standard error of observed lake data})^2 + (\text{model standard error for the increment of loading change})^2$$

A future phosphorus concentration is then calculated as:

$$P_{future} = (\text{observed current phosphorus concentration} + \text{predicted net phosphorus concentration change}) \pm \text{prediction error.}$$

Table 40 contains predicted lake phosphorus concentrations for incremental reductions in watershed loading using the Reckhow (1977) model, and 95% confidence limits for these predictions using the prediction error reducing method described above. The predictions made for pristine (pre-development) conditions should represent the lowest possible concentrations

Table 40. Predicted lake phosphorus concentrations for different levels of watershed loading, based on the Reckhow (1977) model and the prediction error reduction method of Reckhow and Chapra (1983).

% Watershed Loading Reduction	Watershed Loading Reduction (kg)	Watershed Loading (kg)	Predicted Lake [P] (ug/l)	Predicted Lake [P] Change (ug/l)	Corrected Future Lake P[b] (ug/l)	95% Prediction Error (ug/l)
0%	0.0	278.0	27.8	0.0	30.7	5.6[c]
10%	27.8	250.2	25.4	-2.4	28.3	6.0
20%	55.6	222.4	22.9	-4.9	25.8	7.2
30%	83.4	194.6	20.5	-7.4	23.3	8.8
40%	111.2	166.8	18.0	-9.8	20.9	10.7
Pristine conditions[a] (70% reduction)	194.1	83.9	8.8	-19.0	11.7	18.5

[a]calculated using a phosphorus export rate of 0.10 kg P/ha/yr, an atmospheric load of 15 kg and a septic system load of 0 kg

[b]observed lake P (30.7) + predicted lake P change

[c] 95% limits of observed lake data

for Lake Iroquois. The watershed loading portrayed for pristine conditions was based on a phosphorus export rate of 0.10 kg P/ha/yr. This export rate was slightly lower than the average export rate (0.11 kg P/ha/yr) from the four sub-basins in the Lake Iroquois watershed with the lowest export rates. Since these four sub-basins have experienced some degree of disturbance, a slightly lower export rate seemed justifiable. In comparison, data from 116 watersheds in the Northeast revealed a phosphorus export rate for undeveloped areas of 0.09 kg P/ha/yr (Meta Systems, 1978). Atmospheric loading was also reduced to 15 kg/yr, and septic system loading to 0 kg/yr to represent pristine conditions.

The predictions of lake phosphorus concentrations obtained from Reckhow's model may be conservative since they do not account for any decreases in internal loading which could result from reductions in external loading. LEAP (Lake Eutrophication Analysis Procedure), a model developed by Walker (1982) and calibrated to Vermont lakes, predicts a non-linear response in lake phosphorus levels to changes in external loading. Briefly, the model contains an iterative process whereby reductions in external loading affect lake spring phosphorus concentrations, which in turn reduce hypolimnetic oxygen depletion rates, which causes reductions in internal loading. Internal loading reductions, in turn, further reduce lake spring phosphorus concentrations producing the non-linear effect. At current loading levels, LEAP substantially over-predicts spring phosphorus in Lake Iroquois (Figure 36). The model also predicts a retention coefficient (R_p) of 0.00, as opposed to the 1982-1983 observed mean of 0.71. This seems to indicate that long-term spring phosphorus values in Lake Iroquois are much less sensitive to the effects of internal loading than other lakes of similar morphometric and external loading characteristics. As discussed earlier (see Internal Phosphorus Cycling section), Lake Iroquois has a high iron content resulting in an efficient removal mechanism for internally recycled phosphorus. The high retention coefficient also indicates that internal phosphorus releases,

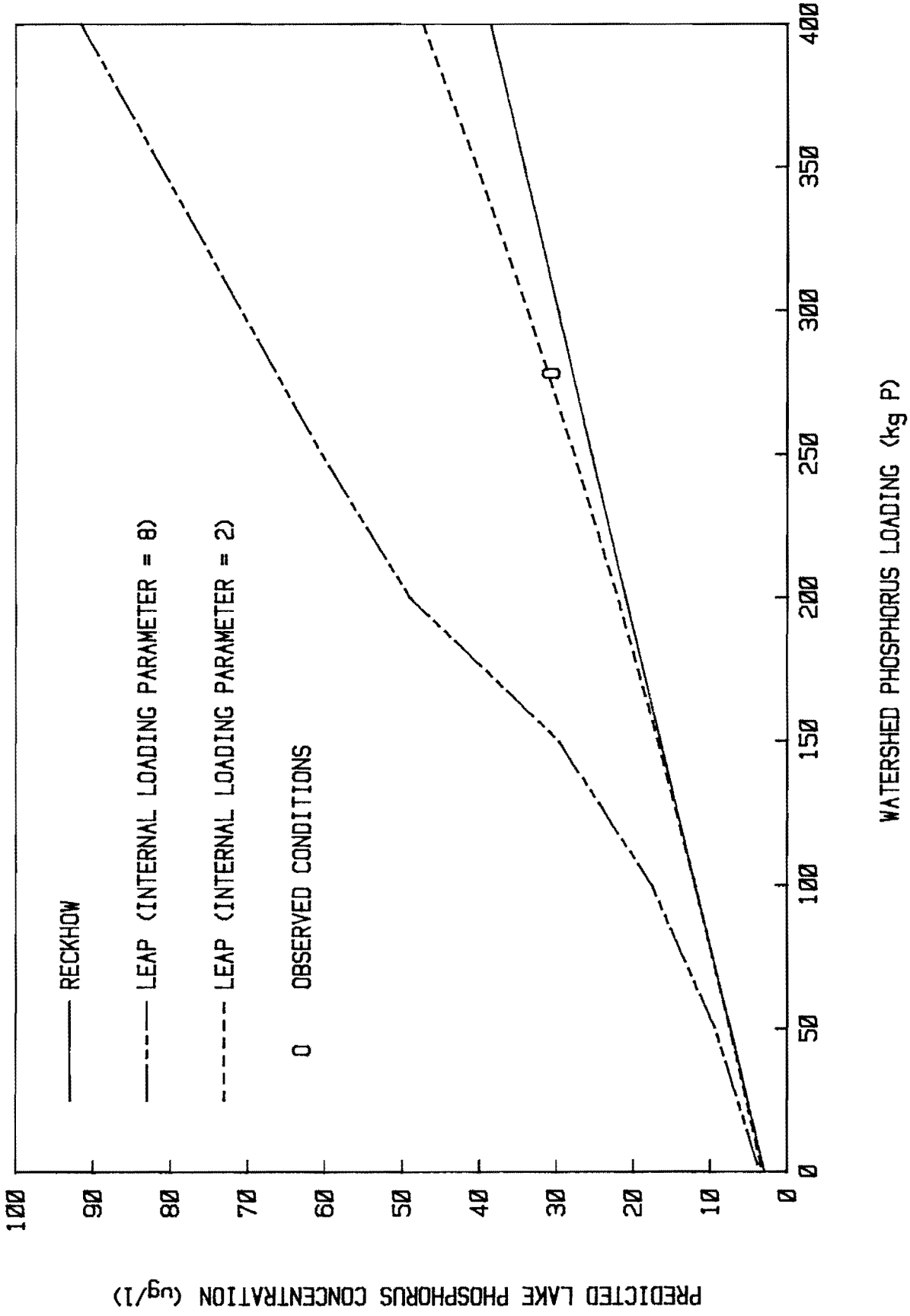


Figure 36. Predicted lake phosphorus concentrations from three phosphorus models (Reckhow, 1977; Walker, 1982).

although substantial during the summer, do not have a significant impact on subsequent spring phosphorus concentrations.

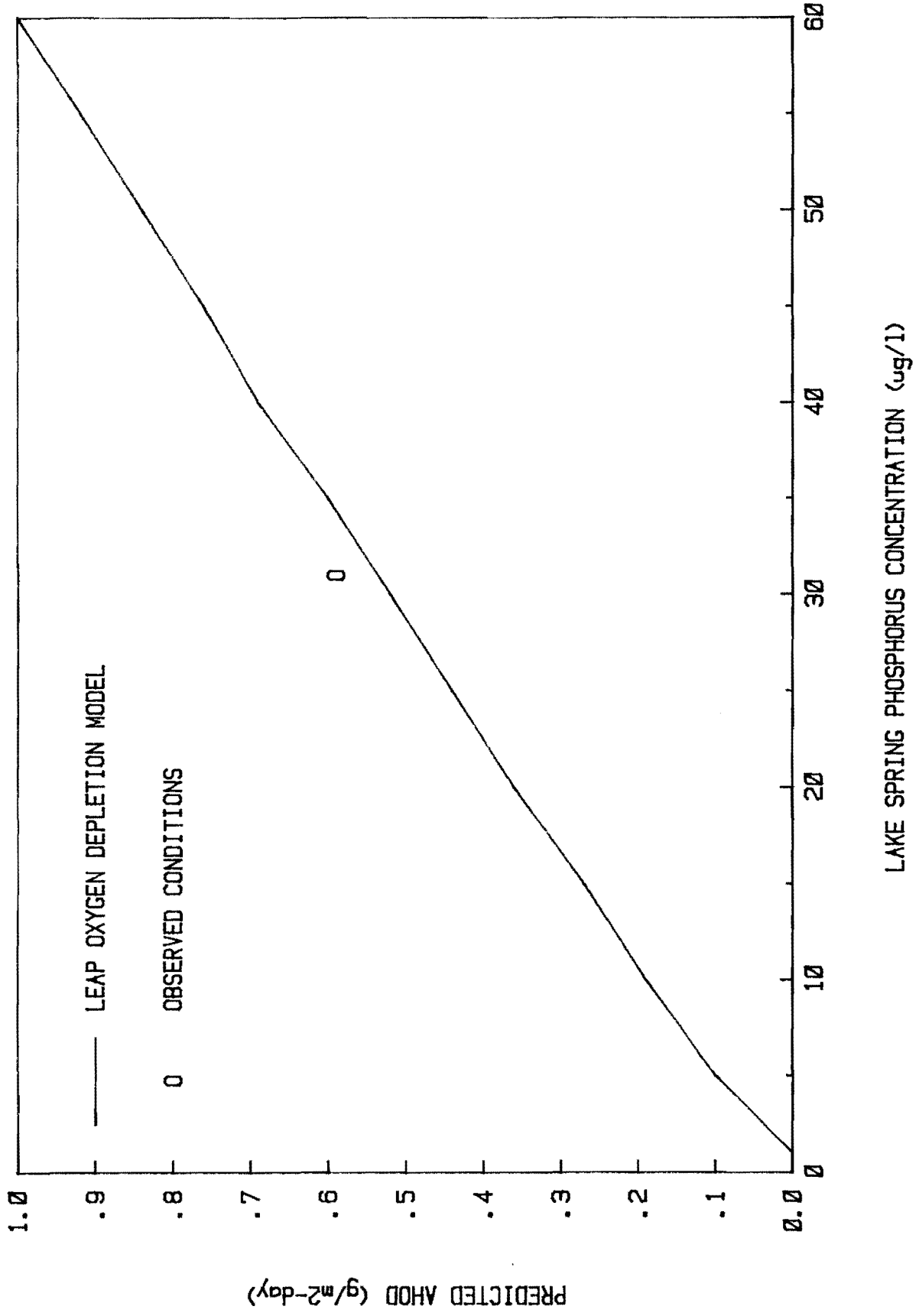
If LEAP is "calibrated" to fit observed conditions in Lake Iroquois through reducing the internal loading parameter from 8 to 2 (see Walker, 1982 for explanation), then the prediction curve becomes nearly linear, indicating a low sensitivity of spring phosphorus to internal loading. This "calibrated" version of LEAP corresponds closely to the prediction curve produced by Reckhow's model (Figure 36).

Although internal loading may not significantly affect long-term phosphorus concentrations, the seasonal impact of internal phosphorus recycling can be substantial. As discussed earlier (see Lake Response to External and Internal Phosphorus Loading section), periods of hypolimnetic phosphorus transport into the epilimnion corresponded to increases in chlorophyll-a concentrations (see Figure 34.) Theoretically, reductions in external phosphorus loading should result in reductions in the hypolimnetic oxygen depletion rate, which would, in turn, controls the magnitude of phosphorus released from sediments. The magnitude of this reduction, however, was difficult to assess. LEAP calculates internal loading as a function of external loading, oxygen depletion, and morphometry. However, as demonstrated, LEAP does not appear to predict conditions in Lake Iroquois well, due to the iterative process which magnifies the effect of internal loading.

The LEAP submodel for prediction of hypolimnetic oxygen depletion rates may be used, though, to assess the effect of changing spring phosphorus values on this parameter. The oxygen depletion submodel was originally developed on a data set of primarily summer phosphorus values from 45 lakes in the United States and Canada (Walker, 1979), and was subsequently calibrated to Vermont lakes and spring phosphorus values (Walker, 1982).

Figure 37 shows predicted oxygen depletion rates given certain levels of spring phosphorus concentrations. The predicted values shown do not correspond to previously stated Lake Iroquois oxygen depletion rates due to different methods of

Figure 37. Predicted areal hypolimnetic oxygen depletion rates for various spring phosphorus concentrations, based on Walker (1982).



calculation. If calculations are normalized, then the model slightly underpredicts AHOD conditions in Lake Iroquois. A reduction in spring phosphorus levels from current conditions of 31 ug/l to 20 ug/l could reduce the areal hypolimnetic oxygen depletion rate by 33%. At present oxygen depletion rates, the hypolimnion of Lake Iroquois reaches low oxygen concentrations (1 mg/l) conducive to sediment phosphorus release in approximately 60 days, or by early June. A reduction of 33% in depletion rates could extend this period by at least 20 days, delaying the onset of sediment phosphorus release and thereby reducing the impact of internal phosphorus recycling. Data for 1982 and 1983, which showed dramatic differences in the degree of hypolimnetic anoxia (see Figure 27) and also in the amount of phosphorus released into the epilimnion (see Table 31), demonstrate the substantial impact that reduced oxygen depletion rates could have on internal loading.

Chlorophyll Models

LEAP also predicts mean summer chlorophyll-a levels as a function of lake spring phosphorus concentrations using a relationship developed by the Vermont Department of Water Resources and Environmental Engineering (1980). The submodel was developed using a data base of observed spring phosphorus and summer chlorophyll-a values from 40 Vermont lakes (Vermont Dept. of Water Resources and Environmental Engineering, 1980). The model was used to predict Lake Iroquois chlorophyll-a concentrations at various phosphorus levels. The seven year (1977-1983) observed mean summer chlorophyll-a concentration in Lake Iroquois was 11.0 ug/l. Using the seven year mean spring phosphorus concentration of 30.7 ug/l, the model predicted a summer chlorophyll-a concentration of 12.5 ug/l. Figure 38 portrays this predictive relationship, and Table 41 gives the predicted future mean summer chlorophyll-a concentrations for the same scenarios used for spring phosphorus prediction. These predictions do not reflect the effect any internal loading

Figure 38. Predicted summer chlorophyll-a concentrations for various levels of spring phosphorus concentrations, based on Walker (1982).

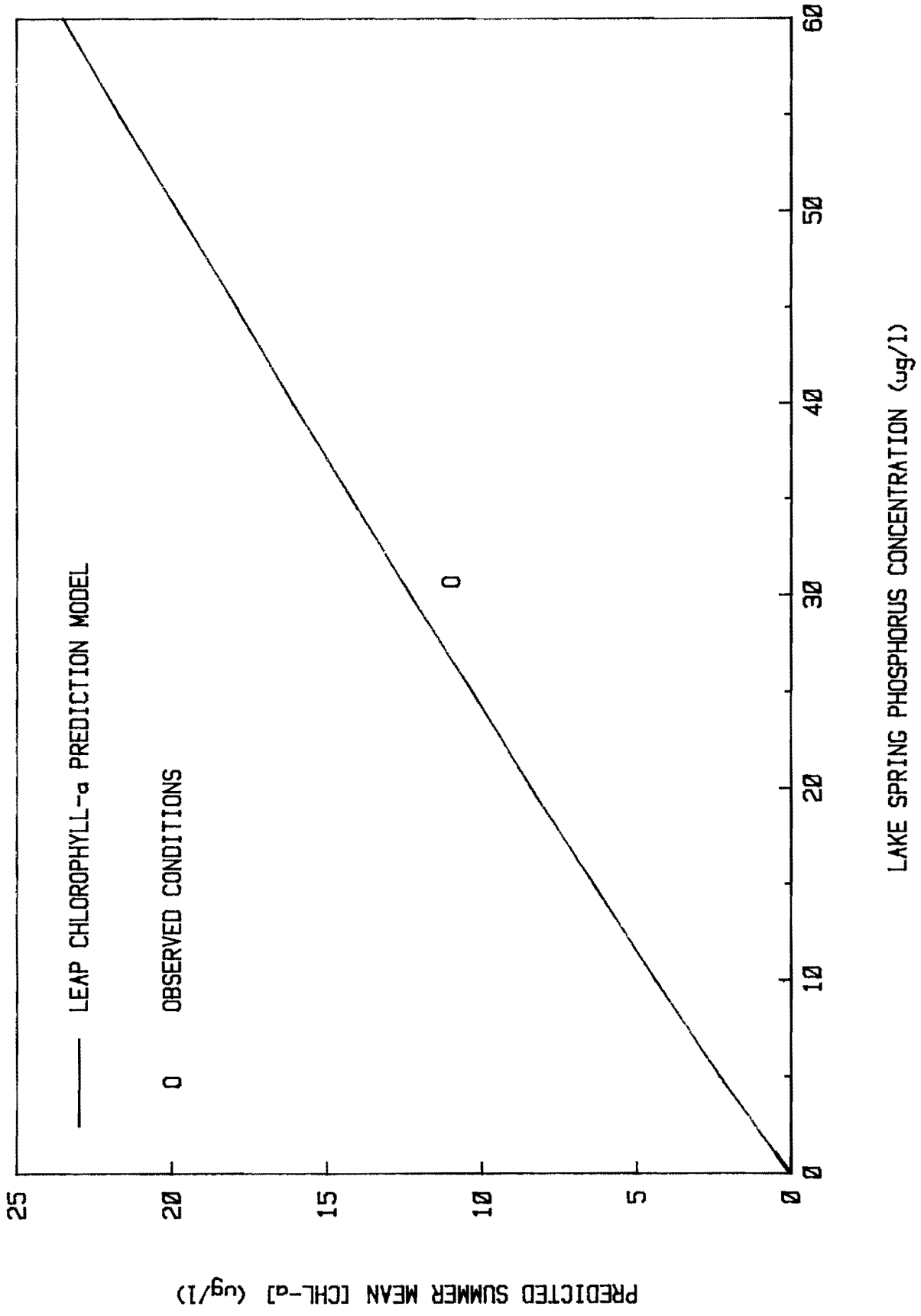


Table 41. Predicted mean summer chlorophyll-a concentrations for different levels of watershed loading.

% Watershed Loading Reduction	Watershed Loading (kg)	Future Lake P (ug/l)	Predicted Mean chl-a (ug/l)	Predicted Mean chl-a Change (ug/l)	Corrected Future Mean chl-a (ug/l)[a]	95% Prediction[b] Error (ug/l)
0%	278.0	30.7	12.5	0.0	11.0	3.4[c]
10%	250.2	28.3	11.6	-0.9	10.1	3.4
20%	222.4	25.8	10.6	-1.9	9.1	3.7
30%	194.6	23.3	9.7	-2.8	8.2	4.0
40%	166.8	20.9	8.7	-3.8	7.2	4.4
Pristine (70% reduction)	83.9	11.7	5.1	-7.4	3.6	6.6

[a]observed mean chl-a (11.0) + predicted chl-a change

[b]prediction error for the spring P/chlorophyll model only and does not include error propagation involved in prediction of spring P

[c]95% limits for observed lake data

control measures such as alum treatments or hypolimnetic aeration would have in reducing vertical phosphorus transport. Therefore, the chlorophyll predictions should be viewed as conservative in the event that internal control treatments are applied.

Transparency Models

The submodel developed and calibrated to Vermont lakes used in LEAP predicts water transparency (measured as Secchi disc depth) as a function of mean summer chlorophyll-a concentrations. The LEAP model predicts a mean summer Secchi disc measurement of 2.8 meters versus a seven year (1977-1983) observed mean of 3.4 meters. While the predicted value is not extremely close, it lies within the 95% confidence limits of the observed data (Table 42). This seven year mean also includes the values from 1982 and 1983, which were years of much higher transparency than previous years. The model was developed using a pre-1982 data base from 18 Vermont lakes. The pre-1982 observed mean Secchi disc value (1977-1983) was 2.7 meters, which was more closely approximated by the model. Whether 1982 and 1983 represent an actual shift in the chlorophyll/Secchi relationship, or whether they merely reflect year to year variability is not clear. Figure 39 plots chlorophyll-a versus 1/Secchi for the years of 1979-1983. Although there is considerable scatter to the points, it is interesting to note that the years of 1979-1981 generally lie above the regression line, while the years of 1982 and 1983 lie exclusively below the regression line. Again, this would seem to indicate a shift in the chlorophyll/Secchi relationship, although this cannot be confirmed with two years of data.

Since it is possible that Lake Iroquois may be experiencing a shift in chlorophyll/Secchi relationships, the transparency model from LEAP (also depicted in Figure 39) was used to predict future transparency conditions instead of a model developed using Lake Iroquois specific data. LEAP was calibrated and tested with data from lakes representing a wide variety of trophic status, and thus possesses a greater degree of generality. Table 42

Table 42. Predicted Secchi disc measurements for different levels of watershed loadings.

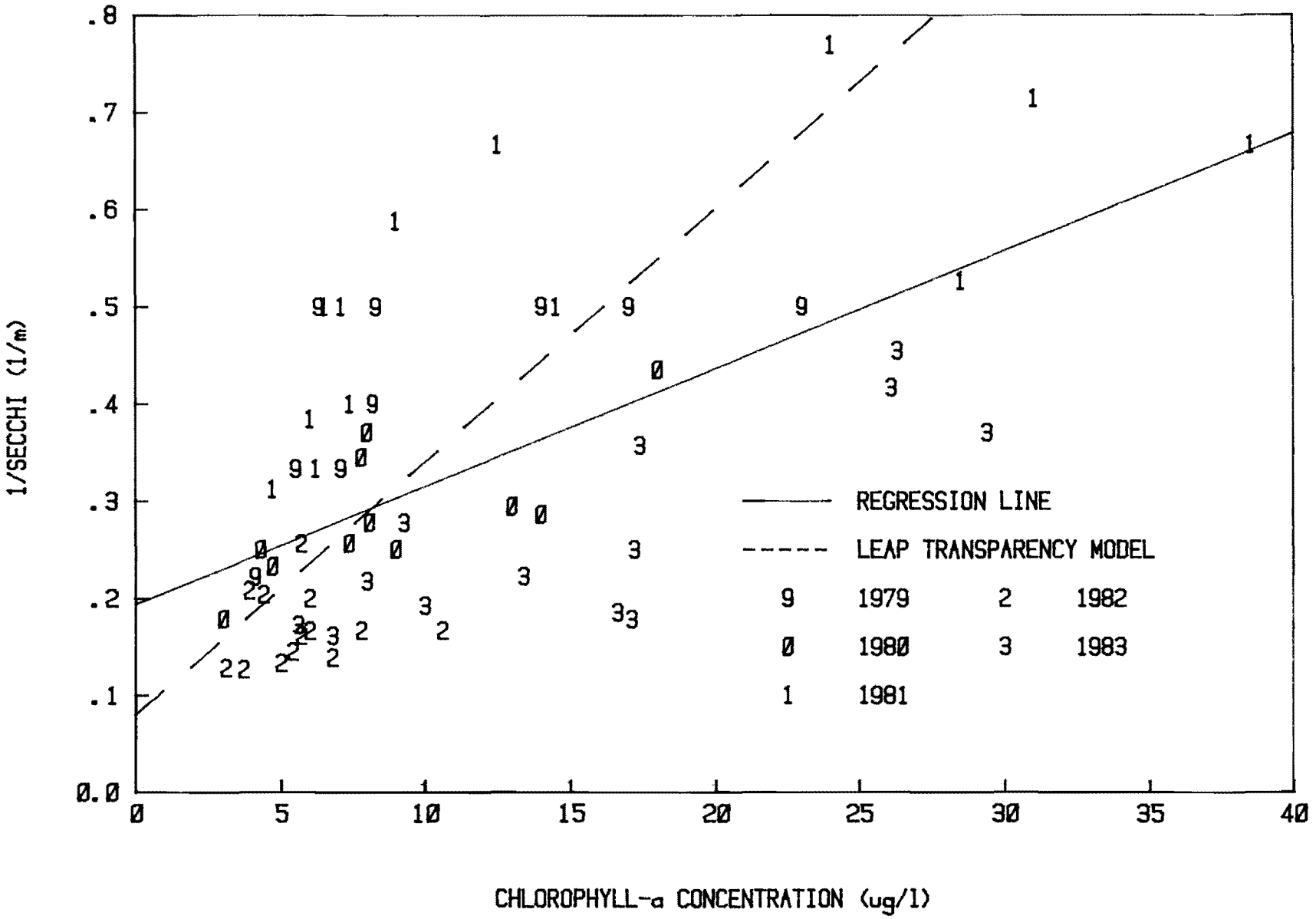
% Watershed Loading Reduction	Watershed Loading (kg)	Future Mean [chl-a] (ug/l)	Predicted Secchi (m)	Predicted Secchi Change (m)	Corrected Future Secchi[a] (m)	95% Confidence Limits[b] (m)
0%	278.0	11.0	2.8	0	3.4	1.33[c]
10%	250.2	10.1	3.0	+0.2	3.6	1.33
20%	222.4	9.1	3.3	+0.5	3.9	1.34
30%	194.6	8.2	3.5	+0.7	4.1	1.35
40%	166.8	7.2	3.8	+1.0	4.4	1.37
Pristine 70% reduction	83.9	3.6	5.9	+3.1	6.5	1.66

[a]observed mean Secchi plus predicted Secchi change

[b]prediction error for the chlorophyll/Secchi model only and does not include error propagation involved in prediction of chlorophyll

[c]95% limits for observed lake data

Figure 39. Chlorophyll-a and Secchi disc relationship for Lake Iroquois during 1979-1982.



lists predicted transparency values under the different scenarios used for other parameters. Again these predictions should be viewed as conservative in the event of application of internal loading control measures.

HISTORICAL AND PALEOLIMNOLOGICAL STUDIES

Historical water quality data can be extremely useful in any diagnostic lake study both in documenting changes in water quality parameters and in correlating these changes with man's activities in the watershed. Such a historical perspective can enhance the ability to make management decisions by more clearly revealing the mechanisms involved in a lake's water quality degradation (Smeltzer and Swain, 1984). Regular water quality monitoring has been conducted on Lake Iroquois only since 1978, and even sporadic sampling extends back only as far as 1940. In order to obtain a clearer picture of the lake's water quality history, both before any anthropogenic influences occurred in the watershed and in more recent times, a core was taken of the profundal sediments of Lake Iroquois and analyzed for various parameters to provide an indication of past water quality conditions. In addition, historical information was collected on the settlement and development of the shoreline and watershed in order to relate any changes of chemical parameters in the sediments to specific activities which could have influenced such changes. The methods used in core collection and subsequent dating and chemical analyses are described in the Methods section earlier in this report. Results of these analyses are included in Appendix J.

Watershed History

Town and county historical reports discuss the major land use trends in the Lake Iroquois area. However, specific land clearing, farming and residential developments which occurred in the watershed are not well documented. Thus, the watershed history, being gleaned largely from knowledge of region-wide land use trends, is fairly general and imprecise, but nevertheless provides some idea of the activities which occurred during various time periods. Some more specific information has also been obtained from interviews with local residents, and has been used to augment the general land use history picture.

The two towns which comprise the major portion of the Lake Iroquois watershed, Williston and Hinesburg, were both first settled in the late 1700's. The Town charters were granted in 1763 and 1762, respectively, and carried with them a condition that "every grantee, his heirs or assigns, shall plant and cultivate five acres of land within the term of five years for every fifty acres contained in his or their share or proportion of land in said Town and continue to improve and settle the same by additional cultivations on penalty of forfeiture" (Wright et al., n.d.). Thus, it is likely that land clearing began to occur in the watershed as early as the latter part of the 18th century. The extent and location of farms in the watershed in the early 1800's are not known specifically, but by the 1820's sheep farming in the area had become quite widespread (Moody and Putnam, 1961), and thus it is possible that the Lake Iroquois watershed had been substantially cleared by this time. In the latter half of the 19th century, sheep gradually gave way to more intensive dairy farming, but it is not known specifically how this affected farming in the watershed. What is currently one of the major dairy farms in the watershed was definitely in operation at least by the 1870's (Isham, personal communication). One prominent farmer in the area whose farm may have been partially located in the area had 300 cows by 1870 (Moody and Putnam, 1961). Child's Gazetteer (Child, 1882) lists a number of watershed farmers, some with as many as 35 cows.

Farming most likely continued to be a dominant land use at least in the northern and western sections of the watershed in the 1900's. One farmer stated that as many as 14 farming operations existed along the west shore which currently is in residential use (Isham, personal communication). The conversion of the western part of the watershed from farming to residential land use has been occurring since the 1940's.

The first known residential development of the lake's shoreline began in the late 1800's, with many of the present camps on the south and east shores being built in the 1890's. From then it appears that shoreline development has occurred

gradually and more or less continuously to the present (Stolper, personal communication). The one other watershed activity which bears mentioning is the logging on the east and southeast shore between 1900 and 1920 (Stolper, personal communication).

The dam which maintains Lake Iroquois at its current water level was built in the early 1900's (Anderson, 1969). The dam raised the lake about seven feet above its natural water level (Slaney, 1979). The historical record mentions the existence of a small dam on the outlet of Lake Iroquois, originally constructed around 1816, to hold back water for a gristmill (Rann, 1886). However no information exists as to the actual size of this dam, and thus its impact on the lake's morphometry. This dam may or may not have still been in existence at the time the present dam was constructed.

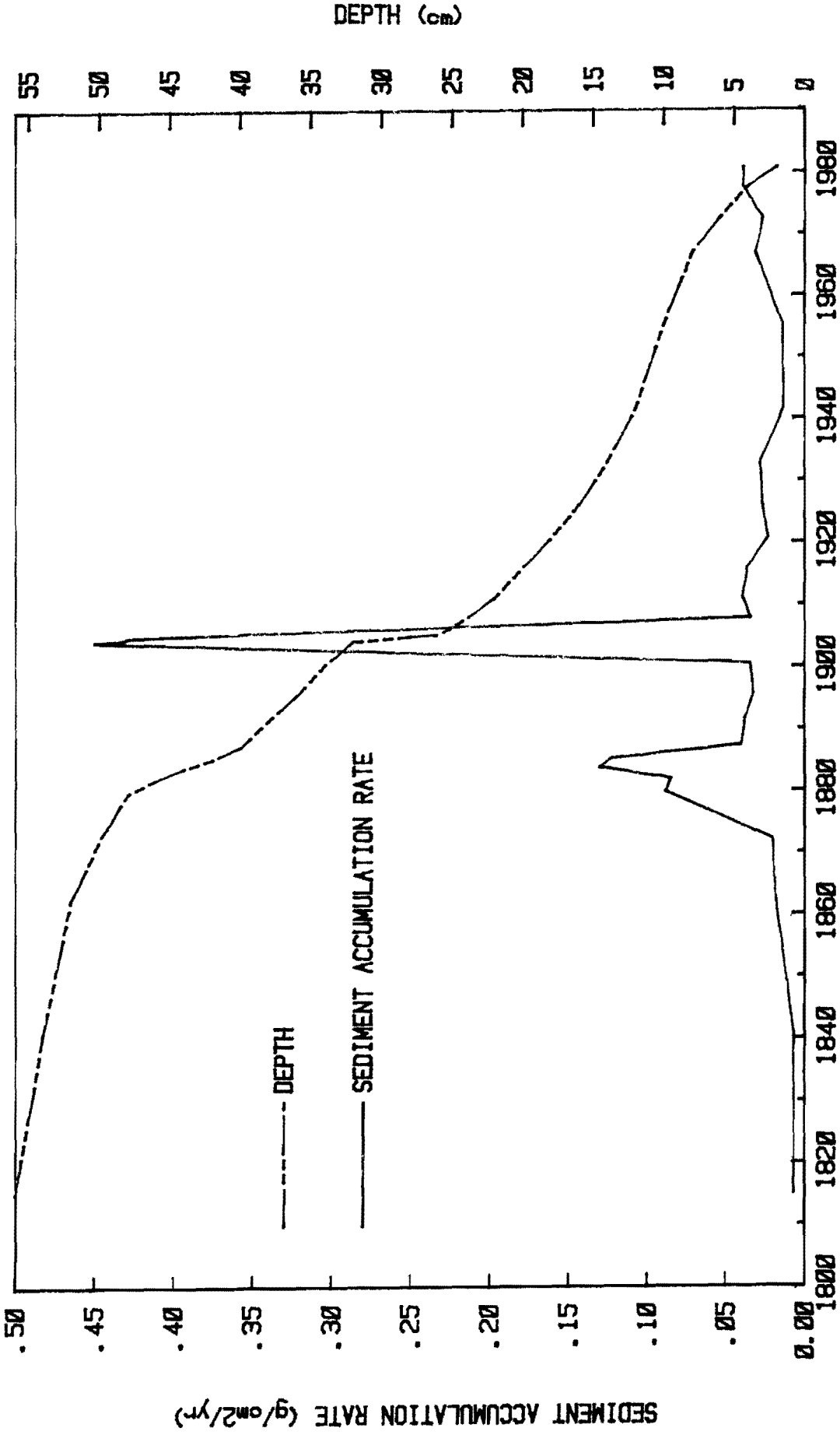
Paleolimnological Study

Sediment Accumulation Rate

Figure 40 portrays depth in the core and sediment accumulation rates between 1815 and 1983. The initial accumulation rate portrayed in the figure, although low (0.007 grams dry weight/cm^2/year), does not necessarily represent precultural, steady-state conditions, since a substantial amount of land clearing may already have occurred prior to 1815. Although the lead-210 dating could not be extended past this point, if the sediment accumulation rate between 1815 and 1841 is extrapolated back, then depths below 60 centimeters in the core would represent precultural conditions.

After 1841 a rise in the sediment accumulation rate occurred, with a sharp peak between 1872 and 1887. There is no known event in the watershed history which would account for such a dramatic peak in sedimentation, although the presence of sawmills in Hinesburg indicates that logging activity and associated erosion could have been occurring at this time. The peak occurring between 1901 and 1908 can be accounted for by the

Figure 40. Depth-in-core and sedimentation rates plotted vs. time for the Lake Iroquois core.



construction of the dam and resedimentation of flooded terrestrial soils. The dam could have elevated the water level by as much as seven feet and increased the lake's surface area by as much as 65 acres. That the dam is the probable cause of this sedimentation peak is further substantiated by the existence of a layer of older sediments (25-28 cm) interrupting two layers of younger sediments (21-24 cm and 29-32 cm). Brugam (1983) stated that this interrupting layer had only supported lead-210 without added unsupported lead-210, and, therefore, the source of the material must have either been watershed subsoils that were washed into the lake, or redistributed lake sediment that was older than 120 years. Wave action occurring on recently submerged substrates could have caused the sedimentation of the older soils to the core site.

The morphometry change resulting from the dam construction could have had a substantial effect on the pattern of sediment deposition throughout the lake and on the intensity of sediment focusing to the core site. The increased lake surface area created several shallow bays at the mouths of tributaries and could have caused sediment to settle out sooner, thereby reducing sedimentation rates at the core site. Alternatively, however, the increased lake volume could have reduced the hydraulic detention time to an extent that sedimentation rates were increased and more sediment accumulation occurred at the core site. Since it was difficult to predict the effect changing morphometry had on sediment accumulation rates in the core, it was difficult to interpret differences in sediment accumulation rates before and after the dam construction. Sediment accumulation rates through the 20th century have remained two to five times higher than the initial rate occurring between 1815 and 1841, yet this comparison is complicated by the possible changing pattern of sediment deposition. Since the accumulation rates of other sediment parameters were based on the sediment accumulation rate, this posed a problem in discussing the changes in these other parameters as well.

Sediment accumulation rates have risen in recent decades above those occurring in the 1940's and could be reflecting the increased erosion of mineral soil as the west shore becomes more intensively developed as a residential area.

Sediment Organic Content

The percent organic content of sediments can be an indicator of a lake's productivity, with higher productivity being evidenced by a higher percent organic content in the sedimenting materials. However, if periods of increasing productivity in a lake's history are accompanied by increased erosion of mineral soil, the evidence of increasing organic content may be obscured in the core by simultaneously increasing inorganic content. A better picture of a lake's changing productivity may be seen through organic matter accumulation rates. However, as stated earlier, comparison of accumulation rates in the Lake Iroquois core outside of the 20th century was difficult.

Figure 41 shows that percent organic content in the Lake Iroquois core never rose significantly above levels found at the bottom of the core. Percent organic content remained extremely constant at about 38% from the bottom of the core (82 cm) up through 1890 (38 cm). At this point, percent organic content steadily decreased until the 1930's. The increased inorganic content in the sediments could be the result of increased shoreline development during this period which has been sustained in intensity through the present. Since the 1930's, the percent organic content has remained very constant, but organic accumulation rates (Figure 41) have shown a slight rise beginning in the 1950's. It is difficult to say whether this reflects a change in productivity, or merely increased erosion rates.

Iron and Manganese

Several mechanisms can affect iron and manganese concentrations in the sediments. Changes in sediment loading to the lake, changing redox conditions at the core site, and

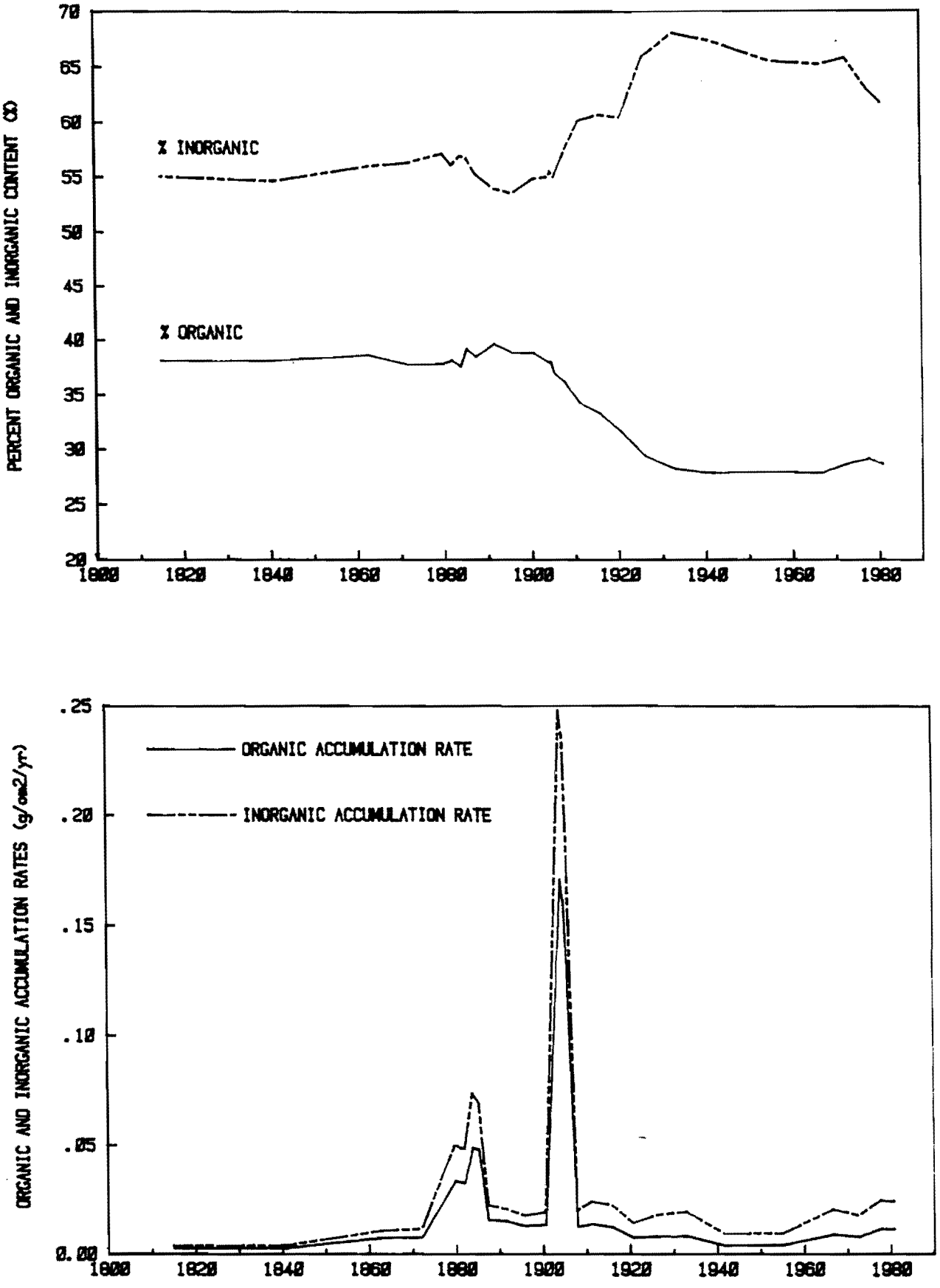
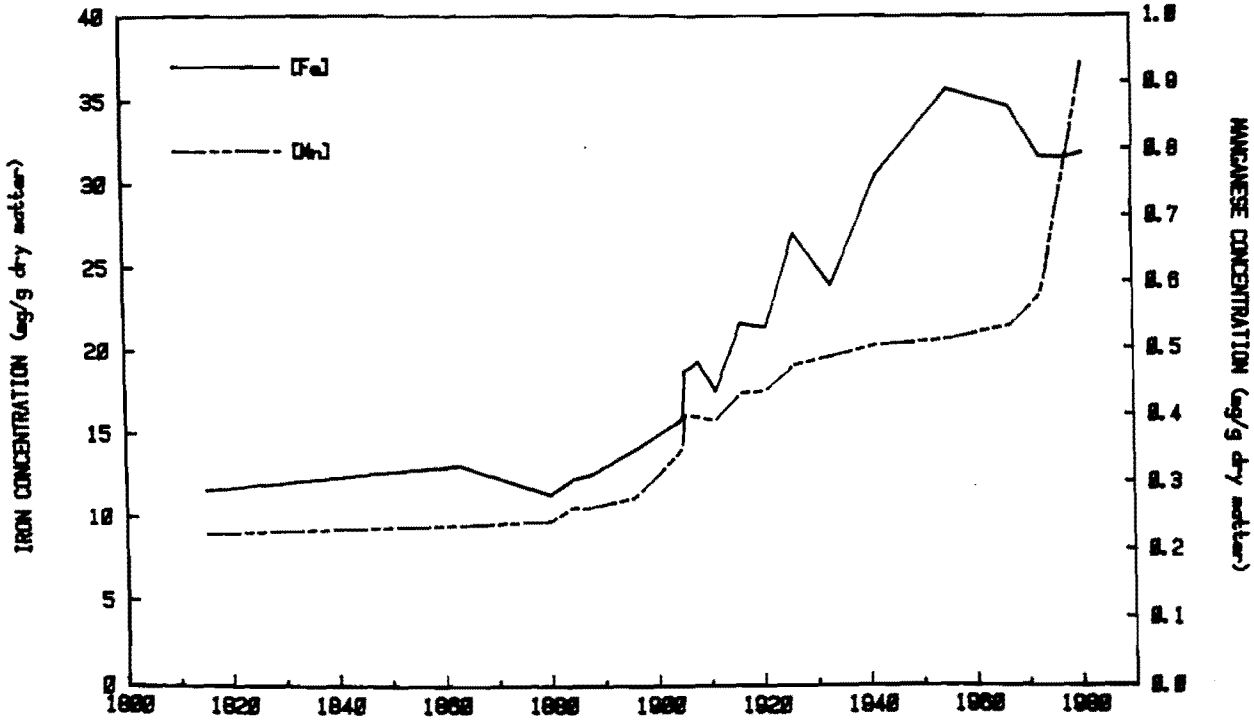


Figure 41. Sediment organic and inorganic composition and accumulation rates.

migration of ions along concentration gradients within the core can all affect the concentrations found at any given level within the core. The effect of depleted oxygen levels on iron and manganese concentrations can also vary. Mackereth (1966) associated low oxygen levels with declines in iron and manganese concentrations through the mechanism of iron and manganese mobilization from anoxic sediments into the water and subsequent loss from the lake. Alternatively, Engstrom et al. (in press) have associated low oxygen conditions with increases in manganese concentrations through the mechanism of dissolution from anoxic sediments and enhanced redeposition in the profundal regions of the lake. The extent of hypolimnetic oxygen depletion may be a critical factor in determining the fate of mobilized ions. Mild oxygen depletion, as opposed to complete hypolimnetic anoxia, may result in a net gain of manganese and iron in the profundal sediments (Engstrom et al. in press)

The Lake Iroquois core showed an increase in both iron and manganese concentrations beginning in the 1880's (Figure 42). If this increase reflected the onset of partial hypolimnetic oxygen depletion, one would expect to see a selective enrichment of manganese relative to iron, since this ion dissolves at higher redox potentials and would therefore be more readily mobilized. This was not the case, however, in the Lake Iroquois core, and iron appeared to be selectively enriched relative to manganese (Figure 42). The dramatic rise in manganese concentration at the top of the core (Figure 42) does not necessarily reflect an increase in manganese deposition, but may be the result of upward diffusion along a concentration gradient since interstitial fluids below the oxidized microzone are enriched in iron and manganese relative to the surface muds. Such ion migration within the core can result in iron and manganese peaks at the top of the sediment profile (Engstrom and Wright, 1983). If this was the case, however, it is not clear why the iron profile did not show a similar peak. Since a number of mechanisms were possibly affecting the iron and manganese concentration profiles, it is difficult to draw any conclusions, particularly concerning bottom oxygen conditions, with this evidence alone.



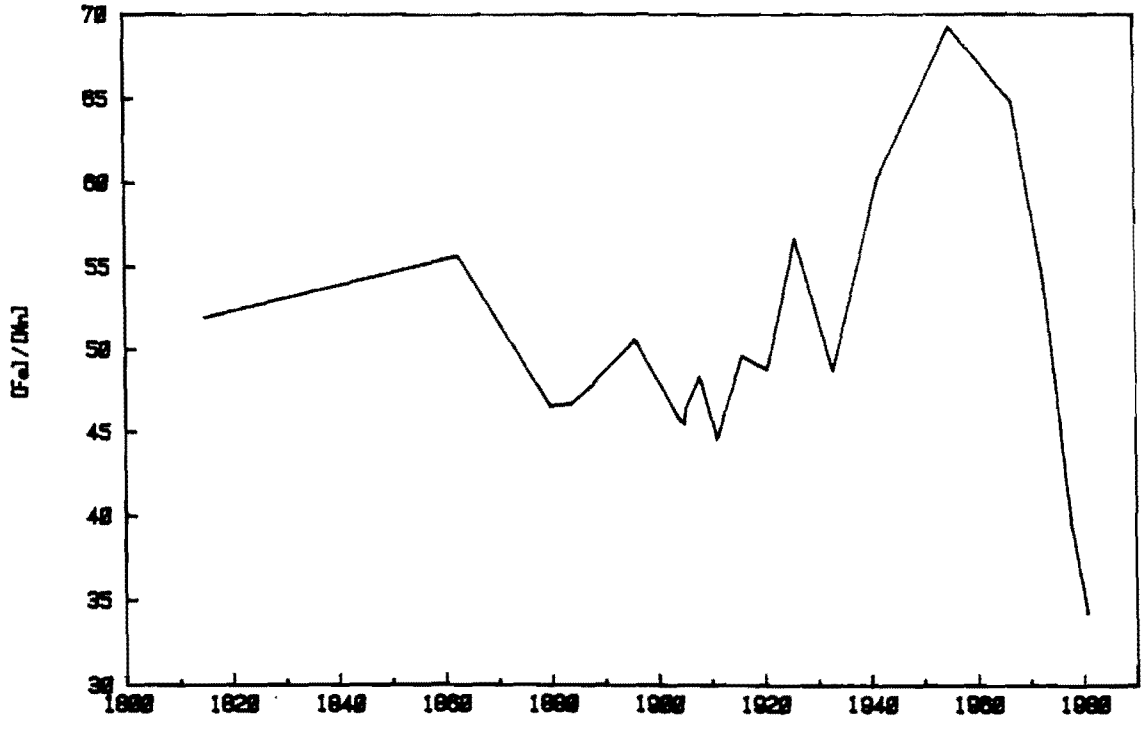


Figure 42. Sediment iron and manganese concentrations and ratios.

Pigments

Concentration profiles of chlorophyll derivatives (CD) and total carotenoids (TC) and their corresponding accumulation rates and ratios are presented in Figure 43. The concentrations of these two groups of pigments and the ratios between them can reflect qualitative and quantitative changes in a lake's productivity, changes in hypolimnetic oxygen conditions, and periods of high organic inputs from the watershed. Since not only these three mechanisms, but changing organic accumulation rates can affect the pigment concentration profiles, they must be interpreted cautiously.

The qualitative and quantitative nature of productivity can affect pigment concentrations (expressed as absorbance units per gram of organic matter). The effect of self-shading in a high standing crop of algae is to cause more chlorophyll to be produced per gram organic matter than in a less productive situation where self-shading is less intense (Steele, 1962; Meeks, 1974). Lakes which are dominated by blue-green algae will tend to produce disproportionately more carotenoids, relative to chlorophyll derivatives, affecting the ratio between the two pigments (Harris, 1978).

The oxygen regime in the bottom of a lake can influence the sediment pigment concentrations by controlling the degree of pigment degradation which takes place relative to the organic matrix. More productive lakes tend to have lower hypolimnetic oxygen concentrations, a situation which enhances the preservation of algal pigments. Analyses of sediment pigment concentrations in dimictic eutrophic lakes (Swain, 1985) revealed an increase in concentrations of both chlorophyll derivatives and total carotenoids along transects from the well oxygenated shallow water sediments to the poorly oxygenated deep water sediments. The same study also showed higher CD/TC ratios in the shallow water oxygenated sediments and lower ratios in the deep water poorly oxygenated sediments, suggesting that under oxidizing conditions, carotenoids degrade faster than chlorophyll derivatives. Gorham and Sanger (1967) also found evidence that

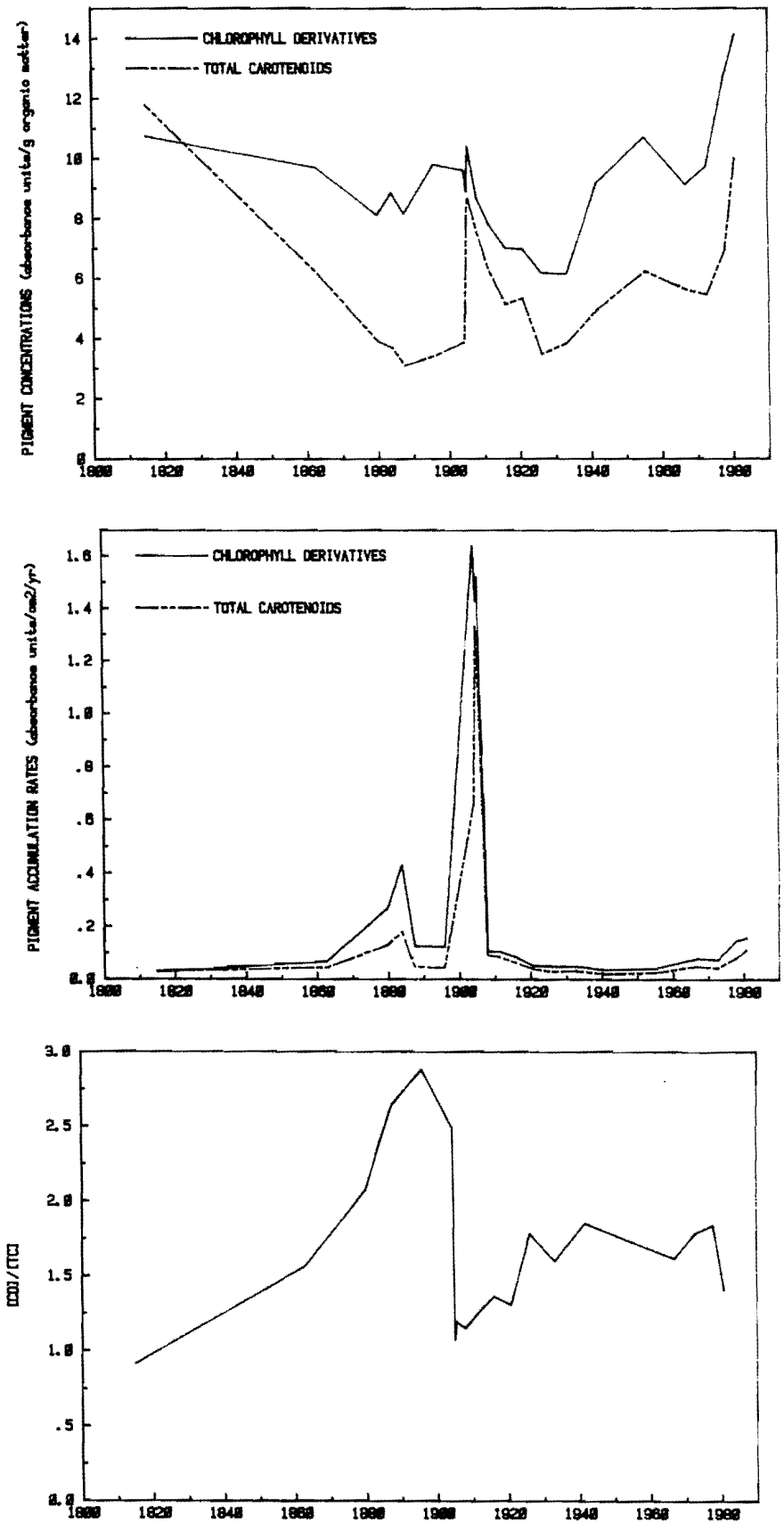


Figure 43. Sediment pigment concentrations, accumulation rates, and ratios.

corroborates this observation. Thus, low oxygen conditions, which enhance preservation of pigments, will not only tend to drive concentrations higher, but will also tend to favor carotenoid concentrations relative to chlorophyll derivative concentrations, thus lowering the CD/TC ratio.

Periods of accelerated inputs from the watershed can have differing effects on pigment concentrations in the sediments. Terrestrial organic matter tends to be low in pigment concentrations and to possess high CD/TC ratios. Thus, if allochthonous inputs are extremely high, the effect of dilution of autochthonous materials may be sufficient to lower sediment pigment concentrations and raise the CD/TC ratio. Alternatively, however, if sedimenting material from the watershed is deposited rapidly enough, the effect of enhanced preservation from rapid burial may raise sediment pigment concentrations.

The pigment concentration profiles in the Lake Iroquois core (Figure 43) do not conclusively indicate that significant changes have taken place in either productivity or preservation conditions from pre-cultural times to the present. Concentrations of both chlorophyll derivatives and total carotenoids declined through the 1800's, particularly total carotenoids, with a corresponding increase in the CD/TC ratio (Figure 43). The 1800's was a period of increasing sedimentation rates and thus, the decline in pigment concentrations could be due to dilution from allochthonous organic matter. At the time of dam construction, the CD/TC ratio dropped sharply and was accompanied by sharp peaks in pigment concentration, suggesting that the rapid burial which occurred at this time enhanced preservation conditions. Since the 1930's, concentrations of both pigments have risen dramatically above post-dam levels, suggesting a possible shift in productivity or oxygen depletion. However, whether this is actually the case is unclear since concentrations did not rise significantly above pre-cultural levels, and pigment accumulation rates (Figure 43) did not reflect such a dramatic increase. The CD/TC ratio following the period of dam construction remained higher than pre-cultural ratios, again suggesting that the twentieth century, particularly

since the 1930's, has been a period of sustained high watershed inputs.

One other parameter which can be used to indicate changing oxygen depletion conditions is percent native chlorophyll, shown in Figure 44. Percent native chlorophyll is based on absorbance measurements made before and after acidification and thus reflects the degree of pigment preservation (Swain, 1985). Beginning in the 1930's, the Lake Iroquois core showed a dramatic rise in percent native chlorophyll, far above any pre-cultural levels. This rise was associated with the increased pigment concentrations, and suggests that the latter may indeed have been responding to lower oxygen concentrations, perhaps indicating the onset of eutrophication.

Diatoms

Selected levels of the Lake Iroquois sediment core were analyzed for diatom remains by Kingston (1984). Based on an interpretation of taxonomic shifts in diatom frustules found in the sediment, Kingston concluded that the lake has experienced significant changes in trophic state during its history. Species indicative of an oligotrophic or mesotrophic condition were present in the core level corresponding to the period prior to European settlement of the region. Since about 1920, however, there has been a significant shift to diatom species indicative of eutrophic conditions.

Summary

The Lake Iroquois watershed has been experiencing cultural impacts since the late 1700's. Evidence of increased erosional processes can be found in the sediments from at least the mid 1800's. Increases in sediment accumulation rates, percent inorganic content, iron and manganese concentrations, and CD/TC ratios are all evidences of increased watershed sediment inputs. Increases in pigment concentrations and percent native chlorophyll suggest that Lake Iroquois began to exhibit signs of

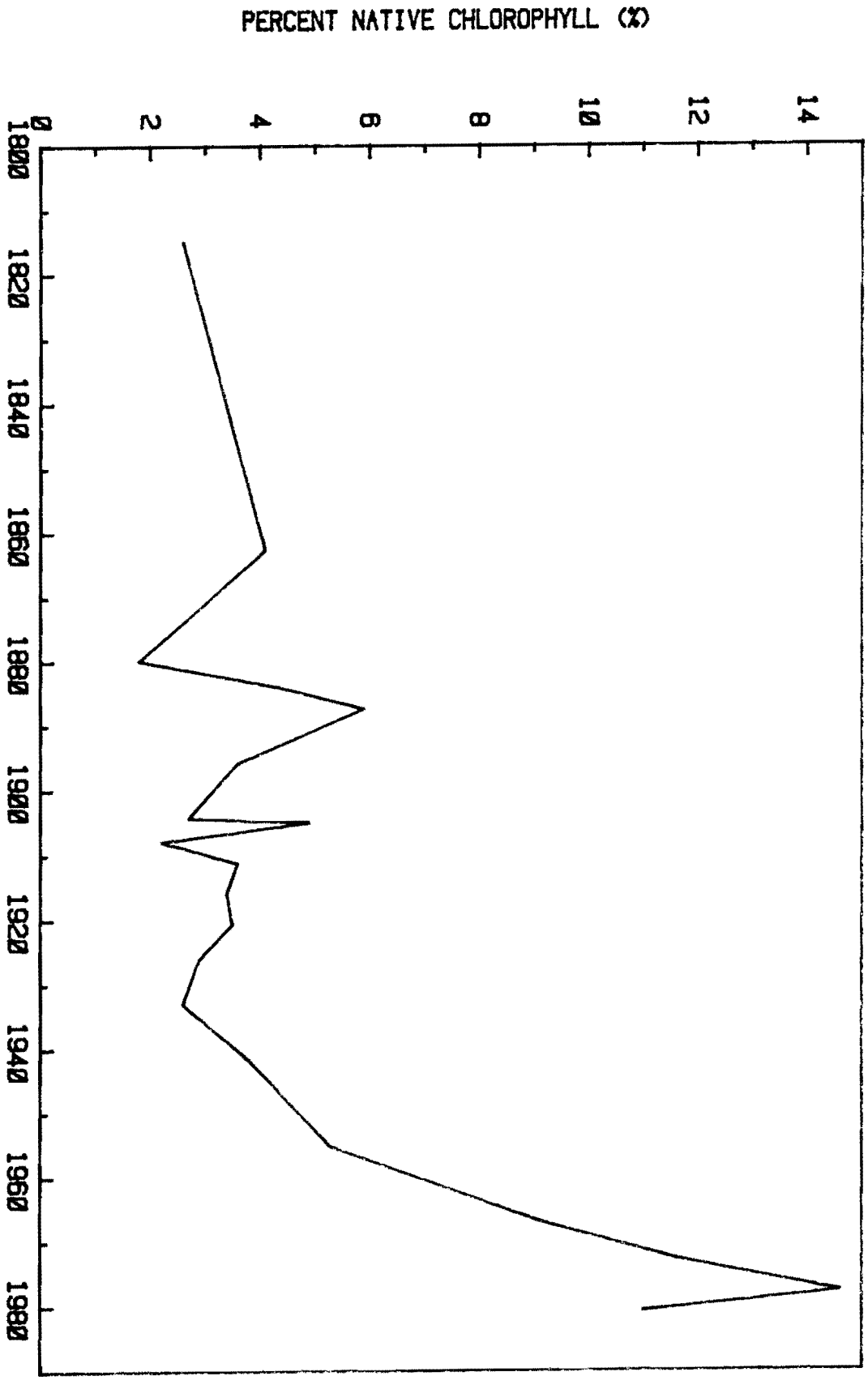


Figure 44. Sediment percent native chlorophyll.

increasing hypolimnetic oxygen depletion in the 1930's. This response could have been due to increased watershed activity, which appears to have been fairly intensive throughout the 20th century. Alternatively, it could reflect the effect of a morphometry change imposed by the dam. The rise in water level would have greatly increased the extent of the hypolimnion, thereby promoting anoxic conditions. However, this response did not occur until nearly 30 years after dam construction. Finally, the diatom analysis showed evidence of a shift to a more eutrophic condition, particularly since about 1920, probably as a result of nutrient enrichment from watershed activities such as residential development and farming.

AQUATIC MACROPHYTES

Species Composition

A list of the 29 species of aquatic macrophytes found in Lake Iroquois, in their approximate order of abundance during 1982, is presented in Table 43. The four most dominant species were *Elodea Nuttallii*, *Heteranthera dubia*, *Potamogeton praelongus*, and *Potamogeton zosteriformis*. General observations during biomass sampling indicated that *E. Nuttallii* was probably much more dominant than the weighted frequency value in Table 43 suggests.

Chara sp. was identified in the lake, but *Nitella*-like species present could not be definitely identified since no fruiting structures were found. These may be a different form or species of *Chara*. However, during the survey all *Chara* and *Nitella*-like species were grouped as *Chara/Nitella* spp. Two species of *Najas* were observed, but neither could be positively identified due to the absence of fruiting structures. This was also the case for the two *Eleocharis* species present.

Distribution and Biomass

The distribution of macrophytes in the lake is shown in Figure 45. The area of plant coverage was 37% of the total lake area, with the largest expanse being found at the lake's shallow northern end. The mean greatest depth of plant growth was 3.7 meters. Insufficient light penetration restricted the development of macrophyte communities beyond this depth.

The low density shallow macrophyte zone (Figure 45) along sections of the lake's east shore reflected the existence of a coarser substrate (sand, gravel, and rubble; Appendix H) in this area which provided a poorer medium for plant growth. The coarser substrate was due largely to the steeper slope of the lake bottom which facilitated the movement of fine materials to deeper water. Lower shore area erosion rates resulting from minimal land development were probably also a contributing factor.

Table 43. Macrophyte species list and frequency ratings.

Species	Common Name	Frequency (%)*	Weighted Frequency**
Elodea Nuttallii	Waterweed	71	101
Heteranthera dubia	Water Star Grass	61	87
Potamogeton praelongus	Whitestem Pondweed	51	74
Potamogeton zosteriformis	Flat-stemmed Pondweed	53	69
Chara/Nitella spp.	Chara/Nitella	46	61
Najas spp.	Bushy Pondweed	51	59
Vallisneria americana	Wild Celery	34	36
Myriophyllum exalbescens	Water Milfoil	42	35
Potamogeton Spirillus	A Pondweed	37	34
Potamogeton amplifolius	Large-leaf Pondweed	25	32
Potamogeton Vaseyi	A Pondweed	34	29
Potamogeton gramineus	Variable Pondweed	31	24
Potamogeton Richardsonii	Clasping-leaf Pondweed	19	18
Nymphaea odorata	Water Lily	7	8
Potamogeton crispus	A Pondweed	10	7
Ranunculus subrigidus	A Buttercup	12	7
Isoetes Braunii	A Quillwort	8	5
Eleocharis sp.	A Spike Rush	10	5
Sagittaria rigida	Stiff Wapato	7	5
Polygonum natans	Amphibious Smartweed	2	4
Ranunculus reptans***	Creeping Spearwort	0	0
Eriocaulon septangulare***	Pipewort	0	0
Nuphar variegatum***	Yellow Water Lily	0	0
Potamogeton epihydrus	Leafy Pondweed	0	0
Potamogeton americanus***	A Pondweed	0	0
Fontinalis sp.***	Water Moss	0	0
Typha latifolia***	Common Cattail	0	0
Eleocharis sp.***	A Spike Rush	0	0

*percentage of sample sites at which species was present
**summation of relative abundance values from all sites for a species
***these species were found in the lake, but were not present at any of the sampling sites

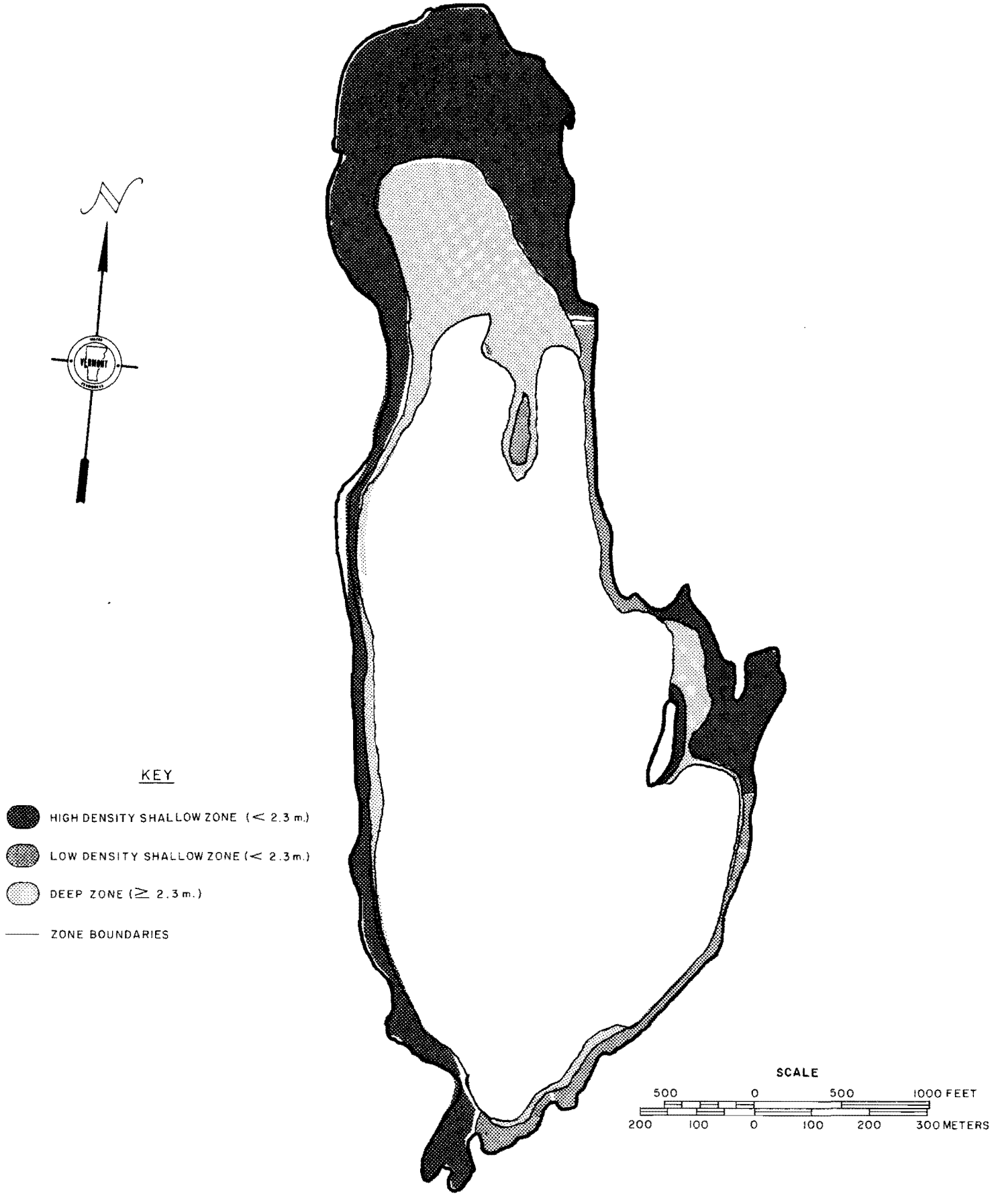


Figure 45. 1982 Lake Iroquois macrophyte distribution.

Biomass estimates for the macrophyte population are presented in Table 44. Total macrophyte biomass (dry wt.) for the lake was determined to be 24,610 kg (±11% at the 95% confidence level), with a mean density of 68.0 g dry wt/m^2. The high density shallow zone (Figure 45) had the highest mean density at 101.9 g dry wt/m^2, while the low density shallow zone had the lowest mean density at 33.8 g dry wt/m^2.

The high density shallow zone contained 87% of the total plant biomass within the lake. The combined shallow and deep zones forming the large expanse of plants at the northern end of the lake contained 68% of the lake's biomass.

Aquatic plant biomass in lakes typically increases through the spring and early summer, peaks in late summer, and declines thereafter. Data presented by Wile et al. (1979) showed that the late August/early September macrophyte biomass never differed by more than 17% from the peak annual biomass over a six-year period at Chemung Lake, Ontario. Nichols (1982) recommended mid-July to early September sampling to provide suitable estimates of maximum macrophyte biomass. The biomass estimate for Lake Iroquois was based on samples taken in late August and early September, and is believed to approximate the maximum biomass for the year.

Additional biomass sampling was conducted in the shallow zone (less than 2.3 m) at the lake's north bay during late August of 1983 to provide an indication of annual biomass variation (Appendix K). The mean biomass was 125.4 g dry wt/m^2 (n = 25) during 1983 compared to the mean of 102.7 g dry wt/m^2 (n = 80) found for the same area during 1982. The difference was significant at the 95% confidence level. Complaints of macrophyte nuisance problems by lakeshore residents also increased during 1983. General observations indicated that tall growing species approached the water surface earlier in 1983, possibly as a result of the unusually warm water temperatures during June of that year. *Elodea Nuttallii* appeared to be less abundant in the north bay during 1983 than in 1982.

The likelihood of substantial annual variations in maximum biomass for Lake Iroquois beyond those found in 1982 and 1983 is

Table 44. 1982 Macrophyte biomass estimates.

Area	Macrophyte Density g dry wt./m^2		Area of Macrophyte Coverage m^2	Mass (dry wt.) kg x 10^2	
Total lake	lower*	60.6	376,675	lower*	228.3
	mean	68.0	376,675	mean	256.1
	upper*	76.1	376,675	upper*	286.6
High density shallow zone	lower*	91.9	227,026	lower*	208.6
	mean	101.9	227,026	mean	231.3
	upper*	112.7	227,026	upper*	255.9
Low density shallow zone	lower*	21.8	32,227	lower*	7.0
	mean	33.8	32,227	mean	10.9
	upper*	48.6	32,227	upper*	15.7
Deep zone	lower*	15.4	117,422	lower*	18.1
	mean	20.4	117,422	mean	24.0
	upper*	26.5	117,422	upper*	31.1

* lower or upper 95% confidence limit

uncertain. The results of macrophyte studies at other lakes provide some additional insight. Maximum macrophyte biomass in Chemung Lake, Ontario, averaged approximately 350 g dry wt/m^2, with a maximum variation of only 15% over six consecutive years (Wile et al., 1979). At Lake Wingra, Wisconsin, maximum biomass of Myriophyllum spicatum averaged about 220 g dry wt/m^2 (ash free) with a maximum variation of 40% over five consecutive years (Carpenter, 1979). However, following a decline in the M. spicatum population (an invading exotic species), maximum biomass values as low as 70 g dry wt/m^2 (ash free) were estimated.

Comparison to Previous Surveys

A non-quantitative macrophyte survey was conducted by the Vermont Department of Water Resources at Lake Iroquois in July, 1977. This survey found a similar species composition and distribution as the 1982 survey, with some exceptions. Most notably, the 1977 survey indicated that some shoreline stretches were plant-free or nearly so. Macrophyte presence in the sections between sites 46-48, 43-45, and 31-35 (see Figure 13) was described as "none", "sparse or none", and "none found", respectively. The survey also stated that no plants were found between the island and the prominent point to its east. All these areas were found to be plant covered in 1982 and two of them had a high density of plant growth. Poor water clarity was noted during 1977, which hindered observations and might also have reduced macrophyte growth due to light limitation. One species, Heteranthera dubia, found to be common during the 1982 survey, was not noted during 1977. It is probable that H. dubia was not distinguished from Potamogeton zosteriformis, which resembles it closely and was found to be abundant in 1977. Macrophyte distribution observations indicated on a 1972 public information chart of the lake (see Figure 1) show plants were present between the island and the point to its east at the time the survey was conducted. Most local residents strongly feel that aquatic plants have increased over their period of memory, and the survey data generally supports this conclusion.

Internal Loading Potential

The average phosphorus content of bulk macrophyte tissue from five lakes is listed below.

% Phosphorus (dry wt. basis)	Lake, Location	Reference
0.28	Bomoseen, Vermont	Garrison (1979)
0.29	Memphremagog, VT-Quebec	Carignan and Kalff (1982)
0.27	Sallie, Minnesota	Peterson *et al.* (1974)
0.25	Long, Washington	Welch *et al.* (1979)
0.27	Chemung, Ontario	Wile *et al.* (1979)

Using the estimate of 25,610 kg dry wt. of plant tissue for the lake, and assuming a phosphorus content of 0.27%, the total phosphorus content of the macrophytes in Lake Iroquois was 69 kg.

Studies by Gabrielson (1978), Smith (1978), and Barko and Smart (1979) have shown that aquatic macrophytes typically derive the great majority of their phosphorus from the lake sediment. Carignan and Kalff (1980) found that 100% of the phosphorus in nine macrophyte species growing in mesotrophic and mildly eutrophic waters was derived from the sediment. The release of phosphorus to the water by plants is a mechanism for pumping sediment phosphorus into the water column. The potential for phosphorus release during active growth has been examined by a number of investigators. Welch (1979) conducted culture experiments with healthy *Elodea densa* and found no net leakage of phosphorus to surrounding waters. Peverly and Brittain (1978) and Smith (1978) showed extremely minimal releases of phosphorus by *M. spicatum* during active growth. Barko and Smart (1979) found that less than 3% of the phosphorus mobilized by *M. spicatum* was secreted.

While apparently only a small percentage of the phosphorus content of aquatic macrophytes is released during healthy growth, a very substantial release to the water column has been

documented to occur during senescence and decay (Welch, 1979; Lie, 1979; Landers, 1979; Barko and Smart, 1979). Smith (1978) estimated that 40% of the sediment phosphorus taken up by M. spicatum was subsequently lost to the water column through sloughing and decay. Nichols and Kenney (1973) demonstrated phosphorus losses ranging from 64-77% within two weeks following the initial decay of M. spicatum. Jewell (1971) determined that 24% of aquatic macrophyte phosphorus was refractory, and so returned to the sediment.

The bulk of tissue decay would be expected to occur after the biomass peak (August/September), although some highly productive species, notably M. spicatum, sustain high rates of tissue loss prior to this time. In a review of the topic, Westlake (1975) suggested that total annual production by submersed aquatic macrophytes is typically only 120-125% of the maximum biomass. Therefore, possible phosphorus release from plant tissue lost prior to the period of maximum biomass would not greatly increase the annual estimate of total potentially recyclable plant phosphorus. A noticeable degree of macrophyte decay was observed at Lake Iroquois after mid-September in 1982.

Implications for Harvesting

Removal of aquatic plants by harvesting has the benefit of removing phosphorus from the lake system as well as improving the lake's recreational potential. If 100% plant removal were accomplished, approximately 69 kg of phosphorus would be removed. This is approximately 22% of the average annual external loading to the lake measured during 1982 and 1983. However, not all of the plant phosphorus removed would have contributed to lake loading. Also, total lake loading, both external and internal, would need to be considered to better determine the impact that plant removal might have on water quality. Most release of phosphorus by plants occurs in the fall after recreational lake use has declined and prior to an extended period of ice cover,

thereby limiting the effects of this loading source. Of course, 100% plant removal would be neither desirable nor achievable. Some optimal harvesting quota would need to be selected.

Improvements in lake water quality resulting from plant harvesting have been difficult to document. A large scale harvesting operation at Lake Sallie, a hypereutrophic Minnesota lake, succeeded in removing the equivalent of less than 2% of the annual phosphorus inflow, and produced no detectable change in water quality (Peterson et al., 1974). An improvement in water quality was observed at Lake Bomoseen, Vermont, during a three-year harvesting project which removed phosphorus equivalent to 5% of the total loading each year, but the improvement was attributed to sewage diversion, rather than to the harvesting project (Garrison, 1979). Most notably, a seven-year harvesting program at Chemung Lake, Ontario, which removed the phosphorus equivalent of up to 47% of the annual external loading in one year, produced no significant changes in ambient nutrient levels or algal biomass (Wile et al., 1979).

Although other benefits of harvesting are difficult to quantify, a number of positive effects are likely:

- improved recreational opportunities (boating, swimming, fishing)
- aesthetic benefits
- reduction in internal phosphorus loading
- reduction in plant regrowth following harvesting
- reduction in sediment accumulation rate
- improved feeding efficiency of adult fish on young fish which can result in improved growth rates of young fish due to less competition (Bartell and Breck, 1979).

There are also positive values associated with the presence of aquatic plants in a lake. Macrophytes provide habitat for fish and fish food organisms, and provide food and shelter for a wide variety of other organisms. Macrophyte beds buffer shoreline areas from waves, and can slow water movement and increase sedimentation rates near the mouths of surface inlets,

thereby protecting the main lake body. Macrophytes also provide structural support for epiphyte communities which do derive their phosphorus from the water and thus compete for nutrients with phytoplankton. Values derived from the presence of macrophytes, as well as from their removal, should be considered in a harvesting program.

SEPTIC SYSTEMS

Residential septic systems along a lakeshore have the potential for contributing phosphorus (P) to the lake if groundwater movement is toward the lake, as is the case at Lake Iroquois. Residential wastewater carries about 0.5 kg P/person/year (Ellis and Childs, 1973), which is derived largely from urine and feces, with food wastes and wash water making smaller contributions. The use of phosphorus-based laundry detergents can increase this value by two to four times (Ellis and Childs, 1973; Sawyer, 1965). However, the sale of phosphorus-based detergents was banned in Vermont in 1978.

The total annual input of P into lakeshore septic systems at Lake Iroquois was estimated to be about 60 kg, based on the following assumptions:

22 permanent, 70 seasonal residences
3 month occupancy of seasonal residences
3 persons/residence
0.5 kg P/person/year

Commonly, a large percentage of this P is retained by the soil through adsorption and mineralization processes (Jones and Lee, 1977). However, numerous cases of inadequate P retention have been cited (Hook *et al.*, 1978). Septic tank disposal field limitation ratings for shoreline soils at Lake Iroquois indicate severe use limitations for all soils (Allen, 1974; see Table 10). Limitations include a high water table, slow permeability (due to hardpans), slopes, and shallowness to bedrock. Although these limitations do not necessarily affect P retention, they do suggest an increased likelihood of poorly operating systems. Many of the systems are old, probably poorly designed, and close to the shoreline. Additionally, the public frequently perceives septic systems as a major source of pollutants to a lake. For these reasons, a well monitoring program was conducted during the summer of 1983 to determine what percentage of the P entering septic systems reaches the lake (see Methods section for a description of the well sampling program).

One or more wells (per system) were installed near the shoreline down gradient from the septic systems of three permanent and ten seasonal residences. Additional wells were installed at shoreline sites remote from septic systems to determine background conditions. A total of 43 wells were sampled. Results of sample analyses are given in Appendix D.

The percentage of P in septic tank effluent retained by the soil can be determined by a method described by Tofflemire et al. (1973) if the chloride and P concentration of the effluent and the natural groundwater are known with a reasonable degree of accuracy. Chloride serves as an effective tracer of effluent since it is one of the elements least retained by soils and is commonly present in much higher concentrations in effluent than in natural groundwater. The degree of effluent dilution by natural groundwater between the septic tank and the monitoring well can be estimated by the following formula:

$$D = \frac{(Cl_m - Cl_g)}{(Cl_i - Cl_g)}$$

where: D = dilution factor: volume of effluent/volume of mixture

Cl_g = natural groundwater Cl^- conc.

Cl_i = effluent Cl^- conc.

Cl_m = mixture Cl^- conc. (as found in the well sample)

The expected P concentration after dilution is then determined as follows:

$$\text{Expected P conc.} = D(P_i) + (1-D)(P_g)$$

where: P_i = effluent P conc.

P_g = natural groundwater P conc.

The percentage of effluent P retained by the soil is determined by:

$$\text{P retention} = 1 - \frac{\text{actual P conc. (in well)}}{\text{expected P conc.}}$$

The total P concentration of septic tank effluent (where phosphorus-free detergents are used) typically falls in the range of 5-10 mg/l (Brandes, 1978; Wagner, Heindel and Noyes, Inc., 1983). Natural groundwater total dissolved P concentrations at Lake Iroquois at sites not influenced by wetland seepage ranged from 4-12 ug/l with a mean of 7 ug/l (wells A-1,2,3,6,8; C-1,2). Table 45 can be used to determine which septic system sites exhibited 95% or greater P retention. For example, Table 45 shows that a dilution factor of 5% or greater and a P concentration of 13 ug/l or less indicates that the site achieved 95% or greater P retention (applying the more conservative effluent P concentration of 5 mg/l).

Table 45. Expected effluent contaminated ground water P concentrations at soil P retentions of 95 and 99%.

Dilution Factor	Assumed Effluent P Concentration (mg/l)	Well P Concentration (mg/l)[1] at 95% Soil Retention	Well P Concentration (mg/l)[1] at 99% Soil Retention
.05	5	.013	.003
.05	10	.025	.005
.10	5	.025	.005
.10	10	.050	.010
.20	5	.050	.010
.20	10	.100	.020
.33	5	.083	.017
.33	10	.167	.033
.50	5	.125	.025
.50	10	.250	.050
.75	5	.188	.038
.75	10	.375	.075
1.00	5	.250	.050
1.00	10	.500	.100

[1]Assuming background P concentrations are 0-12 ug/l.

Dilution factors for the east shore septic system wells could be determined by their chloride concentrations. Septic tank effluent chloride concentrations are usually in the range of 50-100 mg/l, with 75 mg/l appearing to be a good average value (Brandes, 1978; Metcalf and Eddy Inc., 1972; Robeck et al., 1963; Salvato, 1982). Natural groundwater chloride concentrations along the lake's east shore averaged 0.8 mg/l (based on well numbers A-4,6,8; B-17,28,32). Using these values, dilution factors were calculated for the east shore septic system wells, and are listed in Table 46.

Eight of the twelve east shore septic system wells had dilution factors of 5% or greater. Six of these eight had mean P concentrations less than 13 ug/l, and so exhibited a P retention greater than 95%. Two of the eight wells (numbers B-24,27) which were both below the same septic system, had slightly higher mean P concentrations (23.2 and 13.5 ug/l), but still indicated phosphorus retention values in excess of 90%. The four remaining east shore septic system wells had dilution factors less than 5%, and so P retention percentages could not be calculated since predicted P concentrations fell into the range of background P concentrations. However, since the well P concentrations were at background levels, it seems quite likely that a 95% or greater P retention would be applicable to these sites as well.

Dilution factors for west shore septic system wells could not be easily determined from chloride concentrations due to high and variable background chloride concentrations, resulting most probably from road salt application. West shore background wells (numbers A-1,2,3; C-2) had chloride concentrations ranging from 7.6 - 24.0 mg/l, while some septic system wells had chloride concentrations below this range. However, total nitrogen (TN) concentrations could be used to establish minimum dilution factors for most of these wells.

Septic tank effluent commonly has a TN concentration of 40-80 mg/l (Metcalf and Eddy Inc., 1972; Otis and Boyle, 1976; Walker et al., 1973), which is nearly all in the form of ammonium initially. TN concentrations (NH_4, NO_3, and NO_2) in six

Table 46. Mean Cl^-, P, and TN concentrations, dilution factors, and related data for east shore septic system wells.

Well No.	Cl^- (mg/l)	TDP (ug/l)	TN (mg/l)	Dilution Factor[1] (%)	Expected TN[2] (mg/l)	TN Removal[3] (%)
B-13	2.0	9.0	.353	2	1.2	71
B-16	10.8	7.0	.579	17	10.2	94
B-18	3.1	9.0	.052	3	1.8	97
B-19	7.0	6.6	.140	8	4.8	97
B-20	5.2	3.8	.108	6	3.6	97
B-23	2.4	8.4	.148	2	1.2	88
B-24	6.8	23.2	.211	8	4.8	96
B-25	6.3	8.0	.098	7	4.2	98
B-26	3.8	8.6	.076	4	2.4	97
B-27	8.8	13.5	.185	11	6.6	97
B-29	4.5	9.3	3.37	5	3.0	0
A-7	8.4	4.3	.094	10	6.0	98

[1]Assuming an effluent Cl^- concentration of 75 mg/l and a background Cl^- concentration of 0.8 mg/l.

[2]Based on dilution factors listed and assuming an effluent TN concentration of 60 mg/l and a background TN concentration of 0 mg/l.

[3] $1 - \frac{\text{actual TN}}{\text{expected TN}} \times 100$

background wells (numbers A-1,2,3,6,8; C-1,2) ranged from 0.060 - 0.225 mg/l, with a mean of 0.108 mg/l. Assuming an effluent TN concentration of 60 mg/l and a background TN concentration of 0.3 mg/l, a well TN concentration of 3.3 mg/l indicates a dilution factor of at least 5%. Since TN concentrations can be reduced by denitrification, the dilution factor could be an underestimate. Organic nitrogen, which was not measured, would also tend to cause underestimated dilution factors if it were present at significant concentrations in septic system wells. Organic nitrogen concentrations in background wells were probably insignificant.

Nine of the eleven west shore septic system wells had mean TN concentrations greater than 3.3 mg/l (Table 47), and thus dilution factors greater than 5%. Seven of these nine had mean P concentrations of 13 ug/l or less and so exhibited 95% or greater P retention. Two of these nine (numbers B-9, 10) which had mean P concentrations greater than 13 ug/l (17.4, 15.2 ug/l), appeared to have dilution factors of 39% or greater if an adjacent lightly contaminated well (number B-11) is assumed to represent the background chloride concentration. At this dilution factor, these sites exhibited a P retention of about 99%. The two remaining west shore septic system wells with low dilution factors had mean P concentrations within the range of background concentrations, and so again it seems likely that 95% or greater P retention occurred.

It appears that the septic systems examined exhibited a high degree of P retention (greater than 95% for all but one). However some systems did appear to deliver substantial quantities of nitrogen to the lake. Although nitrogen availability generally does not limit productivity as frequently as does phosphorus, it can contribute to eutrophication, and so deserves consideration.

East shore septic system wells at sites with shallow soils and exposed bedrock (all except number B-29) indicated that high rates of denitrification were occurring. Based on dilution factors determined by chloride concentrations (Table 46), an

Table 47. Mean Cl^-, P, and TN concentrations, dilution factors, and related data for west shore septic system wells.

Well No.	Cl^- (mg/l)	TDP (ug/l)	TN (mg/l)	Dilution Factor[1] (%)	Maximum Expected TN[2] (mg/l)	Maximum TN Removal[3] (%)
B-1	12.2	5.8	1.63	?- 20	12.2	87
B-2	20.2	4.2	4.54	5- 34	20.6	78
B-3	25.3	13.1	8.48	5- 42	25.4	67
B-4	9.3	7.4	5.89	5- 16	9.9	41
B-5	11.3	7.7	6.89	5- 19	11.6	41
B-6	15.2	10.1	15.88	5- 25	15.2	0
B-7	12.0	6.7	10.72	5- 20	12.2	12
B-8	8.4	4.6	7.59	5- 14	8.7	13
B-9	54.5	17.4	82.74[4]	5- 91	54.6	--
B-10	28.2	15.2	4.37	5- 47	28.4	85
B-11	6.3	6.7	1.51	?- 10	6.3	76

[1]Any well with a TN concentration greater than 3.3 mg/l has a minimum dilution factor of 5% assuming an effluent TN concentration of 60 mg/l and a background TN concentration of 0.3 mg/l. $\frac{3.3 - 0.3}{60 - 0.3} = 5\%$

Maximum dilution factors are based on an assumed effluent Cl^- concentration of 75 mg/l and a background Cl^- concentration of 0 mg/l.

[2]Based on maximum dilution factors listed and assuming an effluent TN concentration of 60 mg/l and a background TN concentration of 0.3 mg/l.

[3] $1 - \frac{\text{actual TN}}{\text{Max. expected TN}} \times 100$

[4]This TN concentration may not be due to effluent alone. See Appendix B.

assumed effluent TN concentration of 60 mg/l and a background TN concentration of 0 mg/l, nine of these eleven wells showed 94% or greater nitrogen removal. The remaining two wells (numbers B-13, 23) showed 71% and 88% nitrogen removal.

Denitrification requires that the nitrogen in septic tank effluent, which is initially in the form of ammonium (NH_4), be nitrified to the nitrate form (NO_3). This nitrate must then undergo reducing conditions in the presence of denitrifying bacteria. Under these conditions, the nitrate will be converted to gaseous nitrogen and released to the atmosphere (Sikora and Corey, 1976).

Denitrification is greatly influenced by soil conditions at a site. Loams and coarser textured soils are typically sufficiently aerated to allow most effluent ammonium to be converted to nitrate in a leachfield. Silt loams and finer textured soils in leachfields commonly yield a mixture of ammonium and nitrate, or all ammonium (Sikora and Corey, 1976).

Nine of the eleven east shore septic system wells were below sites with very rocky loam surface soils which would generally be conducive to leachfield ammonium oxidation. The wells below these sites frequently had nearly all nitrogen in the form of ammonium, with combined nitrate and nitrite concentrations below detection limits (less than 0.020 mg/l), which indicates that reducing conditions ultimately did develop. Two of these eleven wells (number B-13, 16) had combined nitrate and nitrite concentrations greater than ammonium concentrations. These wells were below sites with gravelly fine sandy loam soils. The occurrence of denitrification at these two sites, therefore, is not as clearly supported, but apparently does occur.

The one east shore septic system well (number B-29) located on a site with deeper soil indicated that this site produced very little nitrogen removal, and so nearly 100% of the nitrogen introduced to the system was delivered to the lake. This is consistent with the observation that the ground in the area of the septic system was wet, which would minimize the initial nitrification of ammonium necessary for denitrification to occur.

West shore septic system wells also indicated poorer nitrogen removal capabilities. Dilution factors could not be easily evaluated for these wells as mentioned previously, but maximum dilution factors could be determined by assuming an effluent chloride concentration of 75 mg/l and a background chloride concentration of 0 mg/l. Using these dilution factors, and assuming an effluent TN concentration of 60 mg/l and a background TN concentration of 0.3 mg/l, maximum expected TN concentrations could be calculated and compared to actual TN concentrations (Table 47).

Five of the eleven wells indicated less than 41% nitrogen removal. Nitrogen removal capabilities for the remaining well sites could not be adequately assessed with the existing data. Actual nitrogen removal could be much less than the maximum values listed in Table 47.

Some consideration of the limitations of the methods applied to estimate P and TN reductions is useful. The formulas presented assume that the effluent components being examined (Cl^-, P, TN) have similar patterns of movement through the soil. While this should generally be true, cases of differential effluent component transport have been documented (Childs et al., 1974). Sources of contaminants other than septic systems could also exist at some sites (lawn fertilization was not reported to have occurred recently at any of the sites). The fact that a fair number of wells were sampled over a period of several weeks should help minimize the influence of any such transient effects.

Some individual sites may not have been adequately assessed by the small number of shallow wells used. Restricted zones of high hydraulic conductivity could possibly transport much of a system's effluent to the lake, especially at sites with shallow bedrock. However, such possibilities could only have been assessed by a much more intensive monitoring program. Somewhat deeper wells (numbers B-4, 21, 26) at three sites generally showed lower concentrations of all effluent components than adjacent shallow wells, indicating that shallow wells were well suited for sampling the zone of peak effluent transport at most sites.

Although uncertainties exist with almost any groundwater sampling program, the data does provide good evidence that P retention greater than 95% occurred at most sites. The simple fact that most wells which showed evidence of effluent contamination (high chloride, nitrogen, and specific conductivity) had P concentrations similar to background wells supports this conclusion.

SOIL PHOSPHORUS ANALYSES

Soil phosphorus analyses were performed so that the relationship between soil erosion and phosphorus yield for the watershed could be more clearly established. The method chosen for soil phosphorus analysis was the same as that used for water sample phosphorus analysis, so problems in relating various soil phosphorus and water phosphorus analytical methods were avoided. Soil sample sites are shown in Figure 15 and described in Table 48. Soil phosphorus concentrations are listed in Table 49. Samples were not texturally classified. Surface soils in the watershed are predominately stone loams. The phosphorus content of the fine fraction ranged from 0.030-0.233% (Table 49), with a mean value of 0.096%. The phosphorus content of the medium fraction ranged from 0.008-0.075%, with a mean value of 0.025%.

Analytical phosphorus yield in the range of soil solutions prepared (100-200 mg/l for fine fraction; 150-300 mg/l for medium fraction) appeared to be relatively constant. Preliminary testing did show slight phosphorus yield reductions in a highly organic, phosphorus rich sample (no. 21 - 0.154% P) at soil solution concentrations greater than 250 mg/l, but not in a less organic sample (no. 18 - 0.095% P) at concentrations up to 1000 mg/l. Spiked samples showed good recoveries. Wet sieving of the medium fraction may have more effectively removed attached fines, and thus reduced the reported phosphorus concentrations of these samples.

Mean phosphorus contents for unfractionated samples within each general category are listed in Table 50. The coarse fraction of the samples was assumed to be phosphorus free for the purposes of these calculations. Using these values to evaluate phosphorus contributions to the lake resulting from soil erosion requires the application of an enrichment factor. The finer soil fraction, which has a higher phosphorus content, is more easily eroded and transported, so the sediment delivered to the lake will be richer in phosphorus than the original soil.

Table 48. Soil sample site descriptions.

Sample No.	Site Description
1	B horizon in ditch bank. 50 ft. above W-4 weir
2	road surface in W-3 sub-basin
3	channel sediment in W-4 weir pool
4	exposed hardpan in ditch bank. 50 ft. above W-4 weir
5	roadside ditch bottom
6	A horizon topsoil, undisturbed site
7	hauled topsoil at construction site
8	road surface (partially composed of crushed limestone)
9	hauled mound sand at construction site
10	drainage ditch bank at construction site
11	exposed road cut bank, 12-24" below surface
12	exposed road cut bank, 1-6" below surface
13	A horizon topsoil, wooded, undisturbed site
14	road surface in W-11 sub-basin
15	road surface in E-11 sub-basin
16	exposed road cut bank in E-9 sub-basin
17	roadside ditch bank in W-9 sub-basin
18	logging trail road surface
19	road surface in W-9 sub-basin
20	organic rich channel sediment in E-2 weir pool
21	organic rich channel sediment in W-6 weir pool

Table 49. Soil fraction percentages and phosphorus content.

Sample Number	% Coarse Fraction (>2.0 mm)	% Medium Fraction (0.25 mm, 2.0 mm)	Medium Fraction P Content (% dry wt)	% Fine Fraction (<0.25 mm)	Fine Fraction P Content (% dry wt)
surface soils					
1	13	13	.016	74	.092
5	14	31	.031	55	.124
6	0	7	.016	93	.095
7	3	9	.070	88	.140
9	3	29	.011	68	.058
10	18	17	.075	65	.102
12	10	15	.013	75	.046
13	22	13	.008	65	.040
17	22	16	.013	62	.085
subsoils					
4	13	13	.020	74	.106
11	17	17	.011	66	.030
16	28	52	.018	20	.081
road surfaces					
2	37	38	.016	25	.071
8	42	25	.027	33	.046
14	30	25	.017	45	.083
15	50	33	.025	17	.081
18	19	19	--	62	.095
19	40	36	.013	24	.070
stream channel deposits					
3	11	34	.022	55	.092
20	0	22	.046	78	.233
21	0	0	--	100	.154

Table 50. Mean phosphorus content of soils samples.

General Category	Mean P Content (% of dry wt.)	No. of Samples
surface soils	.075	9
subsoils	.043	3
road surfaces	.032	6
stream channel deposits	.134	3

FEASIBILITY STUDIES

The diagnostic studies on Lake Iroquois indicated that feasibility studies for lake restoration should be directed in several areas. First, it was apparent that ways should be explored to reduce the elevated phosphorus loading rates from certain portions of the watershed. Two types of watershed controls were examined: structural and regulatory. Feasibility studies for structural controls involved techniques to reduce erosion from roads, streambanks, and houselots, and nutrient runoff from farmland. Feasiblity studies for regulatory controls involved changes in local zoning ordinances and voluntary compliance by lakeshore property owners and local municipalities with guidelines for good erosion control practices. A final area of feasibility study was in-lake phosphorus controls to address the problem of internal phosphorus loading during the summer.

Structural Watershed Controls

A feasibility study for erosion control in the Lake Iroquois watershed was conducted by Barbaro and Ceto (1984) of the Winooski Conservation District and the U.S. Soil Conservation Service. The Barbaro and Ceto (1984) study involved a site by site field survey of areas of erosion and nutrient runoff. Specific recommendations were made for structural control measures at each site that was a significant problem. Streambank erosion was found to be the major problem in the Lake Iroquois watershed, although erosion from roads and house sites also contributed significant quantities of sediment and phosphorus to the lake. Specific recommendations for each erosion site can be found in the Barbaro and Ceto (1984) report and associated field sheets and maps. The study found that implementation of all recommended erosion control measures would reduce the annual phosphorus loading to the lake by 74 kg at a cost of $9,200.

Barbaro and Ceto also recommended certain nutrient runoff control measures for two farms in the watershed. The implementation of these measures, which include improvements in

existing manure handling facilities and diversion of runoff to grass filter strips, would reduce nutrient runoff from these farms by 60-80% (55-75 kg/yr), at a cost of $9,700.

Regulatory Watershed Controls

Many of the problems identified during the diagnostic studies and in the Barbara and Ceto (1984) report could have been avoided if proper erosion control practices had been followed. Regulation of land use activities in the watershed that cause erosion could be an effective way to improve and protect water quality in Lake Iroquois. Regulatory mechanisms available to control erosion in the watershed include the enactment of local zoning ordinances, as well as voluntary self-regulation by Towns and private landowners. Vermont's Land Use and Development Law (Act 250) also provides a mechanism to require proper erosion control practices for new large developments. Regulatory controls would be particularly important in protecting the lake from future degradation resulting from new development and road-building. Considering that most of the eastern portion of the Lake Iroquois watershed is currently undeveloped, it is clear that the potential for further water quality degradation of the lake is great if this area were to be developed without proper erosion controls.

The Barbaro and Ceto (1984) report identified three major erosion sources at Lake Iroquois: streambanks, roads, and house sites. Of these three, streambank erosion was the most critical. Specific measures to protect streambanks and reduce erosion are described in the Vermont Streambank Conservation Manual (Vermont Agency of Environmental Conservation, 1982) and in Barbaro and Ceto (1984). These reports provide guidance in implementing procedures such as the following:

fencing	rip-rapping
revegetating	limiting cutting
clearing channels	protecting vegetation
	constructing sediment traps

The requirement (when applicable) that a streambank erosion control plan consistent with the recommendations of these reports be approved as a condition for a local zoning or building permit, or for Act 250 approval, would be the best way to protect the lake from future degradation from this source.

Erosion from dirt roadbeds and ditches was a second major source of sediment and phosphorus to Lake Iroquois. Specific measures to reduce erosion from existing roads are described in the Vermont Backroad Maintenance Handbook (Zulick, 1979a) and in the Barbaro and Ceto (1984) report. These reports provide guidance for the following maintenance procedures:

road surface care	ditch maintenance
road pitch control	culvert protection
vegetation establishment	inspection

Many of the maintenance and erosion problems on dirt roads can be avoided by proper design and construction techniques. The Vermont Backroad Erosion Control Handbook (Zulick, 1979b) provides guidance in the following areas:

erosion control during construction
vegetation establishment
permanent structural erosion controls

Improved maintenance of existing roads by local governments and private road associations would be the best way to minimize their adverse effect on lake water quality. New road construction in the watershed should be permitted only with strict adherence to the erosion control practices described in the references cited above.

A third source of erosion in the Lake Iroquois watershed was house sites, particularly during construction. Because there is considerable potential for future house construction in the watershed, it would be important to insure that proper erosion control practices are followed for any new construction.

Specific measures to reduce erosion from house construction are described in the Vermont Handbook for Soil Erosion and Sediment Control on Construction Sites (Vermont Geological Survey, 1982) and in the Barbaro and Ceto (1984) report. These measures include the following:

site planning	sodding
water diversion	silt fencing
buffer strips	sediment basins
seeding and mulching	surface roughening
hay bale dikes	stabilizing entrance
snow fencing	

Strict conditions attached to building or land use permits for any new construction in the watershed insuring that all appropriate erosion control practices are followed would be the best way to protect the lake from degradation from this source.

In-Lake Phosphorus Controls

The diagnostic studies showed that phosphorus released from anoxic hypolimnetic sediments was occasionally transported vertically into the epilimnion where it stimulated algal growth. Because it was possible that this internal phosphorus release might still occur to some extent even after all necessary watershed controls were applied, feasibility studies were initiated for ways to control this process as well. Booker Associates (1984) conducted a study of three alternative techniques for reducing internal phosphorus loading: hypolimnetic aeration by compressed air lift, hypolimnetic aeration by pumped sidestream aeration, and phosphorus inactivation using alum. Based on the Booker Associates (1984) report, it appeared that the two leading feasible alternatives were aeration using an Atlas COPCO air lift system ($60,000 initial cost plus $5,000/yr operating cost) and alum treatment ($90,000 initial cost with no further operation cost).

SUMMARY AND CONCLUSIONS

Lake Iroquois is a highly valued recreational resource attracting and supporting such diverse uses as fishing, swimming, boating, and residency in lakeshore homes. These uses all depend on the maintenance of good water quality conditions in the lake. However, water quality problems related to lake eutrophication, such as excessive algae and aquatic plant growth, periodically interfere with these uses.

The diagnostic studies on Lake Iroquois were designed to determine the cause of the water quality problems by evaluating the various sources of phosphorus to the lake and the dynamics of phosphorus within the lake. The phosphorus budget studies showed that the hydrology and phosphorus loading to the lake was dominated by surface inflows. The streams entering Lake Iroquois contributed more than 80% of the total external phosphorus load to the lake during 1982 and 1983.

There were major differences in the phosphorus loading rates observed among the various sub-basins of the lake's watershed. The loading rate differences were related primarily to differences in land use among the sub-basins. The predominantly wooded sub-basins on the east shore of the lake and sub-basin W-10 (see Figure 17) had the lowest areal phosphorus export rates (0.8 - 0.13 kg/ha/yr), as would be expected from relatively undisturbed areas. The sub-basins along the lake's west shore having predominantly residential land use, including sub-basins W-2, W-3, W-4, W-5, W-6, W-8 and W-9 (see Figure 17), had considerably higher areal phosphorus loading rates (0.24 - 0.72 kg/ha/yr). The elevated phosphorus loading rates from these sub-basins were caused by erosion sources associated with residential development, such as ditching, modified or unstable drainageways, and exposed soil on construction sites. High phosphorus loading rates (0.21 - 0.49 kg/ha/yr) were also observed from sub-basin W-0 (see Figure 17). This sub-basin contained two dairy farms on which some nutrient runoff was still occurring, even after the construction of manure storage

facilities and other runoff control measures on these farms in recent years. The effect of the farm runoff on the lake was probably reduced by phosphorus attenuation within the wetland downstream of the farms. Dramatically high phosphorus loading rates (2.43 kg/ha/yr) were observed from sub-basin W-1 (see Figure 17) during 1983 as a result of erosion from poorly conducted excavation work the previous fall by one landowner. Bulldozing of grassland and other regrading and ditching activities caused massive erosion of sediment and phosphorus into the lake and produced the highest areal phosphorus loading rate from any sub-basin observed during the study.

The phosphorus contribution to Lake Iroquois from groundwater and septic systems was relatively minor. Groundwater contributed less than 3% of the total phosphorus budget of the lake, and phosphorus derived from shoreline septic systems was only about 1% of the total external phosphorus load. Studies on individual septic system sites showed that these systems did influence the quality of groundwater entering the lake, particularly with respect to nitrogen. However, their impact on phosphorus loading was small because of the very high phosphorus attenuation rates (greater than 95%) observed in the shoreline soils. Thus, it was concluded that septic systems were not a major influence on eutrophication in Lake Iroquois.

There remain, however, some good reasons for proper maintenance and improvement of existing shoreline septic systems. First, there are obvious public health concerns associated with failing systems and the possible introduction of disease-causing organisms that could infect swimmers and other lake users. A second reason for good septic system maintenance is that some of the aquatic plant growth in Lake Iroquois may be related to localized nutrient sources along the shoreline. A failing septic system could contribute excessive amounts of nutrients to the groundwater passing through the near-shore lake bed, and thereby nourish aquatic plant growth. Thus, while the shoreline septic systems at Lake Iroquois appear to have no significant impact on the lake-wide growth of algae, they may contribute to localized

problems of aquatic plant growth in some areas.

The internal dynamics of phosphorus within Lake Iroquois were found to have a major influence on water quality in the lake. A variety of internal sources contributed phosphorus to the surface waters of Lake Iroquois at various times of the year. These sources included release from anoxic hypolimnetic sediments, decay of macrophytes, and turbulent resuspension. Anoxic sediment release was particularly important during the summer. The lake is prone to this form of internal loading because of its small hypolimnetic volume and rapid hypolimnetiz oxygen depletion rates during the early summer. Phosphorus released from anoxic bottom sediments during the summer was intermittently transported upwards by vertical mixing events into the epilimnion, where algal growth was stimulated. The phosphorus released during the summer did not appear to remain in the water column through the winter to promote algal growth the following spring, however. Short-term phosphorus budgets indicated that, while there was a substantial net release of phosphorus during the summer, this was followed by a period of net sedimentation during the fall turnover and winter. The phosphorus sedimentation resulted from the re-oxygenation of the hypolimnetic waters during the fall turnover and a very effective iron precipitation mechanism in Lake Iroquois. As a result of the efficient sedimentation of phosphorus during and after the fall turnover, Lake Iroquois had annual phosphorus retention characteristics similar to those of lakes without substantial internal loading.

Paleolimnological studies provided a long-term historical perspective on water quality changes in Lake Iroquois. The lake has experienced cultural impacts on water quality since the time of first European settlement around 1800. Land clearing and widespread farming in the watershed during the 1800's corresponded with evidence preserved in the lake sediments of increased erosion and sedimentation in the lake. The major water quality changes have occurred in more recent decades, however. Sediment indicators of hypolimnetic oxygen depletion

showed increases since about 1930. Hypolimnetic oxygen depletion in Lake Iroquois may have been promoted by the rise in water level caused by the construction of the dam at the lake's outlet, but it is not clear why the evidence of oxygen depletion appeared 30 years after the dam construction. Sediment indicators of trophic state also showed a marked change to a more eutrophic condition since about 1920.

The general conclusion from the paleolimnological studies is that water quality in Lake Iroquois has been degraded substantially from its original condition by cultural activities in the watershed. The degradation has been mostly gradual, and the accelerated eutrophication of recent decades has probably been caused by sediment and nutrient inputs from the residential development that has occurred during this century, and from farming operations that have become more intensive and mechanized (although fewer in number).

The studies of aquatic macrophytes in Lake Iroquois showed the extent of their coverage and their species composition. The major area of macrophyte concentration in Lake Iroquois was in the lake's shallow northern end where the plants at times reached nuisance densities and impaired lake uses. There was some evidence that the plants have been increasing in density and areal coverage in recent years. Mechanical harvesting efforts began on Lake Iroquois during 1984, and this method represents the best way to control nuisance plant growth in the lake. Careful and limited mechanical harvesting in Lake Iroquois would provide a number of benefits, including improved recreational uses and aesthetics, and a reduction in internal phosphorus loading by plant senescence and decay. Care should be taken in the harvesting program to preserve the positive values of aquatic plants, including fish habitat, shoreline and sediment stabilization, and other biological values.

The results of the diagnostic studies on Lake Iroquois were used to determine the direction of subsequent feasibility studies for lake restoration. The paleolimnological work indicated that water quality in the lake has been degraded by cultural

activities in the watershed and therefore restoration efforts are believed to be appropriate for the lake. Erosion of sediment and nutrient runoff from residential land and farm areas within the watershed were found to be the chief causes of accelerated eutrophication in Lake Iroquois. These were the sources towards which the feasibility studies were primarily directed. It was also found that internal phosphorus loading from anoxic lake sediments played a role in stimulating algal growth during the summer. Consequently, feasibility studies were also initiated to control the internal loading.

Feasibility studies for watershed controls focused on two approaches: structural controls and regulatory controls. A variety of structural controls were determined to be feasible and beneficial. These measures included rip-rapping of eroding streambanks, construction of sediment traps, regrading and setting down of crushed stone on eroding roadbeds, and mulching and seeding of exposed soils at construction sites. These best management practices would reduce the annual phosphorus load to Lake Iroquois by about 74 kg at a cost of $9,200.

Additional structural controls were recommended for two farms in the watershed, including improvements in existing manure handling facilities and diversion of runoff to grass filter strips. These practices would reduce phosphorus runoff from these farms by about 60-80% (55-75 kg/yr) at a cost of $9,700.

If all the recommended structural watershed controls were implemented, the annual phosphorus loading to the lake would be reduced by about 140 kg from the existing total load of 239-359 kg/yr measured during 1982-1983. This 35-60% reduction could be achieved at relatively minor cost ($18,900), and should have substantial water quality benefits. A lake modeling analysis indicated that a 40% loading reduction would lower average lake phosphorus concentrations by 32%, from around 31 ug/l to 21 ug/l. Algal abundance, as indicated by chlorophyll-a levels, would decline by about the same percentage. Average Secchi disc transparency would increase from around 3.4 meters to 4.4 meters. The areal hypolimnetic oxygen depletion rate would be slowed by

about 33%, delaying the onset of anoxia about 20 days, thereby reducing the extent of internal phosphorus recycling during the summer.

A number of regulatory watershed controls were also found to be feasible as a way to protect the lake from the effects of future development. Much of the lake's watershed is currently undeveloped, and there is potential for further water quality degradation if this area were developed without proper erosion and nutrient runoff controls. Specific erosion control measures for streambanks, roads, and construction sites could be implemented through a variety of regulatory mechanisms, including Vermont's Land Use and Development Law (Act 250 permits), Town zoning ordinances and building permits, and voluntary self-regulation by local lake and road associations or private landowners. These regulatory controls would involve minimal cost to State and local governments and, if enforced energetically, could be of great value in protecting water quality in Lake Iroquois.

Feasibility studies to reduce the internal recycling of phosphorus in Lake Iroquois focused on two alternatives: alum treatment and hypolimnetic aeration. Both alternatives would probably be effective, but their cost would be quite high ($60,000 for hypolimnetic aeration and $90,000 for alum treatment). Furthermore, in-lake treatments such as these would be appropriate only if all possible means to reduce phosphorus loading from the watershed were implemented first. If the recommended structural watershed controls were implemented, and if strong regulatory programs for lake protection were in place, it might be unnecessary to undertake the in-lake efforts. Therefore, the best course of action would be to implement the watershed controls, evaluate the water quality gains, and then pursue the in-lake controls only if all the watershed problems are corrected and the internal phosphorus recycling continues to cause undesirable levels of algae during the summer.

In summary, the opportunity exists to improve water quality in Lake Iroquois and protect the lake from future degradation by

controlling erosion and nutrient runoff from residential development and farmland in the watershed through a combination of structural measures and regulatory efforts. The cost of these measures is relatively minor. The key to their success will be the willingness and dedication of local residents and Town officials in pursuing the recommendations that follow.

RECOMMENDATIONS

1. The structural watershed controls recommended in the Barbaro and Ceto (1984) report should be implemented in order to reduce erosion from residential areas and nutrient runoff from farmland. Technical assistance in designing these structures should be requested from the U.S. Soil Conservation Service. Funding should be sought from the Federal Clean Lakes Program, the Vermont Aquatic Nuisance Control Program, or other sources.

2. The regulatory watershed controls recommended in this report should be implemented in order to protect water quality in Lake Iroquois from the effects of future development. The possibilities for regulatory controls should be explored through discussions with town officials, the District Act 250 Commission staff, and the local residents and lake associations. Technical assistance in establishing regulations and in monitoring compliance should be sought from the Chittenden County Regional Planning Commission, the Vermont Association of Conservation Districts, and the U.S. Soil Conservation Service.

3. Any in-lake treatment such as alum treatment or hypolimnetic aeration to control internal phosphorus loading should be postponed until all possible structural and regulatory watershed controls are in place. In-lake treatments should then be considered only if undesirable levels of algae continue to exist as a result of vertical transport of hypolimnetic phosphorus during the summer.

4. Mechanical harvesting of aquatic plants in Lake Iroquois should continue, with care taken to balance the need for selective removal of nuisance aquatic plants with the need to preserve the beneficial values of these plants.

5. The wetland at the north end of the lake (in sub-basin W-0) should be preserved to help protect Lake Iroquois from the effects of nutrient runoff from the farmland upstream.

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APPENDIX

Appendix A. Lake level measurements.

Date	Staff Gage Elevation (ft.)	Date	Staff Gage Elevation (ft.)	Date	Staff Gage Elevation (ft.)
11-17-81	19.15	05-27-82	18.89	08-15-82	17.48
11-18-81	19.22	05-31-82	18.77	08-17-82	17.40
11-20-81	19.30	06-01-82	18.75	08-18-82	17.40
11-21-81	19.40	06-02-82	18.85	08-20-82	17.34
11-22-81	19.38	06-13-82	18.73	08-23-82	17.22
11-23-81	19.34	06-14-82	18.71	08-24-82	17.22
11-24-81	19.32	06-15-82	18.69	08-26-82	17.18
11-28-81	19.19	06-17-82	18.69	08-29-82	17.12
11-29-81	19.19	06-20-82	18.69	08-31-82	17.08
11-30-81	19.17	06-22-82	18.63	09-02-82	17.04
12-03-81	19.19	06-23-82	18.84	09-06-82	17.00
12-04-81	19.19	06-24-82	18.85	09-09-82	16.90
12-05-81	19.09	06-28-82	18.82	09-16-82	16.79
04-16-82	19.27	07-06-82	18.72	09-23-82	16.63
04-23-82	19.28	07-12-82	18.51	09-28-82	16.62
04-24-82	19.28	07-20-82	18.19	10-06-82	16.51
04-26-82	19.19	07-21-82	18.17	10-15-82	16.54
04-27-82	19.20	07-24-82	18.01	10-21-82	16.52
04-28-82	19.19	07-25-82	17.99	10-27-82	16.47
04-29-82	19.38	07-27-82	17.89	11-04-82	16.50
05-02-82	19.05	07-28-82	17.95	11-10-82	16.59
05-03-82	18.99	07-30-82	17.89	11-17-82	16.58
05-06-82	18.99	08-01-82	17.87	11-23-82	16.58
05-08-82	18.97	08-03-82	17.79	11-29-82	16.76
05-09-82	19.05	08-04-82	17.75	12-08-82	16.91
05-18-82	18.90	08-05-82	17.72	12-17-82	17.18
05-19-82	18.87	08-07-82	17.64	12-28-82	17.55
05-20-82	18.89	08-09-82	17.89	01-11-83	17.83
05-21-82	18.91	08-10-82	17.68	02-03-83	18.97
05-23-82	18.97	08-14-82	17.52	02-04-83	19.15
05-24-82	18.94	08-15-82	17.48	02-10-83	19.19

Appendix A. (cont.)

Date	Staff Gage Elevation (ft.)	Date	Staff Gage Elevation (ft.)	Date	Staff Gage Elevation (ft.)
02-17-83	19.08	05-18-83	19.28	08-09-83	18.50
03-02-83	19.02	05-19-83	19.24	08-17-83	18.41
03-07-83	19.05	05-23-83	19.19	08-23-83	18.32
03-09-83	19.17	05-24-83	19.35	08-24-83	18.29
03-10-83	19.21	05-25-83	19.33	08-29-83	19.29
03-15-83	19.28	05-27-83	19.69	08-31-83	18.28
03-18-83	19.48	05-31-83	19.41	09-01-83	18.29
03-19-83	19.55	06-03-83	19.39	09-02-83	18.25
03-20-83	19.65	06-06-83	19.27	09-04-83	18.19
03-21-83	19.60	06-07-83	19.48	09-07-83	18.13
03-22-83	19.58	06-09-83	19.45	09-14-83	17.97
03-25-83	19.35	06-10-83	19.37	09-20-83	17.86
03-30-83	19.24	06-13-83	19.19	09-21-83	17.87
04-03-83	19.20	06-14-83	19.18	09-22-83	18.19
04-04-83	19.23	06-15-83	19.15	09-23-83	18.19
04-07-83	19.22	06-16-83	19.11	09-26-83	18.19
04-11-83	19.38	06-19-83	19.17	09-28-83	18.11
04-15-83	10.33	06-21-83	19.01	10-05-83	18.13
04-16-83	19.28	06-23-83	18.99	10-12-83	18.22
04-19-83	19.26	06-26-83	18.91	10-14-83	18.33
04-21-83	19.46	06-27-83	18.92	10-19-83	18.21
04-22-83	19.53	06-29-83	18.89	10-27-83	18.31
04-24-83	19.76	07-05-83	18.82	11-02-83	18.35
04-25-83	19.98	07-06-83	18.81	11-04-83	18.54
04-26-83	19.94	07-07-83	18.80	11-05-83	18.97
04-27-83	19.78	07-13-83	18.71	11-07-83	19.09
05-02-83	19.65	07-19-83	18.60	11-09-83	19.15
05-05-83	19.55	07-28-83	18.40	11-11-83	19.23
05-09-83	19.48	08-01-83	18.49	11-12-83	19.61
05-12-83	19.54	08-03-83	18.56	11-13-83	19.49
05-16-83	19.32	08-05-83	18.52	11-14-83	19.44

Appendix A. (cont.)

Date	Staff Gage Elevation (ft.)
11-16-83	19.38
11-21-83	19.51
11-22-83	19.56
11-25-83	19.77
11-29-83	19.54
12-07-83	19.87
12-08-83	19.77
12-13-83	19.63
12-14-83	20.04
12-15-83	20.03
12-16-83	19.84
12-17-83	19.67

Appendix B, Well Descriptions and Comments

Key to Location Sketches

House

Septic Tank or Cesspool

Leachfield

Drywell

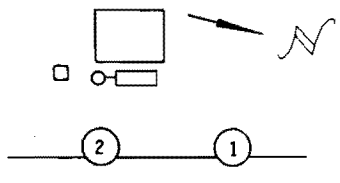
Sampling Well

Shoreline

Well No.	Depth (m)	Comments
A-1	0.9	Immediate shore area is low-lying and swampy.
A-2	1.2	Appeared to penetrate a hardpan layer. Very low water yield restricted sample collection.
A-3	1.3	----
A-4	1.3	May have been slightly influenced by septic tank effluent. See B-14.
A-5	1.2	Located in organic soil and so may not be representative of the bulk of local groundwater inflow.
A-6	1.4	----
A-7	1.4	Nearby septic system apparently influences this well. See B-22.
A-8	1.2	---
A-9	1.2	Wetland areas are upgradient from this well. Well was lost due to an uprooted tree.
B-1	1.2	Located downgradient from septic system described below.
B-2	1.2	

		water use
type of system	- tank and leachfield	
age of system	- 8 years	toilet and sinks - yes
setback distance	- 31 m	shower/bath - yes
recent maintenance	- no	washing machine - yes (to drywell)
number of users	- 2	other - no
permanent or seasonal	- permanent	water supply - lake

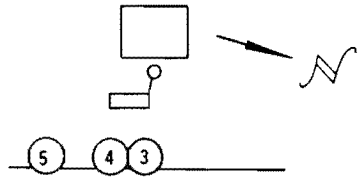
location sketch (not to scale)



Well No.	Depth (m)	Comments
B-3 B-4 B-5	1.2 2.7 1.2	Located downgradient from septic system described below. Wells B-3 and 5 have 91.4 cm screen lengths. Well B-4 has a 30.5 cm. screen length.

		water use
type of system	- tank and leachfield	
age of system	- 13 years	toilet and sinks - yes
setback distance	- 10 m	shower/bath - yes
recent maintenance	- pumped 3-4 years ago	washing machine - yes (to drywell)
number of users	- 2	other - no
permanent or seasonal	- permanent	water supply - lake

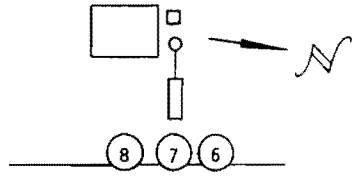
location sketch (not to scale)



B-6 B-7 B-8	1.2 1.2 1.2	Located downgradient from septic system described below.

		water use
type of system	- tank and leachfield	
age of system	- 15 years	toilet and sinks - yes
setback distance	- 10 m	shower/bath - yes
recent maintenance	- pumped 2 years ago	washing machine - yes (to drywell)
number of users	- 2	other - no
permanent or seasonal	- seasonal	water supply - lake

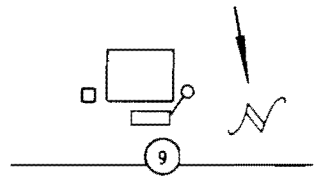
location sketch (not to scale)



B-9	2.0	Located downgradient from septic system described below. This septic tank was replaced during the monitoring period. Release of septage from the old tank may have influenced well samples.

type of system	- tank and leachfield	water use
age of system	- 8 years	toilet and sinks - yes (sinks to drywell)
setback distance	- 5 m	shower/bath - yes (to drywell)
recent maintenance	- tank replaced	washing machine - yes (to drywell)
number of users	- 2	other - no
permanent or seasonal	- permanent	water supply - lake

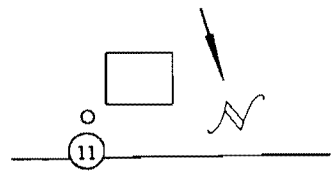
location sketch (not to scale)



Well No.	Depth (m)	Comments
B-10	1.2	Located between two adjacent septic systems. This well was buried by a shoreline improvement project.
B-11	1.2	Located downgradient from septic system described below.

type of system	- cesspool	water use
age of system		toilet and sinks - yes
setback distance	- 5 m	shower/bath - no
recent maintenance		washing machine - no
number of users		other - no
permanent or seasonal	- occasional use only	water supply - lake

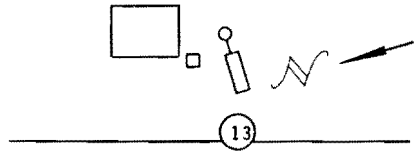
location sketch (not to scale)



B-12	1.2	Located downgradient from a wetland area. May also be influenced by upgradient garden and septic system.
B-13	1.2	Located downgradient from septic system described below.

type of system	- tank and leachfield	water use
age of system	- 52 years	toilet and sinks - yes
setback distance	- 7 m	shower/bath - yes (to drywell)
recent maintenance	- no	washing machine - no
number of users	- 3-4	other - no
permanent or seasonal	- seasonal	water supply - lake

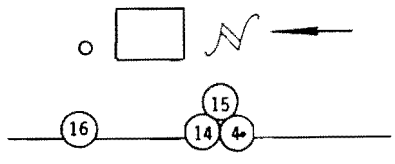
location sketch (not to scale)



Well No.	Depth (m)	Comments
B-14	0.6	
B-15	1.2	Located downgradient from septic system described below. Well A-4 was also located at this site.
B-16	1.2	

type of system	- cesspool	water use
age of system		toilet and sinks - yes
setback distance	- 20 m	shower/bath -
recent maintenance	- no	washing machine -
number of users	- 2	other - no
permanent or seasonal	- seasonal	water supply - lake

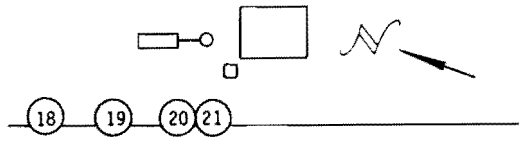
location sketch (not to scale)



B-17	1.2	Located downgradient from a wetland area which was reportedly once used as a dump site.
B-18	0.9	
B-19	1.2	Located downgradient from septic system described below. Wells B-18, 19, 20 have 91.4 cm. screen lengths. Well B-21 has a 30.5 cm. screen length.
B-20	1.2	
B-21	2.7	

type of system	- tank and leachfield	water use
age of system	- 37 years	toilet and sinks - yes
setback distance	- 10 m	shower/bath - yes
recent maintenance	- pumped 5-6 years ago	washing machine - yes (to drywell)
number of users	- 3-4	other - no
permanent or seasonal	- seasonal	water supply - lake

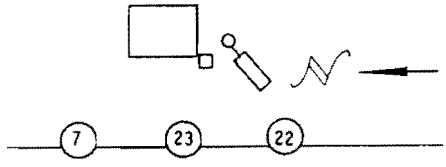
location sketch (not to scale)



Well No.	Depth (m)	Comments
B-22	1.2	Located downgradient from septic system described below. Well A-7 was also located at this site.
B-23	1.2	

type of system	- tank and leachfield	water use
age of system	- 28 years	toilet and sinks - yes
setback distance	- 9 m	shower/bath - yes
recent maintenance	- pumped 1 year ago	washing machine - yes (to drywell)
number of users	- 2-3	other - dishwasher (to drywell)
permanent or seasonal	- seasonal	water supply - lake

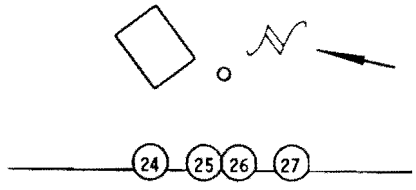
location sketch (not to scale)



B-24	1.2	Located below septic system described below. Wells B-24, 25, 27 have 91.4 cm. screen lengths. Well B-26 has a 30.5 cm. screen length.
B-25	1.2	
B-26	2.0	
B-27	1.2	

type of system		water use
age of system	- 26 years	toilet and sinks - yes
setback distance	- 11 m	shower/bath - yes
recent maintenance	- no	washing machine - yes
number of users	- 2	other - dishwasher
permanent or seasonal	- seasonal	water supply - lake

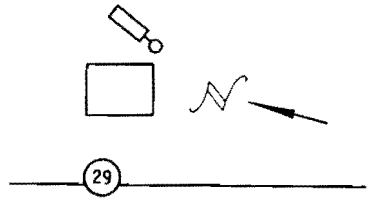
location sketch (not to scale)



Well No.	Depth (m)	Comments
B-28	1.2	Located below a septic system. Sampling was discontinued for this well since it did not appear to be influenced by effluent.
B-29	1.2	Located below septic system described below.

type of system	- tank and leachfield	wateruse
		water supply - lake
setback distance	- 21 m	
permanent or seasonal	- seasonal	

location sketch (not to scale)



B-30	1.2	Located downgradient from a wetland area.
B-31	1.2	Located downgradient from a wetland area. This well was located in approximately the same location as well A-9.
B-32	1.2	----
C-1	2.0	----
C-2	2.0	----

Appendix C. Groundwater gradients (elevation above lake level) for wells C-1 and C-2.

Date	Elevation above Lake Level (ft.) C-1	C-2	Date	Elevation above Lake Level (ft.) C-1	C-2
11-29-82	1.30	3.70	11-21-83	--	1.51
11-30-82	1.49	--	11-29-83	0.89	1.41
12-01-82	--	3.65	12-07-83	0.69	1.17
12-16-82	1.37	3.68	12-13-83	0.48	0.97
12-23-82	1.47	2.94	12-28-83	0.66	1.36
12-28-82	1.76	3.14			
01-06-83	1.35	2.71			
02-03-83	1.03	2.25			
03-02-83	0.80	1.57			
03-09-93	1.00	1.63			
03-30-83	0.82	1.47			
04-04-83	0.85	1.54			
04-21-83	--	1.42			
04-26-83	0.71	1.17			
05-02-83	0.83	1.31			
05-12-83	0.83	1.29			
05-23-83	0.77	1.48			
06-06-83	0.77	1.47			
06-21-83	0.54	1.24			
06-27-83	0.56	1.46			
07-05-83	0.32	1.08			
07-28-83	0.03	0.76			
08-17-83	0.22	0.26			
09-15-83	0.07	1.16			
09-28-83	0.66	1.87			
10-05-83	0.85	2.22			
10-12-83	0.90	2.13			
10-19-83	0.95	2.18			
10-27-83	1.16	2.35			
11-09-83	0.88	1.49			
11-14-83	0.86	1.24			

Appendix C (cont.)

Mean Monthly Groundwater Inflow (Derived from Well C-2)

Month	Mean Gradient (ft.)	% of Groundwater Inflow	Daily Groundwater Inflow (X286 CFSD)
Jan.	2.56	14.6	1.3470
Feb.	1.97	11.3	1.1542
Mar.	1.56	8.9	0.8211
Apr.	1.38	7.9	0.7531
May	1.36	7.8	0.7196
Jun.	1.39	8.0	0.7627
Jul.	0.92	5.3	0.4890
Aug.	0.26	1.5	0.1384
Sep.	1.25	7.2	0.6864
Oct.	2.22	12.7	1.1717
Nov.	1.41	8.1	0.7722
Dec.	1.17	6.7	0.6181

Appendix D. Well Sample Analyses

WELL NO.	DATE 10-29-82	12-07-82	05-13-83	06-23, 24, 28-83	07-6, 7-83	07-21, 22-83	08-4, 5-83	08-18, 19, 24-83	08-31, 09-2-83	09-13, 14, 16-83	09-29, 30-83	10-17, 18	10-31, 11-12	PARAMETER
A-1	10	17	12	9		17		100		13		8	10	TDP (ug/l)
				233		249		252		246		243	255	Cond. (umhos)
	13.0	12.8	15.4	15.7		14.5		14.4		12.4		13.5	14.8	Cl^- (mg/l)
				.020		.020		.020		.020*		.020*	.020	$NO_3 + NO_2 - N$ (mg/l)
				.023		.047		.034		.054		.042	.107	$NH_4 - N$ (mg/l)
A-2					8									TDP (ug/l)
				291		328		310		272		320	265	Cond. (umhos)
				9.1		7.6		9.9		7.9		9.5	8.0	Cl^- (mg/l)
														$NO_3 + NO_2 - N$ (mg/l)
														$NH_4 - N$ (mg/l)
A-3	6	4	7	3		3		6		6		5	6	TDP (ug/l)
				167		142		213		130		200	202	Cond. (umhos)
	10.1	16.6	2.0	9.4		7.7		11.2		7.0		8.9	11.1	Cl^- (mg/l)
				.020		.192		.224		.238		.202	.143	$NO_3 + NO_2 - N$ (mg/l)
				.071		.016		.030		.044		.116	.058	$NH_4 - N$ (mg/l)
A-4	15	10	8	5	3	7	5	9	10	16		7	9	TDP (ug/l)
				79	72	74	81	86	88	82		89	104	Cond. (umhos)
	1.3	1.2	0.9	1.1	0.9	1.0	1.2	1.2	1.3	1.2		0.3	0.6	Cl^- (mg/l)
				.386	.345	.367	.326	.684	.670	.479		1.24	.855	$NO_3 + NO_2 - N$ (mg/l)
				.015	.018	.032	.098	.030	.136	.030		.189	.014	$NH_4 - N$ (mg/l)
A-5		18	112	152		113		41		44		37	48	TDP (ug/l)
				425		422		389		375		358	350	Cond. (umhos)
		10.6	10.4	10.2		9.5		9.5		8.6		8.8	8.6	Cl^- (mg/l)
				.020		.020		.058		.020*		.020	.020	$NO_3 + NO_2 - N$ (mg/l)
				.322		.210		.154		.236		.153	.095	$NH_4 - N$ (mg/l)
A-6		8	10	9		4		6		10		8	8	TDP (ug/l)
				139		168		218		179		123	108	Cond. (umhos)
		1.0	0.8	1.5		0.8		0.7		0.7		0.3	0.2	Cl^- (mg/l)
				.020		.025		.042		.020*		.020	.020	$NO_3 + NO_2 - N$ (mg/l)
				.093		.110		.098		.085		.082	.060	$NH_4 - N$ (mg/l)

WELL NO.	DATE 10-29-82	12-07-82	05-13-83	06-23, 24, 28-83	07-6, 7-83	07-21, 22-83	08-4, 5-83	08-18, 19, 24-83	08-31, 09-2-83	09-13, 14, 16-83	09-29, 30-83	10-17, 18	10-31, 11-12	PARAMETER
A-7	5	3	3	3	3	3	4	5	6	8		5	6	TDP (ug/l)
				65	71	86	92	94	93	92		100	98	Cond. (umhos)
	7.0	5.6	1.0	7.2	8.6	8.7	8.7	8.8	8.6	8.8		8.2	8.1	Cl^- (mg/l)
				.020	.020	.020	.020	.039	.020	.020		.020	.020	$NO_3 + NO_2 - N$ (mg/l)
				.018	.380	.029	.036	.038	.038	.057		.091	.040	$NH_4 - N$ (mg/l)
A-8		3	4	3	3	3	3	6		8		5	8	TDP (ug/l)
				43	44	50	60	67		65		81	78	Cond. (umhos)
		0.6	0.8	0.4	0.3	0.3	0.4	0.6		1.6		1.5	0.7	Cl^- (mg/l)
				.020	.067	.014	.020	.020		.020*		.020	.020	$NO_3 + NO_2 - N$ (mg/l)
				.017	.014	.032	.036	.021		.282		.022	.025	$NH_4 - N$ (mg/l)
A-9	121	70	75											TDP (ug/l)
														Cond. (umhos)
	1.8	2.2	1.8											Cl^- (mg/l)
														$NO_3 + NO_2 - N$ (mg/l)
														$NH_4 - N$ (mg/l)
B-1				9	3	4	4	6	6	10	7	5	5	TDP (ug/l)
				204	204	218	222	225	227	217	235	236	235	Cond. (umhos)
				11.5	11.6	11.3	11.7	12.4	12.5	12.7	12.8	12.2	13.0	Cl^- (mg/l)
				.367	.697	.831	.934	1.31	1.64	1.70	2.04	2.10	2.10	$NO_3 + NO_2 - N$ (mg/l)
				.089	.032	.032	.076	2.1	.035	.057	.098	.041	.066	$NH_4 - N$ (mg/l)
B-2				6	3	3	3	6	6	5	5	3	6	TDP (ug/l)
				331	366	420	376	375	392	368	363	343	338	Cond. (umhos)
				19.4	20.1	20.7	20.3	21.2	22.2	22.4	19.2	18.3	18.5	Cl^- (mg/l)
				4.44	5.80	5.83	5.75	5.41	5.40	5.65	4.53	4.31	4.66	$NO_3 + NO_2 - N$ (mg/l)
				.022	.046	.039	.028	.034	.022	.111	.010	.061	.510	$NH_4 - N$ (mg/l)
B-3				9	31	13	10	12	10	11	8	20	7	TDP (ug/l)
				670	700	770	570	613	590	660	498	470	480	Cond. (umhos)
				28.3	34.8	33.1	22.8	25.7	25.2	28.0	20.0	17.0	18.2	Cl^- (mg/l)
				6.88	11.0	5.37	6.41	8.92	7.70		6.31	7.15	11.4	$NO_3 + NO_2 - N$ (mg/l)
				.780	1.03	1.50	.520	.740	.330		.042	.063	.189	$NH_4 - N$ (mg/l)

WELL NO.	DATE 10-29-82	12-07-82	05-13-83	06-23, 24, 28-83	07-6, 7-83	07-21, 22-83	08-4, 5-83	08-18, 19, 24-83	08-31, 09-2-83	09-13, 14, 16-83	09-29, 30-83	10-17, 18	10-31, 11-12	PARAMETER
B-4							15	8	7	8	3	5	6	TDP (ug/l)
							470	462	439	481	452	440	458	Cond. (umhos)
							9.7	9.0	9.4	9.6	8.7	8.5	10.1	Cl^- (mg/l)
							5.17	5.24	5.32	5.75	5.65	5.95	6.60	$NO_3 + NO_2 - N$ (mg/l)
							.312	.118	.652	.061	.179	.173	.046	$NH_4 - N$ (mg/l)
B-5				12	6	6	6	8	8	9	8	6	8	TDP (ug/l)
				359	297	398	351	370	361	386	342	325	342	Cond. (umhos)
				11.8	10.4	13.6	10.2	11.7	10.6	11.1	11.6	10.3	11.4	Cl^- (mg/l)
				6.29	5.55	6.88	5.54	6.16	6.18		6.88	7.60	10.0	$NO_3 + NO_2 - N$ (mg/l)
				.286	.132	.065	.124	.073	.098		.020	.104	.060	$NH_4 - N$ (mg/l)
B-6				10	9	13	9	10	14	13	7	9	7	TDP (ug/l)
				343	370	475	479	494	470	503	495	530	579	Cond. (umhos)
				12.1	11.4	11.9	12.1	13.5	14.8	15.8	18.0	19.4	22.6	Cl^- (mg/l)
				2.62	2.15	2.41	2.14	2.40	5.69	3.65	12.0	4.00	10.9	$NO_3 + NO_2 - N$ (mg/l)
				7.61	8.50	10.6	10.0	12.7	10.6	11.2	12.0	14.1	13.5	$NH_4 - N$ (mg/l)
B-7				10	9	7	4	7	7	8	5	4	6	TDP (ug/l)
				455	455	481	461	435	375	350	317	283	291	Cond. (umhos)
				19.9	19.4	18.6	14.6	12.7	8.1	7.7	7.4	5.6	5.9	Cl^- (mg/l)
				7.39	9.05	7.64	7.02	5.74	5.46	4.64	6.72	5.80	6.05	$NO_3 + NO_2 - N$ (mg/l)
				4.78	5.15	5.10	5.60	6.05	4.54	4.40	3.36	2.09	1.57	$NH_4 - N$ (mg/l)
B-8				4	4	3	3	8	5	9	4	3	6	TDP (ug/l)
				445	475	470	440	416	402	385	353	338	351	Cond. (umhos)
				16.0	14.5	11.7	9.6	7.8	6.3	5.5	4.7	3.9	4.4	Cl^- (mg/l)
				8.18	11.1	10.5	8.32	5.95	5.36	5.05	5.45	4.78	4.89	$NO_3 + NO_2 - N$ (mg/l)
				.090	.201	.440	.700	.790	.767	.673	.707	.532	.600	$NH_4 - N$ (mg/l)
B-9				21	12	11	14	11	13	18	11	44	19	TDP (ug/l)
				293	590	495	810	880	868	937		660	920	Cond. (umhos)
				27.6	40.8	44.9	56.7	61.5	56.5	61.3	71.5	62.0	62.2	Cl^- (mg/l)
				7.25	42.0	21.8	61.0	67.5	71.5	72.0	102	15.2	66.5	$NO_3 + NO_2 - N$ (mg/l)
				6.91	12.5	16.2	36.4	42.8	50.6	39.4	40.0	24.9	30.9	$NH_4 - N$ (mg/l)

WELL NO.	DATE 10-29-82	12-07-82	05-13-83	06-23, 24, 28-83	07-6, 7-83	07-21, 22-83	08-4, 5-83	08-18, 19, 24-83	08-31, 09-2-83	09-13, 14, 16-83	09-29, 30-83	10-17, 18	10-31, 11-12	PARAMETER
B-10				25	14	14	12	11						TDP (ug/l)
				340	420	450	560	511						Cond. (umhos)
				9.9	14.6	26.8	45.1	44.8						Cl^- (mg/l)
				.020	.049	4.55	6.09	7.22						$NO_3 + NO_2 - N$ (mg/l)
				.303	.390	.620	1.10	1.41						$NH_4 - N$ (mg/l)
B-11				8	4	3	7	8	14	9	4	6	4	TDP (ug/l)
				315	292	305	318	293	301	297		258	265	Cond. (umhos)
				6.4	5.4	9.2	6.0	4.9	5.2	5.6	8.0	5.6	7.1	Cl^- (mg/l)
				.636	.713	.468	1.56	1.85	2.10	2.68	1.72	1.38	1.58	$NO_3 + NO_2 - N$ (mg/l)
				.030	.032	.026	.034	.030	.043	.074	.041	.042	.062	$NH_4 - N$ (mg/l)
B-12										37		47	49	TDP (ug/l)
										308		251	250	Cond. (umhos)
										12.1		9.5	8.8	Cl^- (mg/l)
										.020*		.020	.020	$NO_3 + NO_2 - N$ (mg/l)
										.274		2.58	1.68	$NH_4 - N$ (mg/l)
B-13				5	11					11	8	9	10	TDP (ug/l)
				284	292					287		259	250	Cond. (umhos)
				1.9	1.5					3.7	1.6	2.0	1.6	Cl^- (mg/l)
				.300	.175					.233	.250	.274	.236	$NO_3 + NO_2 - N$ (mg/l)
				.079	.102					.124	.156	.122	.069	$NH_4 - N$ (mg/l)
B-14								18	13					TDP (ug/l)
								182	167				88	Cond. (umhos)
								1.0	1.0				1.0	Cl^- (mg/l)
								.952	.604					$NO_3 + NO_2 - N$ (mg/l)
								.170	.282					$NH_4 - N$ (mg/l)
B-15								9	7					TDP (ug/l)
								88	90					Cond. (umhos)
								0.8	0.8					Cl^- (mg/l)
								.829	.738					$NO_3 + NO_2 - N$ (mg/l)
								.059	.084					$NH_4 - N$ (mg/l)

WELL NO.	DATE 10-29-82	12-07-82	05-13-83	06-23, 24, 28-83	07-6, 7-83	07-21, 22-83	08-4, 5-83	08-18, 19, 24-83	08-31, 09-2-83	09-13, 14, 16-83	09-29, 30-83	10-17, 18	10-31, 11-12	PARAMETER
B-16				3	10	3	5	8	6	8	3	4	21	TDP (ug/l)
				231	214	215	230	235	218	217		173	150	Cond. (umhos)
				14.0	11.6	12.3	13.0	13.8	12.8	12.4	11.3	4.9	2.3	Cl^- (mg/l)
				.325	.350	.456	.447	.388	.256	.089	.044	1.33	1.78	$NO_3 + NO_2 - N$ (mg/l)
				.040	.048	.046	.033	.049	.023	.020	.036	.016	.012	$NH_4 - N$ (mg/l)
B-17				30			25	34		48		20	19	TDP (ug/l)
				86			83	93		79		84	85	Cond. (umhos)
				1.7			0.9	1.2		0.9		0.6	0.4	Cl^- (mg/l)
				.020			.027	.059		.020*		.020	.020	$NO_3 + NO_2 - N$ (mg/l)
				.121			.106	.092		.108		.106	.075	$NH_4 - N$ (mg/l)
B-18				5	6	4	13	15				11		TDP (ug/l)
				49	58	58	55	51	51	51		64	62	Cond. (umhos)
				3.2	2.2	1.6	1.8	3.4	3.3	4.7	2.7	3.8	4.0	Cl^- (mg/l)
				.020	.020	.020	.025							$NO_3 + NO_2 - N$ (mg/l)
				.034	.048	.036	.035							$NH_4 - N$ (mg/l)
B-19				4	8	3	4	8	11	9	4	9	6	TDP (ug/l)
				146	121	128	129	125	132	139		108	110	Cond. (umhos)
				5.6	9.0	8.3	8.5	7.3	7.3	7.0	5.5	5.6	5.9	Cl^- (mg/l)
				.020	.394	.020	.034	.060	.020*	.020*			.020	$NO_3 + NO_2 - N$ (mg/l)
				.064	.074	.068	.124	.063	.182	.110	.044	.063	.036	$NH_4 - N$ (mg/l)
B-20				3	6									TDP (ug/l)
				146	142	112		123		106		115	123	Cond. (umhos)
				8.6	7.4	6.5		3.6		4.0	3.5	4.0	3.6	Cl^- (mg/l)
				.020										$NO_3 + NO_2 - N$ (mg/l)
				.064	.133									$NH_4 - N$ (mg/l)
B-21						3	3	3	6	8	3	3	6	TDP (ug/l)
						71	67	63	63	56		62	61	Cond. (umhos)
						1.3	1.2	0.9	1.4	1.1	1.6	0.7	0.3	Cl^- (mg/l)
						.059	.103	.138	.171	.105	.368	-	.331	$NO_3 + NO_2 - N$ (mg/l)
						.039	.032	.010	.044	.028	.034	.046	.035	$NH_4 - N$ (mg/l)

WELL NO.	DATE 10-29-82	12-07-82	05-13-83	06-23, 24, 28-83	07-6, 7-83	07-21, 22-83	08-4, 5-83	08-18, 19, 24-83	08-31, 09-2-83	09-13, 14, 16-83	09-29, 30-83	10-17, 18	10-31, 11-12	PARAMETER
B-22				6	16								11	TDP (ug/l)
				61	69								75	Cond. (umhos)
				0.9	1.5								2.0	Cl^- (mg/l)
				.020	.370								.020	$NO_3 + NO_2 - N$ (mg/l)
				.066	.062								.077	$NH_4 - N$ (mg/l)
B-23						3	4	10	12	14	11	4	9	TDP (ug/l)
						85	65	60	65	63	63	65	82	Cond. (umhos)
						3.1	1.4	1.5	1.6	2.4	2.3	2.1	4.5	Cl^- (mg/l)
						.031	.020	.062	.020*	.020*	-	.246	.170	$NO_3 + NO_2 - N$ (mg/l)
						.066	.060	.025	.034	.212	.040	.062	.039	$NH_4 - N$ (mg/l)
B-24				7	23	21	31	29	36	31	26	8	20	TDP (ug/l)
				175	195	159	161	97	93	100	93	155	152	Cond. (umhos)
				8.9	9.0	7.8	5.2	4.2	4.5	6.0	6.2	7.8	8.5	Cl^- (mg/l)
				.020	.020	.020	.020	.062	.020*	.020*	-	.020	.020*	$NO_3 + NO_2 - N$ (mg/l)
					.258	.310	.172	.126	.132	.300	.054	.098	.096	$NH_4 - N$ (mg/l)
B-25				3	5	6	21	8	9	7	8	6	8	TDP (ug/l)
				165	137	148	158	160	147	127	126	127	134	Cond. (umhos)
				8.3	8.8	9.0	8.1	6.2	4.9	5.2	4.0	4.1	4.4	Cl^- (mg/l)
				.020	.020	.024	.020	.061	.020*	.020		.020*	.020	$NO_3 + NO_2 - N$ (mg/l)
				.052	.078	.118	.216	.026	.055	.126	.034	.055	.054	$NH_4 - N$ (mg/l)
B-26						5	4	8	22	8	6	9	7	TDP (ug/l)
						200	187	157	149	128	119	112	109	Cond. (umhos)
						5.1	2.8	3.0	3.2	4.1	3.8	3.6	4.4	Cl^- (mg/l)
						.042	.020	.062	.020*	.020*		.020*	.020	$NO_3 + NO_2 - N$ (mg/l)
						.070	.032	.074	.063	.050	.037	.076	.045	$NH_4 - N$ (mg/l)
B-27				10	13	9	32	11	12	10	6	24	8	TDP (ug/l)
				162	178	236	231	173	170	131	160	232	273	Cond. (umhos)
				9.8	9.3	10.1	9.4	8.5	8.6	8.8	8.3	5.8	8.9	Cl^- (mg/l)
				.040	.020	.020	.020	.020	.020	.020*		.020*	.020	$NO_3 + NO_2 - N$ (mg/l)
				.059	.094	.175	.295	.090	.314	.386	.108	.098	.105	$NH_4 - N$ (mg/l)

WELL NO.	DATE 10-29-82	12-07-82	05-13-83	06-23, 24, 28-83	07-6, 7-83	07-21, 22-83	08-4, 5-83	08-18, 19, 24-83	08-31, 09-2-83	09-13, 14, 16-83	09-29, 30-83	10-17, 18	10-31, 11-12	PARAMETER
B-28				3	3									TDP (ug/l)
				70	73									Cond. (umhos)
				0.9	0.9									Cl^- (mg/l)
				.052	.024									$NO_3 + NO_2 - N$ (mg/l)
				.028	.016									$NH_4 - N$ (mg/l)
B-29							8	10	15	10	7	7	8	TDP (ug/l)
							350	400	370	403	312	328	319	Cond. (umhos)
							4.5	3.5	3.1	5.7	5.9	4.5	4.1	Cl^- (mg/l)
							.020	.020	.020*	.020*		.020	.020*	$NO_3 + NO_2 - N$ (mg/l)
							2.80	3.05	4.90	3.38	3.18	2.95	3.23	$NH_4 - N$ (mg/l)
B-30							20	25		30		17	20	TDP (ug/l)
							342	280		287		250	252	Cond. (umhos)
							3.1	1.6		1.8		0.8	1.0	Cl^- (mg/l)
							.020	.020		.020*		.020*	.020*	$NO_3 + NO_2 - N$ (mg/l)
							.790	.660		.836		.573	.297	$NH_4 - N$ (mg/l)
B-31										43		38	48	TDP (ug/l)
										206		205	232	Cond. (umhos)
										1.7		0.4	0.5	Cl^- (mg/l)
										.020*		.020*	.020*	$NO_3 + NO_2 - N$ (mg/l)
										.678		.451	.291	$NH_4 - N$ (mg/l)
B-32								14		16		9	7	TDP (ug/l)
								102		93		88	90	Cond. (umhos)
								0.9		1.0		0.2	0.3	Cl^- (mg/l)
								.088		.092		.134	.156	$NO_3 + NO_2 - N$ (mg/l)
								.061		.031		.029	.056	$NH_4 - N$ (mg/l)
C-1		10	15	6		11		12		11		4	10	TDP (ug/l)
				148		168		143		161		160	152	Cond. (umhos)
		1.7	1.4	2.9		1.1		1.0		1.0		0.9	0.7	Cl^- (mg/l)
				.020		.036		.020		.020*		.020*	.020*	$NO_3 + NO_2 - N$ (mg/l)
				.058		.044		.060		.026		.042	.046	$NH_4 - N$ (mg/l)

WELL NO.	DATE 10-29-82	12-07-82	05-13-83	06-23, 24, 28-83	07-6, 7-83	07-21, 22-83	08-4, 5-83	08-18, 19, 24-83	08-31, 09-2-83	09-13, 14, 16-83	09-29, 30-83	10-17, 18	10-31, 11-12
C-2		5	4	3		3		6		4		7	3
				249		245		260		278		245	269
		22.0	24.0	19.2		18.7		20.1		20.4		19.5	20.4
				.038		.058		.080		.067		.089	.136
				.279		.008		.044		.031		.008	.029

*Interferences were noted for these nitrate and nitrite analyses

PARAMETER

TDP (ug/l)
Cond. (umhos)
Cl^- (mg/l)
$NO_3 + NO_2 - N$ (mg/l)
$NH_4 - N$ (mg/l)

Appendix E. Piezometer Location Descriptions.*

Piezometer Number	Location	Depth into Lakebed (m)	Distance from Shore (m)	Lake Depth at Time of Installation (m)
A_1	NW end. Out from well #1.	0.51	31.0	0.30
A_2	Same as A_1.	0.61	59.0	0.38
B_1	West side. South of fishing access.	0.58	22.0	0.30
B_2	Same as B_1.	0.66	31.0	0.61
C_1	West side. Out from well #2.	0.74	6.0	0.33
C_2	Same as C_1	0.61	7.5	0.61
D_1	West side. North of dead-end road.	0.64	4	0.30
D_2	Same as D_1.	0.61	6.5	0.61
E_1	West side. Opposite northern-most house on Pine Shore Road.	0.64	6.0	0.30
E_2	Same as E_1.	0.64	8.0	0.61
F_1	West side, Opposite Lantman's.	0.64	5.0	0.30
F_2	Same as F_1.	0.69	7.0	0.91

*Piezometer locations are shown in Figure 3 in the text. Generally two piezometers were installed in each area.

Appendix E (cont.)

Piezometer Number	Location	Depth into Lakebed (m)	Distance from Shore (m)	Lake Depth at Time of Installation (m)
G_1	SE end. Out from well #4,	0.64	13.0	0.30
G_2	Same as G_1.	0.64	20.5	0.61
H_1	South Bay. Off Fourtier's Beach.	0.64	8.0 27.5 from S shore 20 from E shore	0.30
H_2	Same as H_1.	0.64	16.5 from W shore	0.46
I_1	South end. Opposite gravel area.	0.46	10.0	0.30
J_1	SE end. Opposite green camp up on	0.66	7.0	0.38
J_2	Same as J_1.	0.61	10.0	0.61
K_1	East side. South of boathouse by Brown camp with white trim.	0.61	5.5	0.30
K_2	Same as K_1.	0.69	7.5	0.61
L_1	East side. North of yellow house with lattice around bottom.	0.61	6.0	0.36
L_2	Same as L_1.	0.76	8.0	0.91

Appendix E (cont.)

Piezometer Number	Location	Depth into Lakebed (m)	Distance from Shore (m)	Lake Depth at Time of Installation (m)
M_1	East side. By small white boat-house.	0.66	8.5	0.30
M_2	Same as M_1.	0.66	11.0	0.61
N_1	East side. In the center of east bay.	0.76	--	0.30
N_2	Same as N_1.	1.42	--	0.30
O_1	East side. Off point opposite island.	0.66	11.5	0.30
O_2	Same as O_1.	0.66	15.0	0.61
P_1	East side. South of Hall's.	0.61	6.0	0.33
P_2	Same as P_1.	0.69	8.0	0.61
Q_1	East side. Along wooded area.	0.69	5.5	0.30
Q_2	Same as Q_1.	0.74	7.0	0.61
R_1	East side. Small bay with small brown camp. Most north piezometer.	0.61	9.0	0.30
R_2	About 10 m south of R_1.	0.64	10.0	0.30

Appendix E (cont.)

Piezometer Number	Location	Depth into Lakebed (m)	Distance from Shore (m)	Lake Depth at Time of Installation (m)
R_3	About 10 m south of R_2.	0.66	10.5	0.30
R_4	About 10 m south of R_3.	0.61	8.5	0.30
S_1	East side. Opposite yellow camp out from well #9.	0.66	10.0	0.30
S_2	Same as S_1.	0.61	13.5	0.61
T_1	North End. Out from well #10.	0.81	33.0	0.30
T_2	Same as T_1.	0.58	52.0	0.61
U_1	In north bay. Midway between fishing access and municipal beach.	1.60	--	1.30
Q_1	East side. Along wooded area.	0.69	5.5	0.30
Q_2	Same as Q_1.	0.74	7.0	0.61
R_1	East side. Small bay with small brown camp. Most north piezometer.	0.61	9.0	0.30
R_2	About 10 m south of R_1.	0.64	10.0	0.30

Appendix F. Piezometer Head Measurements.

Piezometer Number	Head above Lake Surface (mm) 11/2/82	11/2/82
A_1	16	5
A_2	1	3
B_1	42	47
B_2	14	25
C_1	1	1
C_2	1	0*
D_1	5	8
D_2	5	8
E_1	8	1
E_2	1	-1*
F_1	3	1
F_2	0	0
G_1	13	13
G_2	0	3
H_1	205	181
H_2	9	6
I_1	-2	-3
J_1	-1	0
J_2	-1	-3
K_1	0	-9
K_2	1	-3

Piezometer Number	Head above Lake Surface (mm) 11/2/82	11/22/82
L_1	-2	-1
L_2	-2	-2
M_1	1	0
M_2	1	0
N_1	1	-1
N_2	1	-1
O_1	5*	1
O_2	-1	-1
P_1	0	2
P_2	2	1
Q_1	1	1*
Q_2	0	0*
R_1	0	-1
R_2	1	-4
R_3	-3	-1*
R_4	0	-1
S_1	1	2
S_2	-1	1
T_1	0	-2
T_2	-1	0
U_1	-12	-2

*results suspect

Appendix G. Seepage Meter Location Descriptions.*

#	Bottom Type	Depth (m)	Distance from Shore (m)	Location
1	Sand with mixed gravel. Moderate organic material at surface.	0.34	4.9	West shore. South of fishing access and north of stream (W-2).
2	Gravel with rock	1.28	3.0	West shore. Along stretch with open fields.
3	Silt, sand, gravel and rock with some low growing weeds.	2.13	8.5	West shore. Along stretch with open fields.
4	Sand and fine gravel with overlying silt layer.	0.46	1.8	West shore. Just south of road that dead ends at lake.
5	Silt over mixed sand gravel and rock.	0.55	3.4	West shore. Just south of stream (W-4) entrance.
6	Sand and fine gravel with overlying silt.	0.55	3.0	West shore. Opposite Marlin residence.
7	Sand with rock.	0.64	2.3	West shore. Opposite house with red chimney.
8	Sand with mixed gravel. Some overlying silt.	0.37	4.9	Southwest end. Just south of stream (W-9) entrance.
9	Sand with gravel and small rocks.	0.61	3.7	South end. Opposite elm tree to west of white house.
10	Mixed gravel.	0.70	7.3	South end. Opposite gravel beach with cement retaining wall.

*Seepage meter locations are shown in Figure 2 in the text.

Appendix G (cont .).

#	Bottom Type	Depth (m)	Distance from Shore (m)	Location
11	Gravel and rock.	2.13	8.5	South end. Opposite west side of point near dam.
12	Gravel with rock	0.70	9.1	South end. Opposite west side of point near dam.
13	Sand, gravel and rock. (Atypical site; most is rocky with weeds.)	1.98	10.7	South end. In small bay with dam. Opposite point that defines west bank of bay.
14	Rock with gravel.	1.00	2.4	South end. In small bay with dam. Opposite west bank.
15	Sand with rock.	1.28	7.9	South end. In small bay with dam. Opposite west bank.
16	Sand with fine gravel and rock.	0.79	4.6	South end by dam. Between spillway and west bank.
17	Sand, gravel and rock.	1.68	9.4	South end. In small bay with dam. Opposite spillway.
18	Sand and gravel.	0.76	4.3	South end. In small bay near dam. Just east of concrete apron of dam.
19	Sand, gravel and rock.	1.25	8.2	South end. In small bay near dam. East of concrete apron of dam.
20		0.70	2.0	South end near dam. Opposite green camp on east bank of small bay.
21	Rock and gravel.	1.22	6.4	Southeast shore. Opposite ledge with many overhanging birches.

Meters 13-21 are shown in the inset on Figure 2.

Appendix G (contd.).

#	Bottom Type	Depth (m)	Distance from Shore (m)	Location
22	Fine gravel.	0.70	4.3	Southeast end. North of red house built over water.
23	Mixed gravel.	0.64	1.8	East shore. Between dark red and dark green camps.
24	Mixed gravel with rock.	0.79	2.9	East shore. North side of white house with stone foundation.
25	Mixed gravel.	0.52	3.7	East shore. By the jetty sticking into the bay next to stream (E-11).
26	Fine gravel with rock.	0.79	3.0	East shore. Along wooded, undeveloped area.
27	Medium gravel with some rock.	0.40	3.7	Small bay on east shore. North of brown camp with white trim.
28	Fine gravel with rock.	0.34	1.2	East shore. South of light brown camp with deck and white trim.
29	Gravel with rock.	0.46	3.0	East shore. South of light brown camp with deck and white trim.
30	Gravel with rock.	0.82	6.7	East shore. South of light brown camp with deck and white trim.
31	Light layer of loose	1.49	14.0	East shore. South of light brown camp with deck and white trim.
32	About 5" of loose silty material grading into sand and fine gravel.	2.01	20.4	East shore. South of light brown camp with deck and white trim.

Appendix G (contd.).

#	Bottom Type	Depth (m)	Distance from Shore (m)	Location
33	Gravel with rock.	0.46	3.0	East shore. South of light brown camp with deck and white trim. North of meter #29.
34	Gravel with rock.	0.88	6.7	East shore. South of light brown camp with deck and white trim. North of meter #30.
35	2" of fine material over mixed rock, gravel and sand.	1.80	14.3	East shore. South of light brown camp with deck and white trim. North of meter #31.
36	Mixed gravel. Up to fist size gravel and sand.	0.55	4.3	East shore. Between yellow camp with white trim and light brown camp with deck and white trim.
37	Gravel with rock.	0.88	7.0	East shore. Between yellow camp with white trim and light brown camp with deck and white trim.
38	About 2" of loose material over rock and gravel.	1.49	13.1	East shore. Between yellow camp with white trim and light brown camp with deck and white trim.
39	Medium gravel to sand.	0.46	5.5	Next to first dock east of municipal beach.
40	Sand and fine gravel with some organic material in surface layer. Light covering of low growing Elecharis spp.	0.49	8.2	North end. Opposite east bank of the mouth of W-O.
41	Sand with small gravel and some organic material at the surface.	0.49	7.3	North end. Opposite east bank of the mouth of W-O.

Appendix G (contd.).

#	Bottom Type	Depth (m)	Distance from Shore (m)	Location
42	Sand with small gravel and some organic material at the surface. Some *Elecharis* spp. present.	0.55	13.1	North end. Opposite east bank of the mouth of W-O.
43	Sand with fine gravel.	0.46	7.3	North end. Adjacent to willow on the west bank of the mouth of W-O.
44	Sand with fine gravel and organic material.	0.46	4.3	Northwest end. North of access by large patch of *Nuphar variegatum*.

Appendix H. Macrophyte Survey Site Data.

Site #	Sample	Zone Depth (m)	Distance of Zone from Shore (m)	Species Present - Relative Abundance*	Bottom Type	Biomass Sample Depth (m)	Grams Dry Weight per m^2
1	A	--	--	Potamogeton amplifolius - 3 Vallisneria americana - 3 Potamogeton zosteriformis - 3 Elodea Nuttallii - 3 Myriophyllum exalbescens - 3 Heteranthera dubia - 3 Potamogeton Richardsonii - 2	Muck	1.2	199.12
	B	--	--		Muck	1.2	120.27
1-2	A	--	--		Sand and Gravel	0.8	59.46
	B	--	--		Sand and Gravel	0.8	74.68
2	A	--	--	P. amplifolius - 3 V. americana - 3 H. dubia - 3 P. zosteriformis - 2 E. Nuttallii - 2 M. exalbescens - 2 P. Richardsonii - 2 Potamogeton gramineus - 2	Muck	0.9	104.11
	B	--	--		Muck	0.9	178.11
2-3	A	--	--		Muck	0.9	87.54
	B	--	--		Muck	0.9	64.65
3	A	--	--	Potamogeton praelongus - 4 V. americana - 3 P. amplifolius - 3 P. zosteriformis - 2 M. exalbescens - 2 H. dubia - 2 E. Nuttallii - 2	Muck	1.1	85.79
	B	--	--		Muck	1.1	75.62

* 4-abundant, 3-common, 2-sparse, 1-extremely sparse

Site #	Sample	Zone Depth (m)	Distance of Zone from Shore (m)	Species Present - Relative Abundance*	Bottom Type	Biomass Sample Depth (m)	Grams Dry Weight per m^2
3-4	A	--	--		Muck	0.9	102.49
	B	--	--		Muck	0.9	136.16
4	A	--	--	Polygonum natans - 4 V. americana - 3 P. Richardsonii - 2 Sagittaria rigida - 2 Chara/Nitella spp. - 2 Potamogeton crispus - 1 Eleocharis sp. - 1 Najas spp. - 1	Sand	0.5	31.04
	B	--	--		Sand	0.5	165.19
4-5	A	--	--		Muck	0.9	93.74
	B	--	--		Muck	0.9	54.95
5	A	--	--	P. zosteriformis - 4 P. praelongus - 2 E. Nuttallii - 2	Muck	1.0	59.39
	B	--	--		Muck	1.0	117.64
5-6	A	--	--		Muck	1.2	85.19
	B	--	--		Muck	1.2	183.57
6	A	--	--	P. zosteriformis - 3 E. Nuttallii - 3 P. praelongus - 2 M. exalbescens - 2 H. dubia - 2 V. americana - 2 Chara/Nitella spp. - 2	Muck	1.7	110.77
	B	--	--		Muck	1.7	131.99
6-7	A	--	--		Muck	1.7	45.12
	B	--	--		Muck	1.7	107.68

Site #	Sample	Zone Depth (m)	Distance of Zone from Shore (m)	Species Present - Relative Abundance*	Bottom Type	Biomass Sample Depth (m)	Grams Dry Weight per m^2
7	A	--	--	E. Nuttallii - 4 P. praelongus - 3 P. zosteriformis - 2 M. exalbescens - 1 P. crispus - 1 Ranunculus subrigidus - 1	Muck	1.8	72.73
	B	--	--		Muck	1.8	140.74
7-8	A	--	--		Muck	1.9	205.05
	B	--	--		Muck	1.9	189.90
8	A	--	--	P. praelongus - 3 E. Nuttallii - 3	Muck	1.9	165.66
	B	--	--		Muck	1.9	237.91
8-9	A	--	--		Muck	1.8	141.75
	B	--	--		Muck	1.8	154.21
9	A	--	--	P. praelongus - 3 M. exalbescens - 2 P. zosteriformis - 1 E. Nuttallii - 1	Muck	1.4	156.90
	B	--	--		Muck	1.4	144.71
9-1	A	--	--		Muck	1.6	125.59
	B	--	--		Muck	1.6	90.91
1-10	A	--	--		Muck	1.4	121.55
	B	--	--		Muck	1.4	71.04
10	A	--	--	P. praelongus - 3 E. Nuttallii - 2 M. exalbescens - 1 P. zosteriformis - 1 R. subrigidus - 1	Muck	1.8	131.31
	B	--	--		Muck	1.8	216.50

Site #	Sample	Zone Depth (m)	Distance of Zone from Shore (m)	Species Present - Relative Abundance*	Bottom Type	Biomass Sample Depth (m)	Grams Dry Weight per m^2
10-11	A	--	--		Muck	1.8	108.08
	B	--	--		Muck	1.8	155.56
11	A	--	--	P. praelongus - 3 E. Nuttallii - 3 H. dubia - 2 P. zosteriformis - 2 M. exalbescens - 1	Muck	1.8	198.79
	B	--	--		Muck	1.8	148.89
11-12	A	--	--		Muck	2.1	176.43
	B	--	--		Muck	2.1	120.27
12	A	--	--	P. praelongus - 2 E. Nuttallii - 2 P. zosteriformis - 2 H. dubia - 1	Muck	2.2	113.47
	B	--	--		Muck	2.2	128.62
12-13	A	--	--		Muck	2.1	112.46
	B	--	--		Muck	2.1	99.66
13	A	--	--	P. amplifolius - 3 H. dubia - 3 Chara/Nitella spp. - 2 M. exalbescens - 1 P. praelongus - 1 Potamogeton Spirillus - 1 Najas spp. - 1 P. Richardsonii - 1	Shallow Muck over Rock	1.5	32.32
	B	--	--		Shallow Muck over Rock	1.5	42.42
13-14	A	--	--		Muck over Sand	1.5	58.59
	B	--	--		Muck over Sand	1.5	59.66

Site #	Sample	Zone Depth (m)	Distance of Zone from Shore (m)	Species Present - Relative Abundance*	Bottom Type	Biomass Sample Depth (m)	Grams Dry Weight per m^2
14	A	--	--	*Najas* spp. - 2 *V. americana* - 2 *P. amplifolius* - 2 *P. Spirillus* - 1 *Chara/Nitella* spp. - 1 *P. gramineus* - 1 *P. zosteriformis* - 1 *P. crispus* - 1	Sand	1.6	39.06
	B	--	--		Sand	1.6	52.85
14-15	A	--	--		Muck	2.2	106.40
	B	--	--		Muck	2.2	77.44
15	A	--	--	*Najas* spp. - 1 *Chara/Nitella* spp. - 1	Muck	2.9	11.45
	B	--	--		Muck	2.9	2.02
15-16	A	--	--		Muck	3.2	11.45
	B	--	--		Muck	3.2	6.40
16	A	--	--	*P. crispus* - 1	Muck	3.2	0.00
	B	--	--		Muck	3.2	32.32
16-17	A	--	--		Muck	2.2	101.68
	B	--	--		Muck	2.2	105.05
17	A	--	--	*P. zosteriformis* - 4 *E. Nuttallii* - 3 *P. crispus* - 2 *M. exalbescens* - 2 *Nymphaea odorata* - 1	Muck	1.3	105.72
	B	--	--		Muck	1.3	108.42
17-18	A	--	--		Muck	1.6	273.27
	B	--	--		Muck	1.6	227.27

Site #	Sample	Zone Depth (m)	Distance of Zone from Shore (m)	Species Present - Relative Abundance*	Bottom Type	Biomass Sample Depth (m)	Grams Dry Weight per m^2
18	A	--	--		Muck	2.5	23.57
	B	--	--		Muck	2.5	60.94
18-19	A	--	--		Muck	2.8	0.00
	B	--	--		Muck	2.8	2.02
19	A	--	--		Muck	3.1	6.06
	B	--	--		Muck	3.1	12.12
19-20	A	--	--		Muck	3.0	75.08
	B	--	--		Muck	3.0	126.60
20	A	--	--		Muck	2.2	101.01
	B	--	--		Muck	2.2	55.56
20-25	A	--	--		Muck	1.8	62.63
	B	--	--		Muck	1.8	48.55
21	A	--	--	*P. zosteriformis* - 4 *Chara/Nitella* spp. - 3 *P. amplifolius* - 2 *V. americana* - 1 *H. dubia* - 1 *P. Richardsonii* - 1 *M. exalbescens* - 1	Muck	0.9	108.42
	B	--	--		Muck	0.9	129.97
21-22	A	--	--		Muck	2.6	17.51
	B	--	--		Muck	2.6	missing
22	A	--	--	*E. Nuttallii* - 1	Muck	3.0	9.09
	B	--	--		Muck	3.0	4.04
22-23	A	--	--		Muck	3.0	21.39
	B	--	--		Muck	3.0	38.72

Site #	Sample	Zone Depth (m)	Distance of Zone from Shore (m)	Species Present - Relative Abundance*	Bottom Type	Biomass Sample Depth (m)	Grams Dry Weight per m^2
23	A	--	--	P. praelongus - 2	Muck	2.9	12.79
	B	--	--		Muck	2.9	76.43
23-24	A	--	--		Muck	2.8	68.35
	B	--	--		Muck	2.8	71.31
24	A	--	--	P. praelongus - 3 H. dubia - 2 E. Nuttallii - 2 M. exalbescens - 2 Najas spp. - 1 P. gramineus - 1 Potamogeton Vaseyi - 1 Chara/Nitella spp. - 1	Muck	2.3	108.42
	B	--	--		Muck	2.3	144.44
24-25	A	--	--		Silty Sand	1.9	54.55
	B	--	--		Silty Sand	1.9	92.26
25	A	--	--	P. praelongus - 3 P. zosteriformis - 3 H. dubia - 2 M. exalbescens - 1 Najas spp. - 1 Chara/Nitella spp. - 1	Muck	1.7	60.27
	B	--	--		Muck	1.7	60.94
26	A	--	--	Chara/Nitella spp. - 2 H. dubia - 1 P. praelongus - 1 Najas spp. - 1	Rock and Gravel	1.3	15.15
	B	--	--		Rock and Gravel	1.3	31.99
26-27	A	--	--		Muck	2.9	25.59
	B	--	--		Muck	2.9	37.71

Site #	Sample	Zone Depth (m)	Distance of Zone from Shore (m)	Species Present - Relative Abundance*	Bottom Type	Biomass Sample Depth (m)	Grams Dry Weight per m^2
27	A	--	--	Chara/Nitella spp. - 2 E. Nuttallii - 1	Muck	3.3	8.42
	B	--	--		Muck	3.3	2.02
27-28	A	--	--		Muck	3.4	3.70
	B	--	--		Muck	3.4	26.94
28	A	--	--	E. Nuttallii - 1	Muck	4.1	6.60
	B	--	--		Muck	4.1	0.00
28-29	A	--	--		Muck	5.0	0.00
	B	--	--		Muck	5.0	0.00
29	A	--	--	None	Muck	5.6	0.00
	B	--	--	None	Muck	5.6	0.00
29-30	A	--	--		Muck	3.8	6.06
	B	--	--		Muck	3.8	0.67
30	A	--	--	H. dubia - 3 P. Vaseyi - 2 M. exalbescens - 1 Najas spp. - 1 P. zosteriformis - 1	Muck	1.9	68.01
	B	--	--		Muck	1.9	124.92
30-31	A				Rock and Gravel	1.5	39.73
	B				Sand	2.7	31.99
31	A	0-2.3	3-14	H. dubia - 3 Najas spp. - 2 Chara/Nitella spp. - 2 E. Nuttallii - 1 M. exalbescens - 1 P. Spirillus - 1 P. Vaseyi - 1	Sand	1.5	77.91

Site #	Sample	Zone Depth (m)	Distance of Zone from Shore (m)	Species Present - Relative Abundance*	Bottom Type	Biomass Sample Depth (m)	Grams Dry Weight per m^2
31	B	2.3-3.7	14.36	Chara/Nitella spp. - 3 E. Nuttallii - 2 Najas spp. - 1 P. zosteriformis - 1 P. Spirillus - 1 P. Vaseyi - 1	Gravel	2.7	51.18
31-32	A				Muck		205.59
	B				Sand	3.4	48.62
32	A	0-1.5	2.17	P. Richardsonii - 3 P. zosteriformis - 3 H. dubia - 3 V. americana - 2 E. Nuttallii - 2 Najas spp. - 1	Muck over Rock	0.9	106.20
	B	1.5-3.4	17.30	E. Nuttallii - 4 Chara/Nitella spp. - 2 P. zosteriformis - 1	Sand and Gravel	2.4	52.53
32-33	A				Muck	1.2	270.37
	B				Sand	3.0	19.53
33	A	0-2.1	2-21	E. Nuttallii - 4 P. praelongus - 3 H. dubia - 3 P. zosteriformis - 2 V. americana - 2 M. exalbescens - 2 P. Richardsonii - 2 P. gramineus - 2	Muck	0.9	270.03
	B	2.1-3.7	21-37	E. Nuttallii - 4	Sand	3.0	79.80
33-34	A				Gravel and Sand	1.8	49.70
	B				Gravel and Sand	3.0	20.88

Site #	Sample	Zone Depth (m)	Distance of Zone from Shore (m)	Species Present - Relative Abundance*	Bottom Type	Biomass Sample Depth (m)	Grams Dry Weight per m^2
34	A	0-1.8	3-14	H. dubia - 4 Chara/Nitella spp. - 2 Najas spp. - 2 P. Vaseyi - 2 P. praelongus - 1 P. Spirillus - 1	Gravel and Rock	0.8	28.48
	B	1.8-3.2	14-37	E. Nuttallii - 3 Chara/Nitella spp. - 3 H. dubia - 1 P. Vaseyi - 1	Sand and Gravel	3.0	16.84
34-35	A				Gravel and Rock	2.2	46.67
	B				Sand and Silt	3.5	17.85
35	A	0-1.7	2-22	Chara/Nitella spp. - 3 Najas spp. - 3 H. dubia - 3 P. Spirillus - 2 P. Vaseyi - 2 P. gramineus - 2 P. Richardsonii - 2	Gravel	0.8	63.97
	B	1.7-3.5	22-26	E. Nuttallii - 2 Najas spp. - 2	Muck	3.1	26.40
35-36	A				Gravel	2.2	116.84
	B				Muck over Gravel	2.5	62.29
36	A	0-4	3-16	E. Nuttallii - 3 H. dubia - 2 P. zosteriformis - 2 P. praelongus - 1 P. Vaseyi - 1	Rock and Gravel	1.7	70.91
36-37	A				Muck over Gravel	1.8	141.08
	B				Sand	3.5	9.76

Site #	Sample	Zone Depth (m)	Distance of Zone from Shore (m)	Species Present - Relative Abundance*	Bottom Type	Biomass Sample Depth (m)	Grams Dry Weight per m^2
37	A	0-3.6	3-40	H. dubia - 3 P. praelongus - 3 E. Nuttallii - 2 P. amplifolius - 2 Eleocharis sp. - 1 Isoetes Braunii - 1 V. americana - 1 P. gramineus - 1 Najas spp. - 1 P. Vaseyi - 1 P. Spirillus - 1 M. exalbescens - 1 Chara/Nitella spp. - 1	Gravel	1.1	101.01
	B	3.6-5	40-45	E. Nuttallii - 2 Chara/Nitella spp. - 2	Muck	3.2	39.73
37-38	A				Muck over Gravel	1.8	112.46
	B				Muck over Gravel	2.4	86.87
38	A	0-3.7	2-15	P. praelongus - 3 H. dubia - 3 Najas spp. - 2 E. Nuttallii - 1 P. Spirillus - 1 P. Richardsonii - 1 Chara/Nitella spp. - 1 P. zosteriformis - 1 R. subrigidus - 1 P. Vaseyi - 1 M. exalbescens - 1 V. americana - 1 S. rigida - 1 Eleocharis sp. - 1 I. Braunii - 1	Muck over Rock	1.4	52.53

Site #	Sample	Zone Depth (m)	Distance of Zone from Shore (m)	Species Present - Relative Abundance*	Bottom Type	Biomass Sample Depth (m)	Grams Dry Weight per m^2
38-39	A				Muck	1.1	81.48
39	A	0-3.7	2-38	H. dubia - 2 Najas spp. - 2 P. praelongus - 2 P. Spirillus - 1 P. gramineus - 1 Eleocharis sp. - 1 I. Braunii - 1 M. exalbescens - 1 P. Vaseyi - 1 P. zosteriformis - 1	Gravel	0.9	10.77
39-40	A				Gravel and Rock	1.7	51.52
	B				Muck	3.3	20.54
40	A	0-2.4	0-17	H. dubia - 3 Najas spp. - 2 P. amplifolius - 2 E. Nuttallii - 1 V. americana - 1 P. gramineus - 1 P. Spirillus - 1	Gravel and Rock	2.0	31.92
	B	2.4-4.9	17-29	P. Vaseyi - 3 E. Nuttallii - 2 Najas spp. - 2 Chara/Nitella spp. - 1 H. dubia - 1	Muck	4.0	18.38
40-41	A				Gravel	1.5	45.45
41	A	0-2.7	0-26	P. praelongus - 3 H. dubia - 2 E. Nuttallii - 2 Najas spp. - 2 V. americana - 1 P. gramineus - 1 P. Spirillus - 1 P. Vaseyi - 1	Sand and Gravel	1.4	6.06

Site #	Sample	Zone Depth (m)	Distance of Zone from Shore (m)	Species Present - Relative Abundance*	Bottom Type	Biomass Sample Depth (m)	Grams Dry Weight per m^2
41	B	2.7-3.7	26-40	Najas spp. - 2 Chara/Nitella spp. - 2 P. Spirillus - 1 P. Vaseyi - 1 P. zosteriformis - 1 P. praelongus - 1	Sand and Gravel	2.5	30.64
41-42	A				Gravel	2.0	36.70
	B				Gravel	2.6	16.84
42	A	0-2.4	0-25	H. dubia - 3 P. praelongus - 3 P. zosteriformis - 3 P. amplifolius - 2 Najas spp. - 2 V. americana - 2 P. gramineus - 1 Chara/Nitella spp. - 1 P. Spirillus - 1 P. Richardsonii - 1	Muck over Gravel	0.8	38.45
	B	2.4-4.0	25-42	E. Nuttallii - 2 Chara/Nitella spp. - 2 Najas spp. - 2 H. dubia - 2	Muck	3.0	17.51
42-43	A				Gravel	1.7	20.20
43	A	0-2.7	0-13	P. Spirillus - 3 H. dubia - 2 Najas spp. - 2 V. americana - 2 E. Nuttallii - 1 P. Vaseyi - 1 P. gramineus - 1	Gravel	1.0	38.38
43-44	A				Gravel	1.6	39.06

Site #	Sample	Zone Depth (m)	Distance of Zone from Shore (m)	Species Present - Relative Abundance*	Bottom Type	Biomass Sample Depth (m)	Grams Dry Weight per m^2
44	A	0-2.7	0-10	Najas spp. - 3 P. Spirillus - 3 H. dubia - 2 E. Nuttallii - 1 P. zosteriformis - 1 V. americana - 1 P. Vaseyi - 1 Chara/Nitella spp. - 1	Gravel	1.2	12.79
44-45	A				Sand and Gravel	1.9	40.40
	B				Sand and Gravel	3.5	30.98
45	A	0-2.0	0-15	P. zosteriformis - 3 Najas spp. - 2 H. dubia - 2 E. Nuttallii - 2 Chara/Nitella spp. - 2 P. gramineus - 1 I. Braunii - 1 Eleocharis sp. - 1 P. Spirillus - 1 P. Vaseyi - 1	Sand and Gravel	0.9	134.68
	B	2.0-3.8	15-20	Chara/Nitella spp. - 2 E. Nuttallii - 2 Najas spp. - 2 H. dubia - 1 P. Spirillus - 1	Sand and Gravel	3.0	14.81
49-46	A				Gravel	2.0	89.49
	B				Gravel	2.4	35.69
46	A	0-2.7	0-17	Najas spp. - 2 P. Spirillus - 2 H. dubia - 2 E. Nuttallii - 1 P. Vaseyi - 1 Chara/Nitella spp - 1 P. amplifolius - 1 P. gramineus - 1	Rock and Gravel	1.0	6.73

Site #	Sample	Zone Depth (m)	Distance of Zone from Shore (m)	Species Present - Relative Abundance*	Bottom Type	Biomass Sample Depth (m)	Grams Dry Weight per m^2
46-47	A				Rock and Gravel	1.4	80.00
47	A	0-3.0	0-13	*Najas* spp. - 3 *P. Spirillus* - 3 *V. americana* - 2 *P. gramineus* - 2 *H. dubia* - 2 *P. zosteriformis* - 1 *E. Nuttallii* - 1 *Chara/Nitella* spp. - 1	Sand and Gravel	1.2	31.72
47-48	A				Sand and Gravel	1.9	17.85
48	A	0-2.5	0-12	*H. dubia* - 3 *Najas* spp. - 2 *E. Nuttallii* - 2 *Chara/Nitella* spp. - 2 *P. Spirillus* - 1 *P. Vaseyi* - 1 *P. gramineus* - 1 *P. praelongus* - 1 *P. zosteriformis* - 1 *M. exalbescens* - 1	Rock and Gravel	0.9	13.47
48-26	A				Muck	2.5	126.60
	B				Muck	3.1	10.64
49	A	--	--		Muck	3.0	31.65
	B	--	--		Muck	3.0	6.06
49-50	A	--	--		Muck	4.0	5.39
	B	--	--		Muck	4.0	0.00
50	A	--	--		Sand and Gravel	2.9	8.08
	B	--	--		Muck	3.1	5.39

Site #	Sample	Zone Depth (m)	Distance of Zone from Shore (m)	Species Present - Relative Abundance*	Bottom Type	Biomass Sample Depth (m)	Grams Dry Weight per m^2
51	A	--	--	P. praelongus - 2 E. Nuttallii - 2 Chara/Nitella spp. - 2 P. Vaseyi - 1 Najas spp. - 1	Muck	3.0	24.92
	B	--	--		Muck	3.0	14.14
52	A	--	--	E. Nuttallii - 2 Chara/Nitella spp. - 2 P. praelongus - 1 P. zosteriformis - 1 Najas spp. - 1	Rock and Gravel	0.8	141.08
	B	--	--		Rock and Gravel	0.8	37.37
53	A	--	--	P. praelongus - 3 E. Nuttallii - 2 H. dubia - 1 Chara/Nitella spp. - 1	Muck	3.1	7.68
	B	--	--		Muck	3.1	22.49
54	A	--	--	P. praelongus - 3 Najas spp. - 1 E. Nuttallii - 1 P. Spirillus - 1 M. exalbescens - 1	Muck	1.8	78.79
	B	--	--		Muck	1.8	54.88
55	A	--	--	P. praelongus - 4	Muck	1.4	201.35
	B	--	--		Muck	1.4	105.05
55-shore	A	--	--		Muck	1.0	121.89
	B	--	--		Muck	1.0	152.86

Site #	Sample	Zone Depth (m)	Distance of Zone from Shore (m)	Species Present - Relative Abundance*	Bottom Type	Biomass Sample Depth (m)	Grams Dry Weight per m^2
56	A	--	--	E. Nuttallii - 4 P. amplifolius - 3 H. dubia - 1 M. exalbescens - 1 P. Spirillus - 1 P. zosteriformis - 1	Gravel	0.9	102.69
	B	--	--		Rock and Gravel	1.1	81.48
56-Island	A	--	--		Muck	2.3	30.64
	B	--	--		Muck	2.3	16.50
57	A	--	--	None			
	B	--	--				
58	A	--	--	None			
	B	--	--				
59	A	--	--	P. zosteriformis - 4 P. praelongus - 3 P. amplifolius - 3 E. Nuttallii - 3 Chara/Nitella spp. - 3 H. dubia - 3 S. rigida - 2 Najas spp. - 2 M. exalbescens - 1 P. Richardsonii - 1 I. Braunii - 1 Eleocharis sp. - 1 P. Spirillus - 1 P. gramineus - 1 R. subrigidus - 1 P. crispus - 1	Muck	0.4	263.84
	B	--	--		Muck	0.4	63.64

Site #	Sample	Zone Depth (m)	Distance of Zone from Shore (m)	Species Present - Relative Abundance*	Bottom Type	Biomass Sample Depth (m)	Grams Dry Weight per m^2
60		--	--	N. odorata - 4 Chara/Nitella spp. - 4 P. amplifolius - 3 P. zosteriformis - 2 H. dubia - 1 P. Spirillus - 1 E. Nuttallii - 1 V. americana - 1 P. Vaseyi - 1 Najas spp. - 1	Not Sampled	Not Sampled	Not Sampled
61		--	--	H. dubia - 3 P. zosteriformis - 3 E. Nuttallii - 3 V. americana - 2 Najas spp. - 2 S. rigida - 1 P. Spirillus - 1 P. Richardsonii - 1 R. subrigida - 1 M. exalbescens - 1 P. praelongus - 1 P. Vaseyi - 1	Not Sampled	Not Sampled	Not Sampled
62 (submerged island)		--	--	E. Nuttallii - 3 P. gramineus - 3 H. dubia - 2 M. exalbescens - 2 R. subrigida - 1 P. zosteriformis - 1 P. Vaseyi - 1 Chara/Nitella spp. - 1 Najas spp. - 1	Not Sampled	Not Sampled	Not Sampled

Appendix I. Comparison of Outflow Phosphorus Concentrations with Lake Surface Water Phosphorus Concentrations.

Date	Outlet Flow (cfs)	Total Phosphorus Concentration (ug/l) Outflow	Lake (0m)	Lake (1m)	Date	Outlet Flow (cfs)	Total Phosphorus Concentration (ug/l) Outflow	Lake (0m)	Lake (1m)
03-16	2.90	8	10	11	03-15	9.43	21	14	21
03-30	8.27	11	20	13	04-15	11.20	23	23	23
04-23	8.90	17	29	29	04-21	15.60	21	23	24
05-04	3.08	21	27	27	04-29	33.85	23	21	24
05-10	3.70	26	25	26	05-05	19.50	20	23	24
05-17	3.44	19	21	21	05-12	19.47	19	19	22
05-24	3.20	18	21	19	05-19	7.71	25	25	24
06-02	3.20	14	13	24	05-27	33.66	21	21	27
06-08	3.50	19	16	19	06-15	4.20	13	10	13
06-14	3.20	21	21	24	06-22	1.57	21	11	12
06-28	3.30	13	11	15	06-29	1.08	8	9	10
07-06	3.80	24	14	12	07-05	0.74	14	14	17
07-12	3.30	18	12	11	07-19	1.35	14	11	11
07-20	3.40	14	18	13	07-28	1.34	13	15	17
07-27	2.88	15	14	14	08-09	1.23	15	26	24
08-01	3.40	15	14	14	08-23	1.23	17	23	19
08-09	2.77	15	19	19	08-31	1.32	14	16	16
08-20	2.80	11	11	11	09-07	1.32	18	23	18
08-24	2.70	8	9	10	09-14	1.23	18	18	20
09-02	1.60	12	14	14	09-20	1.15	20	18	20
10-21	0.27	21	25	28	09-28	1.23	17	20	20
10-27	0.26	21	24	25	10-05	1.15	20	23	24
11-04	0.28	25	26	28	10-12	1.11	19	27	26
11-10	0.14	18	23	49	10-19	1.07	23	29	32
11-17	0.24	21	23	22	10-27	0.99	25	30	28
12-08	0.17	13	21	20	11-02	0.99	27	26	28
					11-09	3.28	23	27	30
					11-16	10.66	24	22	24
					11-29	16.97	18	19	21
					12-29	13.43	16	23	20
Flow weighted Means		16.1	18.2	18.4			20.5	20.9	23.1

Appendix I. (cont.)

Standard deviation of flow-weighted means was determined by:

$$S.D. = s_i 2 \frac{q_i 2}{q_t}$$

where: $s_i 2$ = variance of phosphorus sample analysis; estimated as $(5\ ug/l)^2$

q_i = outlet flow associated with each sample

q_t = total outlet flow for all sample dates

S.D. = standard deviation

Ninety-five percent confidence intervals for mean outflow concentrations were approximated as 1 two standard deviations. Confidence intervals for the 1982 and 1983 outflow means are 16.1 ± 2.4 and 20.5 ± 3.0. At this confidence level outflow means calculated from lake samples are not significantly different from those based on actual outflow samples.

Appendix J. Lake Iroquois Sediment Core Data.

Depth (cm)	Date (yrs)	Sediment Accumulation Rate ($g/cm^2/yr$)	Organic Accumulation Rate ($g/cm^2/yr$)	Inorganic Accumulation Rate ($g/cm^2/yr$)	Percent Organic (%)	Percent Inorganic (%)
2	1981	0.0389	0.0112	0.0240	28.7	61.8
4	1978	0.0392	0.0114	0.0247	29.2	62.9
6	1973	0.0266	0.0076	0.0175	28.7	65.8
8	1967	0.0311	0.0087	0.0203	27.9	65.2
10	1955	0.0140	0.0039	0.0092	28.0	65.4
12	1942	0.0135	0.0038	0.0091	27.8	67.4
14	1933	0.0282	0.0079	0.0192	28.2	68.1
16	1926	0.0267	0.0079	0.0176	29.4	65.9
18	1921	0.0231	0.0073	0.0140	31.6	60.4
20	1916	0.0364	0.0121	0.0221	33.2	60.6
22	1911	0.0396	0.0135	0.0238	34.3	60.1
24	1908	0.0340	0.0122	0.0195	36.0	57.4
26	1905	0.3947	0.1459	0.2170	37.0	55.0
28	1905	0.4288	0.1604	0.2366	37.4	55.2
30	1905	0.4327	0.1643	0.2404	38.0	55.6
32	1904	0.4500	0.1705	0.2477	38.0	55.0
34	1901	0.0344	0.0134	0.0189	38.8	54.8
36	1896	0.0325	0.0126	0.0174	38.8	53.6
38	1892	0.0377	0.0149	0.0204	40.0	54.0
40	1887	0.0402	0.0155	0.0223	38.4	55.3
42	1885	0.1225	0.0480	0.0696	39.2	56.8
44	1884	0.1298	0.0487	0.0738	37.5	56.9
46	1882	0.0846	0.0322	0.0474	38.1	56.1
48	1880	0.0883	0.0334	0.0504	37.8	57.1

Appendix J. (cont.)

Depth (cm)	Date (yrs)	Sediment Accumulation Rate ($g/cm^2/yr$)	Organic Accumulation Rate ($g/cm^2/yr$)	Inorganic Accumulation Rate ($g/cm^2/yr$)	Percent Organic (%)	Percent Inorganic (%)
50	1872	0.0200	0.0075	0.0113	37.7	56.3
52	1863	0.0185	0.0071	0.0103	38.6	56.0
54	1841	0.0068	0.0026	0.0037	38.1	54.7
56	1815	0.0066	0.0025	0.0037	38.1	55.1
58	--	--	--	--	39.0	54.0
60	--	--	--	--	40.1	54.7
62	--	--	--	--	39.9	54.6
64	--	--	--	--	38.1	55.8
66	--	--	--	--	37.4	56.5
68	--	--	--	--	38.0	57.9
70	--	--	--	--	37.7	57.6
72	--	--	--	--	37.9	57.2
74	--	--	--	--	37.7	57.0
76	--	--	--	--	37.5	57.7
78	--	--	--	--	37.8	57.5
80	--	--	--	--	37.1	57.5
82	--	--	--	--	36.7	56.7

Appendix J. (cont.)

Depth (cm)	Iron Accumulation Rate ($mg/cm^2/yr$)	Manganese Accumulation Rate ($mg/cm^2/yr$)	Iron Concentration (mg/g d.m.)	Manganese Concentration (mg/g d.m.)	(Fe/Mn)
2	1.2357	0.0361	31.77	0.9281	34.24
4	1.2326	0.0310	31.47	0.7927	39.69
6	0.8375	0.0154	31.54	0.5805	54.33
8	1.0749	0.0166	34.55	0.5332	64.79
10	0.5000	0.0072	35.61	0.5142	69.26
12	0.4099	0.0068	30.43	0.5051	60.25
14	0.6703	0.0137	23.78	0.4877	48.75
16	0.7198	0.0127	26.92	0.4747	56.70
18	0.4923	0.0101	21.29	0.4363	48.81
20	0.7831	0.0158	21.51	0.4332	49.66
22	0.6910	0.0155	17.45	0.3909	44.63
24	0.6520	0.0135	19.19	0.3963	48.42
26	7.3287	0.1577	18.57	0.3995	46.48
28	6.8257	0.1500	15.92	0.3498	45.52
30	--	--	--	--	--
32	7.0154	0.1535	15.59	0.3411	45.70
34	--	--	--	--	--
36	0.4484	0.0089	13.81	0.2727	50.65
38	--	--	--	--	--
40	0.4944	0.0103	12.29	0.2571	47.81
42	--	--	--	--	--
44	1.5547	0.0333	11.98	0.2564	46.75
46	--	--	--	--	--

Appendix J. (cont.)

Depth (cm)	Iron Accumulation Rate ($mg/cm^2/yr$)	Manganese Accumulation Rate ($mg/cm^2/yr$)	Iron Concentration (mg/g d.m.)	Manganese Concentration (mg/g d.m.)	(Fe/Mn)
48	0.9745	0.0209	11.04	0.2369	46.61
50	--	--	--	--	--
52	0.2366	0.0042	12.82	0.2302	55.67
54	--	--	--	--	--
56	0.0762	0.0015	11.47	0.2208	51.95
58	--		--	--	--
60	--		11.38	0.2067	55.04
62	--		--	--	--
64	--		11.02	0.1977	55.71
66	--		--	--	--
68	--		--	--	--
70	--		11.49	0.2103	54.63
72	--		--	--	--
74	--		--	--	--
76	--		11.38	0.2149	52.96
78	--		--	--	--
80			--	--	--
82			12.05	0.2383	50.58

Appendix J. (cont.)

Depth (cm)	CD Accumulation Rate (abs. units/ cm^2/yr)	TC Accumulation Rate (abs. units/ cm^2/yr)	CD Concentrations (abs. units/ g o.m.)	TC Concentrations (abs. units/ g o.m.)	CD/TC	Native Chlorophyll (%)
48	0.2710	0.1304	8.121	3.909	2.078	1.8
50	--	--	--	--	--	--
52	0.0691	0.0441	9.700	6.195	1.566	4.1
54	--	--	--	--	--	--
56	0.272	0.0297	10.75	11.76	0.9142	2.6
58	--	--	--	--	--	--
60	--	--	9.404	8.981	1.047	2.9
62	--	--	--	--	--	--
64	--	--	7.775	8.390	0.9267	1.2
66	--	--	--	--	--	--
68	--	--	--	--	--	--
70	--	--	8.801	9.478	0.9286	2.2
72	--	--	--	--	--	--
74	--	--	--	--	--	--
76	--	--	8.992	8.756	1.027	3.2
78	--	--	--	--	--	--
80	--	--	--	--	--	--
82	--	--	9.196	9.142	1.006	4.4

Appendix J. (cont.)

Depth (cm)	CD Accumulation Rate (abs. units/ cm^2/yr)	TC Accumulation Rate (abs. units/ cm^2/yr)	CD Concentration (abs. units/ g o.m.)	TC Concentration (abs. units/ g o.m.)	CD/TC	Native Chlorophyll (%)
2	0.1576	0.1119	14.13	10.03	1.409	11.0
4	0.1460	0.0793	12.78	6.944	1.840	14.6
6	0.0744	0.0416	9.766	5.467	1.786	11.6
8	0.0793	0.0489	9.148	5.639	1.622	9.1
10	0.0422	0.0245	10.75	6.224	1.727	5.3
12	0.0344	0.0186	9.179	4.949	1.855	3.8
14	0.0487	0.0304	6.131	3.827	1.602	2.6
16	0.0486	0.0272	6.182	3.462	1.785	2.9
18	0.0510	0.0389	6.985	5.326	1.311	3.5
20	0.0848	0.0620	7.014	5.126	1.368	3.4
22	0.1059	0.0848	7.820	6.263	1.249	3.6
24	0.1063	0.0922	8.684	7.535	1.152	2.2
26	1.5203	1.2728	10.42	8.724	1.194	4.6
28	1.4271	1.3290	8.895	8.284	1.074	4.9
30	--	--	--	--	--	--
32	1.6397	0.6595	9.617	3.868	2.486	2.7
34	--	--	--	--	--	--
36	0.1236	0.0429	9.811	3.404	2.882	3.6
38	--	--	--	--	--	--
40	0.1261	0.0477	8.156	3.084	2.645	5.9
42	--	--	--	--	--	--
44	0.4314	0.1799	8.855	3.692	2.398	4.3
46	--	--	--	--	--	--

Appendix K. 1983 North Bay Macrophyte Sampling Data.

Site No.	Depth (m)	Bottom Type	Dry Plant Weight (g/M^2)
1	1.2	muck	164.4
2	1.0	mucky sand	73.8
3	1.3	mucky sand	60.4
4	0.8	sand	7.4
5	1.3	mucky sand	196.6
6	1.7	muck	159.7
7	1.6	mucky sand	124.8
8	2.0	muck	80.5
9	1.2	mucky sand	157.0
10	1.9	muck	171.1
11	2.0	muck	123.5
12	1.6	sand	45.0
13	1.6	mucky sand	114.7
14	1.7	mucky sand	95.3
15	1.7	muck	269.1
16	2.3	muck	338.3
17	2.0	muck	111.4
18	2.1	muck	140.9
19	1.6	muck	198.6
20	2.0	muck	251.6
21	2.1	muck over gravel	134.2
22	2.1	muck sand	93.9
23	1.9	mucky sand	83.2
24	2.3	muck	102.7
25	1.6	sandy muck	93.9