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Ecological Research Series

NORTH AMERICAN PROJECT - A Study of U.S. Water Bodies

Environmental Research Laboratory
Office of Research and Development
U.S. Environmental Protection Agency
Corvallis, Oregon 97330

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16. ABSTRACT

The Organization for Economic Cooperation and Development, an independent international organization for promotion of economic development in member countries, is concerned with both the qualitative and quantitative aspects of economic growth. The Environment Committee of OECD is assisted by a number of delegate groups concerned with policy development in specific sectors of the overall environmental problem. One of these groups is the Water Management Sector Group, which in 1971 established a Steering Group on Eutrophication Control to develop a series of cooperative projects for monitoring eutrophication in inland waters. The overall objective of these projects was the achievement of comparability on nutrient budgets, chemical balances, and biological productivity in water bodies.

In the United States 22 waterbodies were included in the program. Final reports on the limnology of each have been compiled by the United States investigators and are contained in this publication.

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NORTH AMERICAN PROJECT -- A STUDY OF U.S. WATER BODIES

A Report for the Organization for Economic Cooperation and Development

compiled by

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PREFACE

The Organization for Economic Cooperation and Development, an independent international organization for promotion of economic development in member countries, is concerned with both the qualitative and quantitative aspects of economic growth. The Environment Committee of OECD is assisted by a number of delegate groups concerned with policy development in specific sectors of the overall environmental problem. One of these groups is the Water Management Sector Group, which in 1971 established a Steering Group on Eutrophication Control to develop a series of cooperative projects for monitoring eutrophication in inland waters. The overall objective of these projects was the achievement of comparability on nutrient budgets, chemical balances, and biological productivity in water bodies.

A regional approach was utilized to develop four project groups designed to collect comparable data for developing evidence on the degree and extent to which nutrient loading is correlatable with the eutrophic state, and to measure the rate at which eutrophication is developing. The projects and participating countries were:

Nordic Project	Denmark, Finland, Norway, Sweden
Alpine Project	Austria, France, Germany, Italy, Switzerland
North American Project	Canada, United States
Reservoir and Shallow Lakes Project	Belgium, Germany, Netherlands, Spain, United Kingdom, United States

Dr. Richard Vollenweider of the Canada Center for Inland Waters was designated Director of the North American Project, with Dr. Norbert Jaworski of the U.S. Environmental Protection Agency the United States representative. The specific objectives of the North American Project are:

Develop detailed nutrient (phosphorus and nitrogen) budgets for a given selected number of water bodies,

Assess the chemical, physical, and biological characteristics of these water bodies,

Relate the trophic state of the water body to the nutrient budgets and to limnological and environmental factors, and

Synthesize, based on data from all projects, an optimal strategy for controlling the rate of eutrophication.

In the United States, twenty-two water bodies were included in the program. Final reports on the limnology of each, emphasizing the objective of the Project, have been compiled by the United States investigators and are contained in this publication. A synthesis based on the combined data from these reports, and representing the fourth specific objective, will be published subsequently.

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SECTION I - FLORIDA

ANALYSIS OF TROPHIC CONDITIONS AND EUTROPHICATION FACTORS IN LAKE WEIR, FLORIDA

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INTRODUCTION

Lake Weir is a medium size recreational lake in central Florida. Present water quality is good and the lake is classified as mesotrophic. Uses of the lake are nearly exclusively recreational -- swimming, boating and fishing, and many residences along the shore are second homes. Although the lake is well-known to sport fishermen, especially for its largemouth bass, until recently the lake had received no limnological attention, perhaps because of its good water quality and lack of problems. Background limnological information on Lake Weir is thus sparse. Early scientific expeditions to Florida during the late 1700's by John and William Bartram and nineteenth century excursions by J. W. Bailey generally followed waterways for ease of transportation; thus Lake Weir, with no navigable streams entering or leaving it, was apparently missed by these early naturalists (Yount 1963). Stage data has been gathered for the lake since 1936, and a broad-crested, fixed level weir was built in April of 1938 to prevent possible flood damage resulting from hurricanes. A bathymetric map drawn by U.S.G.S. was published for the lake by Kenner (1964). Lake Weir was included in a 1969-70 study of 55 lakes in north central Florida by Brezonik and Shannon (1971), resulting in the first systematic limnological study of the lake.

DESCRIPTION OF THE STUDY AREA

Lake Weir, Florida is located on the Central Florida Ridge at the southern edge of Marion Country, Florida, about one third of the way down the Florida peninsula and midway between the Atlantic Ocean and the Gulf of Mexico (Figure 1). The centroid of the lake is at

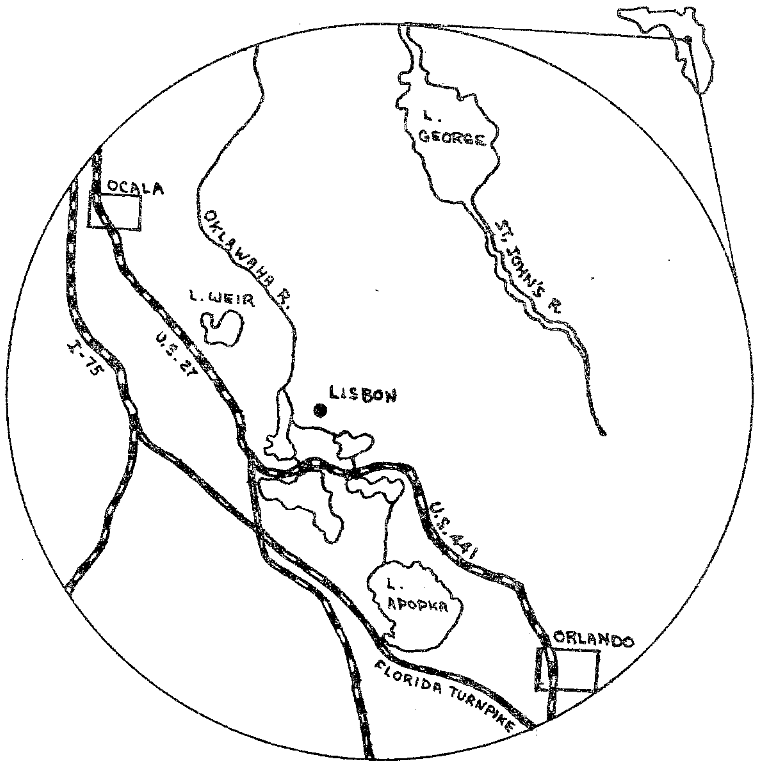

Figure 1. Map of Lake Weir area showing major roads, cities and water bodies.

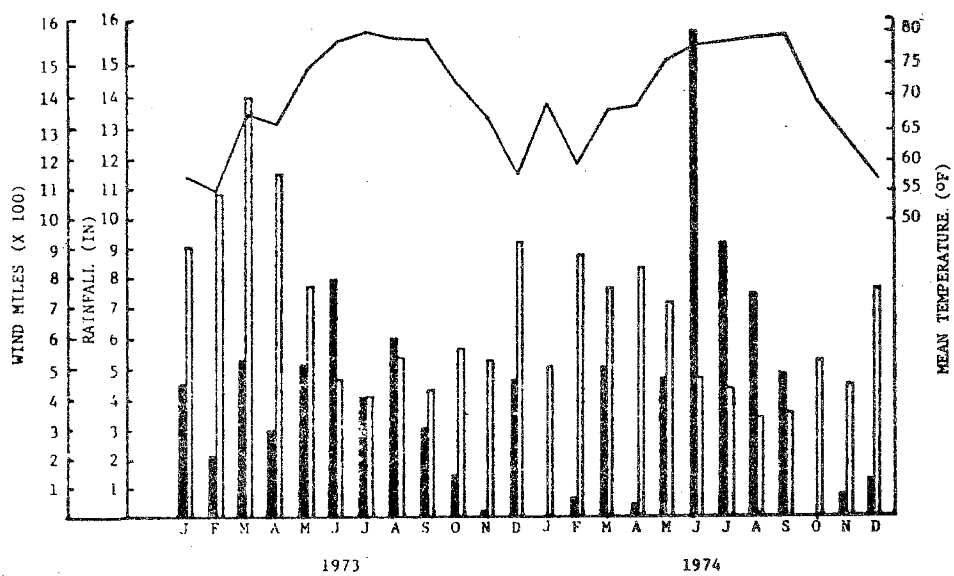

Figure 2. Mean monthly temperature (continuous lines), total monthly rainfall (solid bars) and total monthly wind miles (open bars) for Ocala, Florida, during 1973 and 1974.

29^{o} 1' N, 81^{o} 56' W, and the lake is located in the Lake Weir and Lady Lake Quadrangles of the U.S.G.S. 7.5' topographic maps of Florida. Little Lake Weir is a smaller basin located to the west of the larger lake and is connected to it by an artificial waterway to accomodate the passage of small pleasure boats. The surface elevation of the lakes is 17.4 m above MSL, and the maximum elevation of the surrounding watershed is 42.7 m above MSL. The surrounding terrain consists of numerous small sand hills, and the high permeability of the soil precludes the presence of permanent surface streams in the watershed. Since piezometric maps of the watertable aquifer in the study area do not exist, the area of the watershed was calculated from topographic maps. The Florida Gazeteer of Lakes lists the drainage areas of Lake Weir and Little Lake Weir as 130 km^2 and 33.8 km^2, exclusive of lake surface, respectively. Analysis of the topographic maps, however, revealed a more realistic estimate of 22 km^2 for the watershed of both lakes, exclusive of lake surface. The surface area of the lake was found to be 24.29 km^2 including both basins.

The climate in north central Florida is best described as humid sub-tropical, with short, mild winters and long, hot summers. Average monthly temperatures for Ocala, Florida, 25 km to the northwest, substantiate this point (Figure 2). Average annual rainfall in the area is approximately 133 cm, mainly occurring in the summer months, fall and spring being rather dry (Butson and Prine 1968). Summer rains usually occur as short, convective afternoon showers, while winter precipitation is usually associated with frontal activity. Although the area experiences occasional frosts during the winter months, the total number of hours during which the temperature remains below $0^{o}C$ averages 50 - 67 hr/yr, and below $-2^{o}C$, 17 -33 hr/yr (Johnson 1970). Wind speed is generally light to moderate, blowing from the north and west during the winter, but shifting to easterly in the summer. Hurricanes are seldom in this part of the state. Evaporation from the basin during the study period (1974 calendar year) was calculated to be 122.2 cm/yr, using evaporation data from a standard Weather Bureau pan at Lisbon, Florida, 24 km to the southeast, and monthly pan coefficients determined by Kohler (1954) for Lake Okeechobee, Florida. Evapotranspiration from the watershed was 27.6 x 10^6 m^3 during 1974, based on unpublished calculations by S. Bayley for similar latitudes in the State.

Two distinct aquifers exist in the area of the lake. The upper or watertable aquifer is composed of permeable sand at shallow depth and clayey sand interbedded with some clay lenses at greater depths (Hughes 1974). This shallow aquifer is underlain with a low-permeability sand and clay formation of Miocene origin called the Hawthorn Formation (Snell and Anderson 1970). Below this confining stratum lies the permeable Eocene limestone, the Floridan aquifer, which supplies the State with most of its drinking water (Faulkner 1970; Snell and Anderson 1970). The area surrounding the lake is a principal recharge area for the Floridan aquifer, and in places is covered by only a thin veneer of sand (Snell and Anderson 1970). Most Florida lakes are not connected directly to the deep aquifer, as is evident from their soft water (Brezonik et al. 1969). The General Soil Map of Florida (Beckenbach and Hammett 1962) characterizes the soil in the area as being well-drained to moderately well-drained, thick to moderately thick, acid sands of the Lakeland-Eustis-Blanton association. Because of the high permeability of the soil, land erosion is not a problem in the study area, overland flow being virtually absent except during heavy, long-duration convective storms.

Approximately 55 percent of the land in the watershed is covered by mature citrus groves, and the remaining undeveloped land in the watershed is mainly forested. The area east of the lake is dominated by a pine sandhill association with some mixed hardwoods (oaks, hickory and sweet gum). To the west of the lake, a scrub (turkey) oak association is indicative of the nutrient impoverishment of the well-drained soil. Cypress are found in marshy areas surrounding the lake, and willows can be seen on the undeveloped shoreline. The area between Lake Weir and Little Lake Weir is a marsh dominated by cattail, unbrella-grass, and sawgrass. Some of the shoreline is bordered by well-manicured lawns.

An analysis of 1972 aerial photographs indicates 425 residences in the watershed. Using an average value of 2.5 persons per single family rural residence, a population of 1012 persons is obtained for the watershed. This amounts to a population density of 46.5 persons per km^2. Interviews with local residents reveal, however, that many of the residents are seasonal; thus the year-round population is somewhat lower. The land use characteristics as determined from the aerial photographs are summarized in Table 1. The combination of hilly terrain, well-drained, high hammock soil, and the propinquity of a frost-damping deep lake make the area particularly suited for the growing of citrus crops (Lawrence 1963) which, besides recreation, accounts for most of the economy in the area. Many of the homes are built around the edge of the lake, and three public boat ramps and several public and private beaches provide lake access for residents and visitors. Fishing for largemouth bass and water sports are popular activities. No sewage treatment plants discharge into the lake, and the residences are served by individual septic tanks. Because of the availability of high-quality groundwater from the Floridan aquifer, lake water is used neither for drinking nor irrigation.

MORPHOMETRY AND HYDROLOGY

A comparison of the morphometric characteristics features of Lake Weir (Table 2) with those of temperate lakes of glacial origin indicates that the lake is of rather modest proportion. Compared to sub-tropical Florida lakes, however, it is one of the deeper lakes in the state (Kenner 1964; Brezonik and Shannon 1971). Hypsographic curves for the lake basins (Figure 3) and the bathymetric map (Figure 4) indicate that the lake has relatively steep sides, and a relatively

Table 1. Land use characteristics of the Lake Weir watershed.

Agricultural land (primarily citrus groves)	12.2 km^2
Pasture	1.1
Forest	3.5
Urban (suburban) area	1.5
Wetlands	3.7
Total land area	22.0 km^2

Table 2. Morphometric features of Lake Weir.

Surface area	
main basin	22.78 km^2
Little Lake Weir	1.51 km^2
total	24.29 km^2
Maximum length	5.3 km
Maximum width	5.0 km
Shoreline development index (D_L)	1.7
Volume	
main basin	146.7×10^6 m^3
Little Lake Weir	5.3×10^6 m^3
total	152.0×10^6 m^3
Maximum depth (z_m)	
main basin	10.4 m
Little Lake Weir	6.4 m
Mean depth (both basins) ($\bar{z}$)	6.3 m
Volume development index (D_v)	1.8

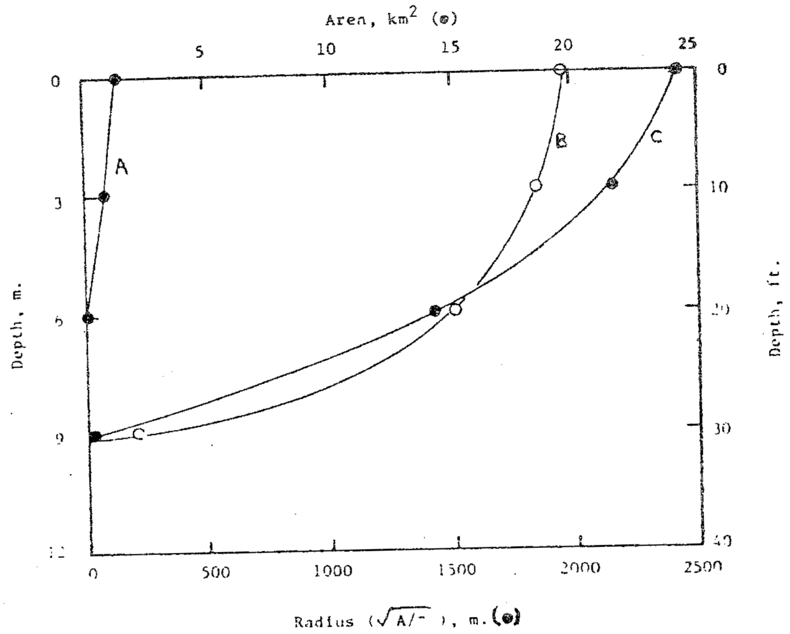

Figure 3. Hypsographic curves for Lake Weir. A: Little Lake Weir, B and C: Lake Weir including both basins.

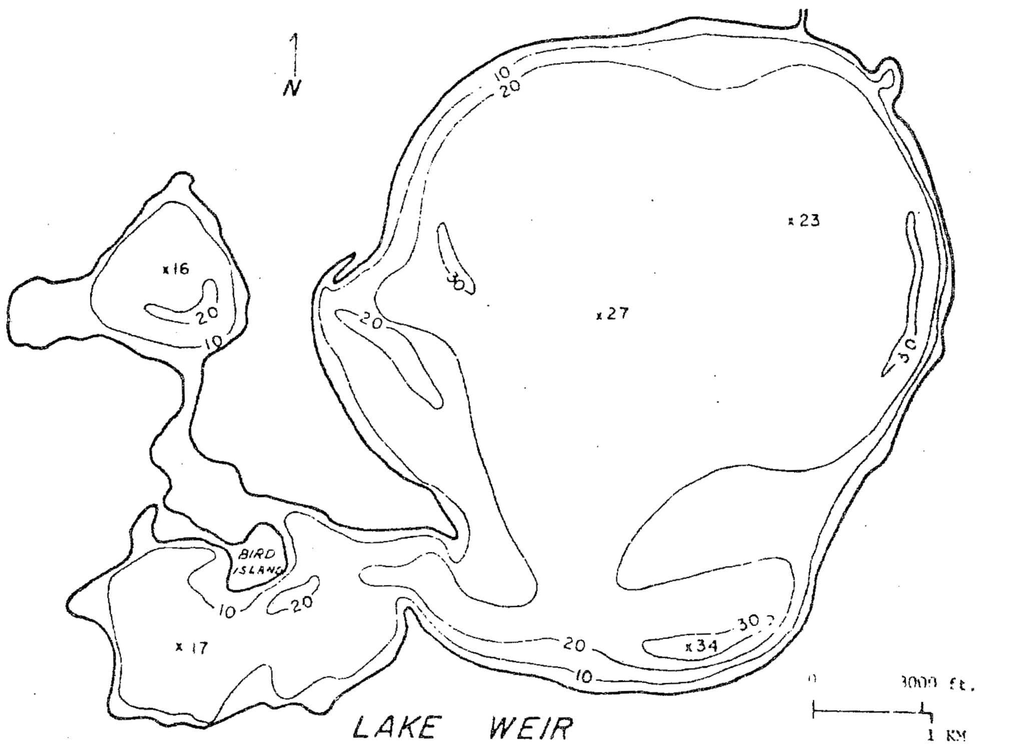

Figure 4. Bathymetric map of Lake Weir, Florida. Bottom contours are in feet.

underdeveloped littoral zone (for a Florida lake). The lake is of solution origin and probably resulted from the fusion of three dolines; this is a common lake form in areas of karst topography (Hutchinson 1957; Yount 1963). The development of volume index (1.92) and the $\bar{z}/z_m$ value (0.64) are indicative of the relatively flat lake bottom. Little Lake Weir has a smaller area/depth ratio than the larger basin. Bird Island, located in the southwestern basin and connected to the mainland by a causeway, was developed by dredge and fill methods in the 1950's. Lake volume and elevation are regulated to some extent by a fixed-level weir at the north end of the lake. Maximum and minimum lake level elevations for the period of record are 18.17 and 16.29 m, respectively, but an examination of the hydrograph indicates much smaller annual and monthly fluctuations (0.4 m for 1974 water year).

At no time during the period of study was there observed evidence of stable stratification of the water column. This apparently reflects the lake's large area/depth ratio; stable stratification occurs in many smaller Florida lakes of comparable depth. The largest difference between surface and bottom temperatures observed was 2.3°C, 1.1°C of which was accounted for in the first meter below the surface. This is a common situation in Florida lakes experiencing intense heating by the summer sun. Temperatures as high as 30°C were observed in the surface waters, and the minimum temperature in winter is about 11°C. In light of the absence of stratification of chemical parameters, and considering the long fetch of the lake, it does not appear that the water column stratifies for more than a few days at a time.

The littoral areas of Lake Weir and the center of the big basin have a sandy bottom, but a loose organic ooze is found in most of the lake bottom. In areas covered by Nuphar in sheltered bays and in the bay north of Bird Island, the sediments are composed of a reddish peat. The smaller, southwestern basin of the big lake is covered by a gelatinous muck. Organic silt deposits are found in some shallow areas. The organic muck is dark-gray to brown in color and has a faint odor of H_2S. The muck in the big lake is thin and unconsolidated, as can be seen from the depths to which a weighted pail and a narrow pipe sank at three locations:

Apparent depth (m)		Difference (cm)
Pipe	Pail	
7.21	6.80	41
7.00	7.00	0
8.02	7.45	57

The narrow pipe penetrated the thin sediment, while the broader pail was stopped more readily. These data indicate that about a half meter of thin, unconsolidated sediment occurs in some areas of the lake. Table 3 gives some chemical characteristics of Lake Weir sediments.

Table 3. Chemical Characteristics of Lake Weir sediment*

Volatile solids (%)	54.7
Total carbon (%)	31.8
Total nitrogen (mg/g)	23.9
Free ammonia (mg/g)	0.15
Total phosphate (mg/g)	0.62
C/N	13.3
N/P	38.5
Iron (mg/g)	3.4
Manganese (mg/g)	0.1

*Results expressed on a dry weight basis.

Table 4. Precipitation recorded at stations near Lake Weir (in cm).

1974	Lisbon, Fl.	Ocala, Fl.
January	0.25	0.30
February	4.27	1.60
March	9.07	12.60
April	2.57	1.12
May	9.58	11.84
June	34.29	40.59
July	17.70	23.16
August	14.20	18.72
September	10.34	12.04
October	0.58	0.05
November	1.65	1.78
December	4.93	3.02
Total	109.43	126.82
Average rainfall	119.7	

Warburg respirometry indicated that the sediments consumed 310 µl O_2 g/dry wt. during a 24 hr period. This value is intermediate between the low oxygen demand of oligotrophic lake sediments (50-200 µl O_2/g dry wt.-day) and eutrophic Florida lake sediments (1500-4000 µl O_2/g dry wt.-day)(Brezonik, unpublished data). Since Lake Weir is a soft water lake, there is no $CaCO_3$ in the sediment; volatile solids should reflect its organic content. The high C:N ratio is typical of Florida lakes and the high N:P ratio reflects the chemical conditions in the overlying water.

Calculation of a water budget for the lake is complicated by the lack of surface drainage in the sandy watershed. Water input is thus the sum of rainfall on the lake surface plus seepage from the shallow watertable aquifer. Some surface sheet-flow from the immediate shoreland area probably occurs during intense rainfalls, but on a relative basis this is considered a negligible input. Atmospheric precipitation is recorded at Ocala (25 km northwest) and at Lisbon (24 km southeast) of the Lake Weir watershed. While micrometerological peculiarities may influence convective precipitation patterns near the lake, the Lisbon station is located in an area surrounded by lakes and probably has similar precipitation patterns. In a similar hydrologic study on Lake Kerr in the nearby Ocala National Forest, Hughes (1974) found that averaging rainfall measurements from stations at this distance resulted in 70-85 percent of the calculated monthly rainfall averages being within 3 cm of the "actual" value. The rainfall patterns on the watershed are summarized in Table 4.

On an annual basis the net contribution of groundwater (i.e. seepage into lake minus groundwater recharge) is small. The net contribution can be calculated from the formula

$$\Delta S = R + (S - GWR) - E - O$$

where ΔS is change in storage, R is rainfall on lake surface, (S-GWR) is net groundwater contribution (S = seepage, GWR = groundwater recharge), E is evaporation and O is surface outflow. Outflow occurs during part of the year over a broadcrested weir at the north end of the lake and into a canal that feeds into a marsh and eventually to the Oklawaha River. All of the above terms except S and GWR can be directly evaluated: ΔS from stage records for the lake, R and E from rainfall and pan evaporation measurements as described above, and O by calculation from the difference between recorded lake stage and the known elevation of the weir using the formula

$$Q = 3.3\ b\ (\Delta H)^{3/2}$$

where Q is flow (in cfs), b is width of weir (in feet) and H is difference between lake and weir elevations (in feet). These terms are tabulated in Table 5, and from the values a net groundwater contribution of $0.83 \times 10^6 m^3$ for calendar year 1974 is calculated.

The above value is misleadingly small in terms of the importance of groundwater flows into and out of the lake. Depending on the

Table 5. Water budget for Lake Weir (Calendar 1974)

A.	Water In		10^6 m^3
	Rainfall (Table 4 and Figure 5 (1.20 m/yr)(24.3 km^2)	=	29.1
	Seepage*	=	2.87
		Total	31.97
B.	Water Out		
	Evaporation	=	29.6
	Outflow	=	1.1
	Groundwater recharge*	=	4.96
		Total	35.66
C.	Change in Storage		
	Measured: $\Delta S = (-0.1m)(24.3km^2)$	=	-2.43
	Calculated: $\Delta S = \Sigma$ Inputs $- \Sigma$ Outputs $= 31.97 - 35.66$	=	-3.69
	Unaccounted for outflow:		1.26

D. Retention Time $= V/Q_{out} =$

Including evaporation: $\frac{152 \times 10^6 m^3}{35.66}$ = 4.2 yr

Including only surface and subsurface outflows:

$\frac{152 \times 10^6}{6.06 \times 10^6}$ = 25.0 yr

E. Hydraulic Loading $= Q_{in}/A$

$= \frac{8.76 \times 10^4 m^3/day}{24.3 \times 10^6 m^2}$ = $3.6 \times 10^{-3} m/day$

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* Seepage and recharge calculated by integrating the areas above and below the axis of Figure 5, respectively.

relative levels of the lake surface and the shallow watertable aquifer, both of which fluctuate seasonally or even weekly, groundwater can flow into or out of the lake. Monitoring the direction and magnitude of these flows would be an extensive undertaking, requiring a network of wells around the lake. However the net flow on a weekly basis can be calculated from the weekly data for all the other hydrological parameters. If one makes the assumption that simultaneous seepage into the lake and recharge to the aquifer, do not occur, weekly net flows (Figure 5) probably approximate the gross flows during this time interval, and by summing over the year the weeks with net seepage and the weeks with net recharge, the gross flows in each direction can be estimated. It should be noted that Lake Weir is evidently not directly connected with the deeper Floridan aquifer. The low alkalinity and hardness values in the lake indicate no flow of water from this aquifer into the lake. Also, the piezometric surface of the aquifer in this area lies below the surface lake evlevation (Mills and Laughlin 1974). The estimated water budget and hydraulic retention time for Lake Weir is summarized in Table 5.

SUMMARY OF LIMNOLOGICAL CHARACTERISTICS

The physical and chemical characteristics of Lake Weir are summarized in Table 6 . The high temperatures are characteristic of Florida lakes (Yount 1963) and probably lead to more stability in the water column than would occur in colder waters. Secchi disc transparency falls within the mesotrophic range defined for low-color lakes by Brezonik and Shannon (1971). The low color reflects the absence of extensive swamps and pine woods in the area, and turbidity is in the lowest third of the 55 lakes studied by Brezonik and Shannon (1971). The mean solar radiation on the watershed is 400 langleys/day.

The chemical data (Table 6) indicate a slightly acid lake with intermediate levels of nitrogen and phosphorus. Dissolved oxygen is always high throughout the water column during the day, and it is unlikely that anoxic conditions ever develop in the water column. Alkalinity is low, and the pH is near nutrality, showing little seasonal variation. The dominant ions are Na^+ and Cl^-, the concentrations of which are somewhat higher than those in soft water, oligotrophic lakes in unpopulated watersheds east of Gainesville, Florida (Brezonik et al. 1969). Cultural sources may be responsible for some of these ions, but there is little other evidence to indicate much cultural influence on the general chemical composition of the lake. Iron and manganese are near or below their limits of detection throughout the water column at all times. Silica is relatively low and fails to show a pronounced seasonal trend.

Levels of nitrogen and phosphorus species are given in Table 7 , for the 1974 study period, for the period of maximum insolation, and for the earlier (1969-70) study. The mean concentrations represent arithmatic means for all of the samples taken during the study period,

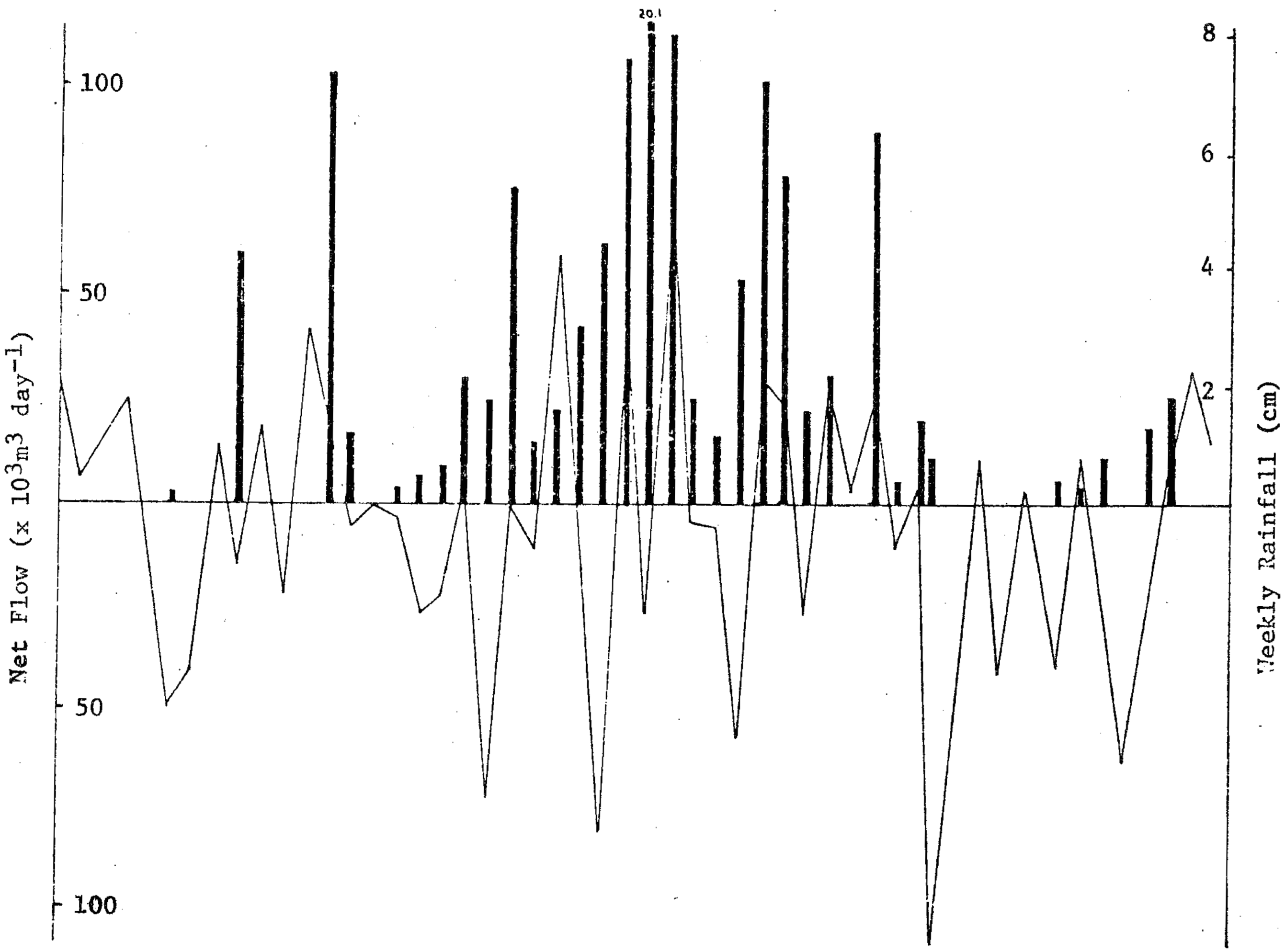

Figure 5. Weekly net underground flow calculated from hydrologic data. Values above the horizontal axis represent seepage into the lake, and values below the line indicate groundwater recharge.

Table 6. Physical and chemical characteristics of Lake Weir.[1]

Parameter	Range	Mean
Turbidity (JTU)	0.7 - 3.1	1.5 $\pm$ .4
Secchi disc (m)	1.1 - 2.8	1.9 $\pm$ .4
Color (Pt units)	5.0 - 30	16 $\pm$ 6
Conductance (μmho cm^{-1})	16 - 520	133 $\pm$ 64
pH	5.3 - 7.2	6.6 $\pm$ .5
Alkalinity as $CaCO_3$	0 - 54	11.5 $\pm$ 6.0
Acidity[2]		1.0
Cl[2]	26.0 - 29.5	27.5
SO_4[2]	4.2 - 8.4	6.0
Ca[2]	3.6 - 7.8	5.3
Mg[2]	3.1 - 5.8	3.9
Na[2]	10 - 21	14.9
K[2]	1.0 - 3.0	1.9
Fe[2]		Trace
Mn[2]	.001 - .03	.01
Fe[2]	.16 - .26	.21
COD[2]	16 - 45	28
SiO_2[2]	0.24 - 0.39	0.33

[1]Values in mg $liter^{-1}$ except where units are specified and pH.

[2]Range of 4-7 measurements during the period 10-68 to 6-70.

Table 7. Summary of nutrient levels in Lake Weir[1]

Constituent	1969-1970[2] Range	Mean	1974-1975[3] Range	Mean
Total organic N	0.67-1.07	0.84	0.45-1.49	0.90±0.26 (0.98)
Ammonia	0.04-0.55	0.17	0.005-0.30	0.038±0.050 (0.018)
Nitrate	0.00-0.06	0.035	0.00-0.20	0.033±0.038 (0.019)
Orthophosphate	0.001-0.014	0.0065	0.005-0.15	0.025±0.020 (0.022)
Total phosphate	0.01-0.055	0.024	0.019-0.40	(0.083±0.062) (0.072)

[1]Results in mg N or P/ℓ.
[2]Range and mean of 7 measurements over 13 months.
[3]Range mean and standard deviation of 15 measurements over 15 months; numbers in parentheses represent means during nominal growing season (June-October).

Table 8. Summary of Biological Parameters for Lake Weir

Parameter	1969-1970* Range	Mean	1974-1975** Range	Mean
Phytoplankton (cells-filaments/ℓ)	-	2626	193-2685	1358
Phytoplankton equitability	-	-	-	0.57
Chlorophyll *a* (mg/m^3)	4.0-10.0	6.0	0.0-33.9	8.2
Primary production (mg/m^3-hr)	5-30	12.5	-	7.6
Zooplankton (organisms/ℓ)	-	-	96-403	261

* Range and mean of 7 measurements over 13 months except for phytoplankton count which is a single composite sample for May 28, 1969.

** Range and mean of 11 sampling dates for all but primary production which was measured 3 times: March, August and December.

regardless of station, depth or date. Since the lake does not stratify stably and no pronounced trends were ever observed depth profiles or areal surveys, the lake can be regarded as essentially homogeneous. Ortho- and toal phosphate were determined by the Murphy and Riley single-reagent molybdenum blue method, adapted to the Technicon AutoAnalyzer (EPA 1971), the total P analysis following digestion with acid persulfate. Inorganic N was determined by AutoAnalyzer methods (EPA 1971), and Kjeldahl nitrogen was determined by Standard Methods (A.P.H.A. 1971). Seasonal variations in these nutrients are graphed in Figure 6.

Brezonik and Shannon (1971) proposed criteria for common trophic state indicators for north central Florida lakes based on similarity (cluster) analysis performed on trophic indicator data from their Florida lake survey, which included Lake Weir. They arrived at ranges for clear mesotrophic lakes of 23 ± 14 mg/m^3 for total P and 730 ± 300 mg/m^3 for total organic N. The values for Lake Weir indicate that the lake exceeds this range for total P, being at the lower edge of the eutrophic range, but the organic N concentration falls within the mesotrophic range. The ratio of total N:total P for 1974 was 13.2 by weight (29 by atoms) which is a considerable decrease from the average ratio for 1969-70. Both ratios are indicative of phosphorus limitation in the lake biocoenosis.

The result of an intensive plankton monitoring program are summarized in Table 8. Chlorophyll analyses (corrected for phaeopigments) on water samples filtered through 0.8 μm membrane filters indicated that chlorophyll *a* was the only important chlorophyll in the water column; phaeopigments were rarely encountered, except in the surface samples. The mean, chlorophyll *a* concentration (8.24 mg/m^3) for 1974 compares with a mean of 6.0 mg/m^3 for 1969-70 and is between the ranges reported for mesotrophic and eutrophic lakes by Brezonik and Shannon (1971). Primary production was measured on three occasions during 1974 using the ^{14}C assimilation method. The mean volumetric fixation rate was 7.6 mg C/m^3-hr. These *in situ* values compare with a mean of 12.5 mg C/m^3-hr obtained for 7 measurements during 1969-70 using a laboratory light box. Integrating the *in situ* measurements taken over the water column during midday incubations and extrapolating to daily rates using diurnal productivity curves typical of the area, yields a very approximate annual C fixation rate of 36 g C/m^2-yr. Again, this represents a value intermediate between mesotrophy and oligotrophy.

The phytoplankton of Lake Weir is dominated by blue-green algae (Cyanophyta). The dominant plankter during summer 1974 was *Oscillatoria submembranosa* Drouet (= *Lyngbya digusteii* Tiffany). During the autumn and winter, *Oscillatoria* alternated with *Microcystis aeruginosa* as the dominant plankter, depending on the station and date. An unidentified coccoid green alga was occasionally dominant in numbers, but was never important in terms of biomass. The pelagic diatom, *Synedra ulna*, was frequently observed in the summer plankton, but disappeared in the winter. Desmids, particularly *Staurastrum* spp., were always

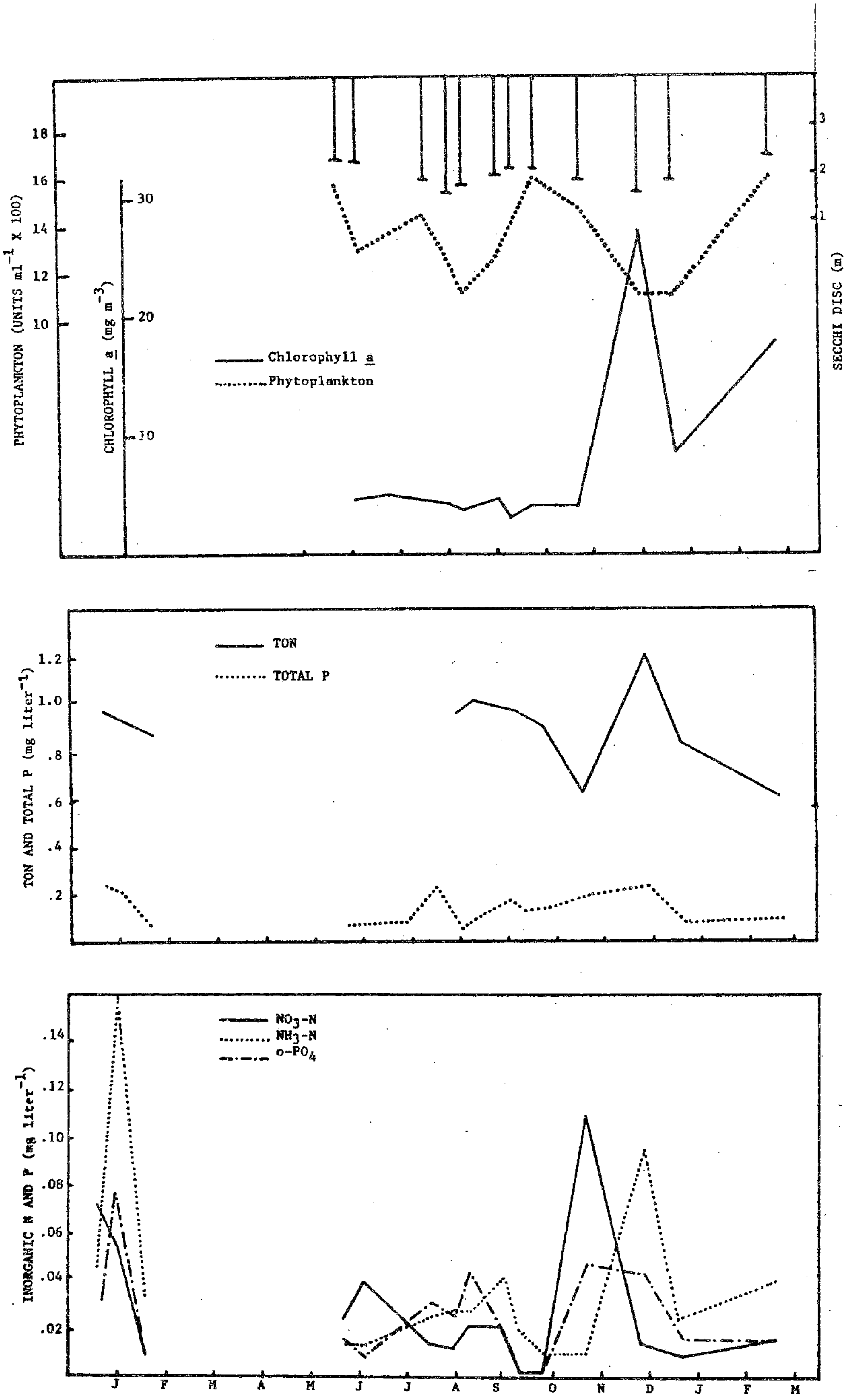

Figure 6. Seasonal variation in Secchi disc transparency phytoplankton, chlorophyll a, and nutrient concentrations in Lake Weir during 1974-1975.

present. January 19, 1975, a bloom of Glenodinium quadriens (Pyrrophyta) in Little Lake Weir resulted in a chlorophyll a concentration of 33.9 mg/m^3, the highest value observed in the lake to date. Although Anabena spp. and Aphanizomenon flos-aquae were occassionally observed in the net plankton, these species, which are common in eutrophic Florida lakes, were never of numerical importance. The mean cell was 1358 cells-filaments/ml, and there was no pronounced seasonal trend in plankton counts. Regression analysis yielded no significant relationship between chlorophyll a and cell counts, but a moderate correlation ($r = -0.57$) was observed between cell counts and Secchi disc transparency. Equitability was calculated by comparing the Shannon-Weaver diversity indices against the MacArthur broken stick model (EPA 1973). This parameter is thought to be a more sensitive indicator of stress than the diversity index alone, and the value for the Lake Weir phytoplankton (.57) is indicative of relatively healthy biocoenosis.

Zooplankton were collected and counted from several stations in the pelagic zone. The mean concentration and standard deviation of all of the samples was 261 ± 99 organisms/ℓ, with the majority of zooplankton being immature stages of copepods. Rotifers were also abundant, particularly Nothalca sp., and Monostyla and Branchionus spp. The cladocera were occasionally represented by Bosmina coregoni, an indicator of good water quality in temperate lakes.

Dredge transects of the pelagic zone indicated that both the muck and sand of the large basin were largely devoid of macroinvertebrates. As the shore is approached (<100m), the ooze and also the sand beaches are populated with Sphaeriid clams, a few gastropods, Hexagenia sp., chironomids, and tubificid worms. These communities are never dense (<500 organisms m^{-2}), and diversity was moderate to high. The presence of Hexagenia is encouraging, as this organism, which is particularly susceptible to low dissolved oxygen levels, provides excellent food for sportfish.

The unconsolidated sediments in the large basin provide an unsuitable substrate for submerged flora, but the shallower bays are covered with Eliocharis elongata and Utricularia sp. The latter covers nearly the entire bottom of the bay north of Bird Island. Potomogeton illinoensis can be observed growing on the sandy beaches. The most conspicuous macrophytes in the lake are the emergent species growing in the littoral. The margin of all three basins exhibits a fringe of Juncus effusus and the grass, Panicum hemitomon, growing out from the shore as far as 20-25 m, particularly between boat docks and wherever shelter is afforded. Juncus grows closer to shore; the Panicum grows out into depths greater than 2 m. In the marsh separating Little Lake Weir from the larger basin, cattail (Typha), bulrush (Scirpus), sawgrass (Cladium), and Pontederia are the dominant forms. The bay north of Bird Island displays Nymphoides aquaticum (big floating heart) and patches of spatterdock (Nuphar lutem) which also grows in sheltered coves. Although waterhyacinth (Eichhornia crassipes) occasionally has been observed in the lake, and along with Salvinia is often in nuisance proportions in the canal leading to the outflow weir, this common pest in Florida waters does not form floating mats in the lake.

NUTRIENT BUDGETS SUMMARY

One of the most unfortunate difficulties encountered in the study of Florida watersheds is the virtually complete lack of data for nutrient loading rates measured in the State. The highly permeable sands, high soil temperatures, unique geology and sub-tropical climate would seem to make application of temperate zone data to southern watersheds a questionable procedure. Nonetheless, with the exception of the data on N and P in rainfall in the Gainesville, Florida area (Brezonik et al. 1969), the reviews of Loehr (1974), Chiu et al. (1973), National Eutrophication Survey (1974), and Uttormark et al. (1974) do not include a single study on sub-tropical watersheds. Determination of a nutrient budget for Lake Weir is further complicated by the fact that the lake has no surface streams or other point nutrient sources flowing into it, and all nutrient loading thus is diffuse. Because of the dearth of information on nonpoint loadings in Florida, nutrient loading rates were calculated for the Lake Weir watershed (Table 9) using a variety of assumptions and areal yield rates for N and P from the literature. In light of this, the loading estimates must be viewed as only approximate and subject to revision as more becomes known about nutrient runoff from the land and subsurface nutrient transport in Florida soils.

Rainfall nutrient levels were taken from Brezonik et al. (1969) for rainfall at Gainesville, 60 miles north of the lake. Urban runoff values are from Weibel (1969) and represent averages for residential-light commercial areas found in the study area. Septic tank contributions were estimated following Brezonik and Shannon (1971). An average septic tank was assumed to have a daily effluent flow of 475 ℓ with total N and P concentrations of 35 and 8 mg/ℓ, respectively. For homes located on the lakeshore, 25 percent of the N and 10 percent of the P were assumed transported to the lake. These values were reduced to 10 percent of the N and 1 percent of the P for houses in the watershed not adjacent to the lake shore. Pasture land and forested land values were obtained from Uttormark et al. (1974). In order to take into account the low nutrient binding capacity of the sandy acid soils in this area, their "average" and "high" areal yield rates were averaged for these two land-use classifications. Nitrogen and phosphorus contributions of citrus groves were taken from estimates by Brezonik and Shannon (1971) based on the average fertilizer composition and application rates to the groves. It was assumed that 10 percent of the N and 1 percent of the P reached the lake water. It is generally agreed (Uttormark et al. 1974; Lee et al. 1975) that wetlands of the general type found in the Lake Weir watershed make no net contributions of N or P to aquatic systems, although they may affect nutrient concentrations by acting as "sinks" during the growing season and as

Table 9. Nutrient Budget for Lake Weir, 1974.

Source	Area (km^2)	Areal Yield Rate (g/m^2 - yr) N	Areal Yield Rate (g/m^2 - yr) P	Nutrient Loading Rate (g/yr) N	Nutrient Loading Rate (g/yr) P
Rainfall	24.29	0.58	0.044	1.409×10^7	1.06×10^6
Urban	1.5	0.88	0.11	0.132×10^7	0.165×10^6
Pasture	1.1	0.75	0.065	0.083×10^7	0.072×10^6
Forest	3.5	0.37	0.060	0.130×10^7	0.216×10^6
Agriculture	12.8	2.24	0.018	2.867×10^7	0.230×10^6
Septic tanks				0.052×10^7	0.042×10^6
				4.67×10^7	1.79×10^6
Loss through outflow ($Q_{out} \times P_{ave}$)					0.097×10^6
Surface loading rate (g/m^2 - yr)				1.92	0.074
Volumetric loading rate (g/m^3 - yr)				0.30	0.012

$\bar{z}/t_w = 6.3/25 = 0.25$ (without evaporation)

$\bar{z}/t_w = 6.3/4.2 = 1.5$ (with evaporation) "sources" during decomposition in the colder months. Lake Weir supports seasonal populations of water fowl, but lack of reliable census figures make an estimate of their nutrient contributions impossible. The densities do not appear to be large, however, and this is unlikely to be a serious source of error.

DISCUSSION

Lake Weir can be characterized as a sub-tropical, low-acidity, soft water lake, low in color and turbidity, and exhibiting no thermal stratification. Although the lake is relatively deep for Florida, the lack of stable stratification precludes formation of a hypolimnion, and dissolved oxygen concentrations are high throughout the water column at all times. Concentrations of nitrogen and phosphorus are moderate to high, exhibiting no distinct seasonal trends. Although the phytoplankton is dominated throughout the year by blue-green algae, diversity in this biocoenosis is relatively high, and nuisance conditions do not occur. Primary productivity in this community is low to moderate. Although macrophytes are common, floating mats of hyacinths or nuisance growths of Hydrilla, Salvinia, or Pistia are not found in the lake. Diversity is relatively high among both the zooplankton and the benthos, and the presence of Bosmina coregoni and Hexagenia in these two habitats, respectively, are indicative of good general water quality. Largemouth bass and a variety of other sport fish are abundant.

Comparability between trophic state indicators in temperate and subtropical lakes, and perhaps permissable nutrient loading rates as well, must be viewed in the light of the fundamentally different patterns of organization in temperate and tropical systems. Whereas the former are adapted to a strong seasonal pulse of insolation which is used to build storages that must tide the community over until the following spring, the latter is organized around a higher overall energy input with much less severe seasonal variation (Odum 1971). In a temperate lake, the spring overturn coincides with a period of high insolation, offering the plankton a banquet of readily assimilable inorganic nutrients regenerated during winter stratification along with the sunlight necessary to incorporate them into biomass. While insolation is lower at the time of fall circulation, a pulse of nutrients from the hypolimnion probably is instrumental in supporting an autumn algal maximum. In tropical lakes insolation is relatively high during the entire year but falls below that of northern latitudes during the summer. In many Florida lakes, spring and fall phytoplankton maxima are replaced by oscillations occurring, seemingly at random, throughout the year. It would seem that a system in which a significant portion of the nitrogen and phosphorus is tied up in more or less refractory algal biomass (Gunnison and Alexander 1975) would be unable to support the same sized blooms, given the same total N and P concentrations or loadings, as a system in which virtually the entire nutrient pool is in the inorganic form at a time favorable for algal growth.

These differences are important not only in measuring the trophic status of tropical and subtropical lakes but may also modify their critical nutrient loading rates. Chlorophyll a values in nutrient rich tropical lakes not dominated by motile Pyrrophyta are reported to be remarkably low (Berman and Pollingher 1974) compared to temperate lakes of similar trophic status. Inasmuch as high gross primary productivity is reported by these authors, the low chlorophyll values probably represent "sun" plants with high gross to net production ratios. Relatively lower net carbon assimilation rates may be expected in the spring in subtropical systems, due to relatively lower vernal insolation and higher maintenance costs of overcoming thermal disordering at the higher water temperatures. Berman and Pollingher (1974) report plankton respiration rates for Lake Kinneret of 40-50 percent of gross photosynthesis.

Several reported criteria for N and P levels associated with various trophic states in temperate lakes were reviewed by Vollenweider (1968). N and P concentrations in Lake Weir exceed the critical concentration for one or both nutrients in every case. The concentrations of total N and P fall just above the mesotrophic range for uncolored north central Florida lakes (Brezonik and Shannon 1971). Most of the biological parameters, however, fall within the mesotrophic (or occasionally the oligotrophic) range for either European or Japanese temperate lakes (Vollenweider 1968; Sakamoto 1966) or for Florida lakes (Brezonik and Shannon 1971). If maximum available nutrient levels after the season of minimum growth are indeed relatively lower in tropical than in temperate lakes, maximum biomass during the season of maximum growth would expectedly be lower in comparably loaded tropical lakes than in temperate lakes.

In order to clarify further the trophic status of Lake Weir, various trophic state indices were calculated for the lake. The TSI derived by Brezonik and Shannon (1971) was recalculated for the recent data and yielded a value of 3.58, within the range for mesotrophic lakes. However, the value has risen from the value (3.30) calculated from 1969-70 data. An increase since 1969-70 was also noted in the concentrations of N and P, and in a decrease in the N:P ratio from 96 (by atoms) to 29. Little change was noted in the biological parameters, although chlorophyll concentrations increased somewhat. It is not known to what extent these changes represent experimental artifacts, stochastic elements in the environment, long term system cycles, or the impact of cultural encroachment on the lake. The TSI equations formulated by Carlson (1975) based on Secchi disk transparencey was applied to the lake data, and values of 51, 51 and 64 were derived for Secchi disk, chlorophyll, and total phosphorus data, respectively. This would indicate that the lake falls almost directly in the middle of a scale based on Secchi disk transparency, but again the nutrient concentration overrates the lake in the direction of eutrophy. While the possibility exists that some of the measured phosphorus is not available to the plankton, it is still tempting to suggest that higher nutrient concentrations are required to produce the same standing crop in southern waters. The agreement of the trophic state indices with the more qualitative biological observations in the lake clearly delineate Lake Weir as a mesotrophic lake.

Calculation of the nutrient loading rates for Lake Weir based on the nutrient and hydrologic budgets presented previously leads to the values presented in Table 9 . Plotting areal P loading against $\bar{z}/t_w$ (Vollenweider 1974), Lake Weir is found to have a P loading rate which falls within twice the permissable loading for a lake of its depth and flushing time. Inasmuch as one of the constraints on this model is that the lake act as a mixed reactor, Lake Weir should be an ideal case. The observed decrease in the N:P ratio in the lake during the past five years is consistent with a relatively high P loading rate, while the relatively stable biological system renders a condition of severe stress unlikely. It is interesting to note that the empirical loading rates for Florida lakes by Brezonik and Shannon (1971) permit higher areal P (and N) loading rates, Lake Weir falling just above the permissible range for nitrogen and at the maximum permissible rate for phosphorus. This model does not take the flushing rate into account.

SUMMARY

Lake Weir, Florida, a 2200 ha soft water lake located on the central Florida ridge, has a watershed dominated by citrus groves and receives no permanent surface streams or wastewater influents. Biological parameters in the lake indicate a diverse, moderately productive ecosystem which exhibits no nuisance conditions associated with excessive growth of macrophytes or algae. Two independently derived trophic state indices bear out biological delineation of the mesotrophic status of the lake. Nitrogen and phosphorus concentrations in the lake are indicative of borderline eutrophic conditions in the lake, and there is some evidence for a significant increase in phosphorus in the lake since a previous study. Lake Weir has a low flushing rate which makes it sensitive to nutrient loading, and application of lake data to the Vollenweider input-output model, indicates that areal P loading rates are just at the danger level.

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SECTION II - MINNESOTA

AN OVERVIEW OF LIMNOLOGICAL CHARACTERISTICS OF SHAGAWA LAKE, MINNESOTA

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INTRODUCTION

Shagawa Lake, known to the Chippewa Indians as Ga-Shagawigumag-sag or "long narrow lake" (Winchell, 1887), is located adjacent to the city of Ely in northeastern Minnesota. The lake, formed during the retreat of the Wisconsin Glacier about 10,000 years ago, lies in a bedrock basin partially dammed by drift produced by glacial erosion and deposition. Significant point-source nutrient enrichment of Shagawa Lake began in 1901 when untreated wastewater from about 3500 people was discharged directly into the lake. Primary treatment began in 1912 and secondary treatment in 1954. This nutrient enrichment distinguished Shagawa from the oligotrophic lakes typically found in this region of Minnesota. In 1973, tertiary treatment was initiated for phosphorus removal.

GEOGRAPHICAL DESCRIPTION OF THE LAKE

The lake is located at latitude 47°55'N, and longitude 91°52'W, and at an altitude of 407.8 m above mean sea level. The drainage basin covers 269 km^2. About 160 km^2 of that includes and drains into Burntside Lake (located to the NW), then into Shagawa Lake through Burntside River. The drainage basin below Burntside Lake and including Shagawa Lake is 109 km^2 (Malueg, et al. 1975).

The climate is relatively severe for the continental United States. Ice covers the lake about 6 mo/yr. For example, the open-water period in 1972 was from 10 May to 15 November.* Monthly air temperature and precipitation averaged 2.8°C and 5.1 cm/mo, respectively. About 70% of the precipitation fell during the open-water season. Lake evaporation for the open-water season was 7.6 cm/mo, while the annual lake evaporation value was 66.7 cm. Wind direction was generally from the N or NW.

*All data unless otherwise stated are for 1972, the year prior to operation of the tertiary wastewater treatment plant at Ely.

The geological formation in the area resists erosion and leaching by surface waters. The bedrock consists of Precambrian, metamorphic rock including granite, slate, and greenstone. A rich iron ore deposit lies along the southern shore of the lake. Overlying the bedrock is a patchy distribution of glacial sediments including sand and gravel, plus lucustrine silts and calcareous clays that apprently also underlie the organic sediments of Shagawa Lake (Bradbury and Waddington, 1973).

Forest and marsh comprise most of the land, 77% and 15% respectively, while construction and agriculture use 7% and 1%, respectively. Deciduous forests of aspen (Populus tremuloides) and birch (Betula papyrifera), plus coniferous forests of jack pine (Pinus banksiana), spruce (Picea mariana), and fir (Abies balsomea) are the dominant forms of vegetation.

In 1888 Ely was incorporated as a village with a population of 104. By 1900 the population had increased to 3717 as a result of the developing iron mining and logging industries. The population peaked at 6151 in 1930, and has since declined and stabilized at about 5000 residents today. Now the major industry is tourism, with heavy emphasis on water sports, primarily fishing and some "sea plane" activity.

In 1901 Ely began discharging its untreated municipal wastewater directly into Shagawa Lake. Although primary treatment began in 1912, secondary treatment did not follow until 1954--and all treated water was discharged into the lake. Furthermore, during the mining years of 1884-1967 (Somrock, 1974) an unknown amount of mine sump water was discharged into the lake.

That dual abuse of Shagawa Lake has produced problems. Although Shagawa Lake was the original source of Ely's drinking water, in 1932, the city constructed a pipeline to draw drinking water from nearby Burntside

Lake. Furthermore, Shagawa Lake was closed to swimming between 1968 and 1972 because bacteria originating from wastewater sometimes exceeded health standards.

A tertiary treatment plant designed to reduce wastewater total phosphorus concentration to <50 μg/l commenced operation in early 1973, thus reducing the input of wastewater phosphorus by 99% and the total phosphorus input to the lake by 70-80%. Approximately 100 unsewered homes and resorts which dot the lake shoreline have septic tanks and contribute an unknown amount of phosphorus to the lake.

Descriptions of Morphometric and Hydrologic Characteristics of Shagawa Lake

Shagawa Lake has a surface area of 9.2 km^2 and is approximately 6.6 km long and 2.9 km wide (Figure 1). The 34.1 km of shoreline includes 5.1 km of island shoreline. The lake has a maximum depth of 13.7 m, mean depth of 5.7 m, and volume of 5.3×10^7 m^3. There is no man-made regulation for depth control. Natural volume variation is approximately $\pm 5\%$ of the mean. The surface area ratio of "shallow" to "deep" waters is 1.54 with the 5.25 m depth arbitarily separating shallow and deep water; the ratio of volume is 2.75. A clearly defined hypolimniom rarely exists. Summer thermal stratification usually develops in early June and extends to early September, although gradients are sometimes minimal.

Generally the sediments of nearshore areas are very sandy with little organic matter, the sediments of the mid-depths are composed of algal biopel and silt-sized particles, and the sediments of the deep holes are composed of algal biopel and clay-sized particles (Waddington & Wright, 1974). Phosphorus primarily exists in the sediments in association with iron.

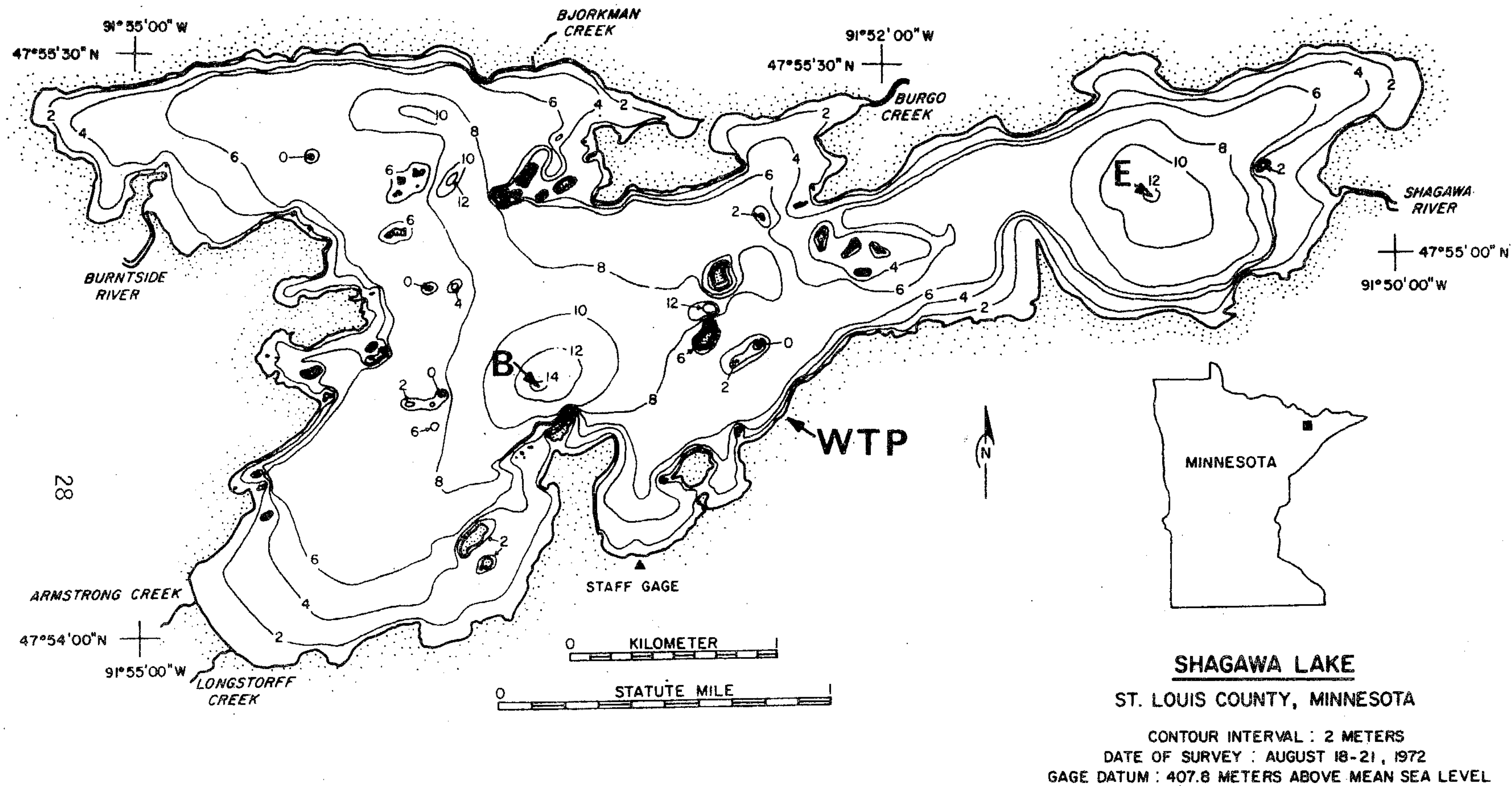


Figure 1 Bathymetric map of Shagawa Lake showing sampling locations, Brisson's Point (B) and East End Deep (E), and wastewater treatment plant (WTP).

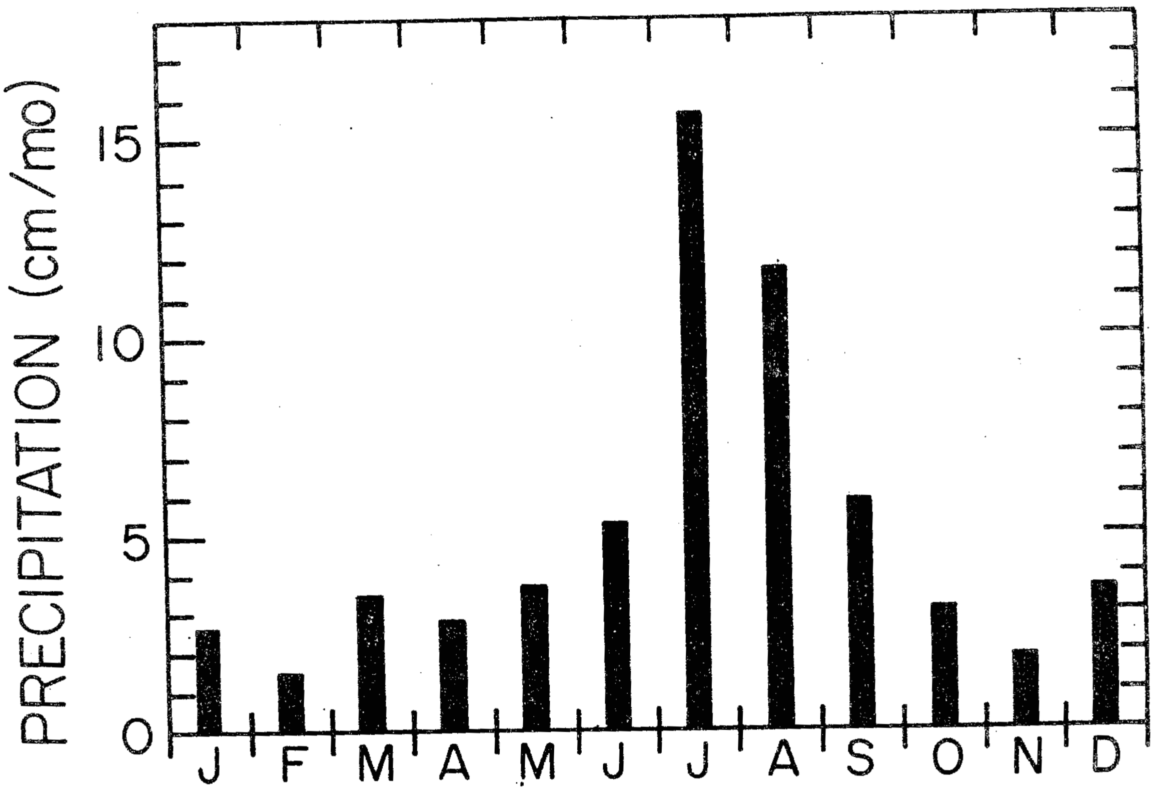


Figure 2 Seasonal variation of precipitation (cm/mo) for 1972. (National Weather Service data).

During 1972 water inflow was 6.2×10^7 m^3, approximately 70% from Burntside River and the remainder from minor tributaries, wastewater, indirect flow, and precipitation. The U.S. Geological Survey determined that groundwater flow was negligible. The outflow was 6.7×10^7 m^3. Malueg, et al. (1975) present details of the water budget for the years 1967-1972. The water retention time (based upon outflow) was 0.79 yr during 1972. Figure 2 summarizes seasonal variation in rainfall.

Water currents have not been surveyed extensively. Some dye studies indicate that treated effluent may move along the shore to the east while other dye studies show movement towards the central basin of the lake. Aerial photographs generally show surface algae "streaming" from west to east in the lee of islands.

Limnological Characterization of Shagawa Lake

Physical, chemical, and biological sampling and analytical techniques are summarized in Larsen and Malueg (1975). Most of the data reported herein are from that paper and, as such, represent only one sampling station, Brisson's Point (Figure 1). Variables were monitored approximately weekly at 1.5-m depth intervals.

During the ice-covered months, temperatures ranged from near 0°C at the surface to slightly greater than 5°C at the bottom (Figure 3a). After the ice broke up in May, the lake rapidly warmed. A thermocline developed in June and deepened during the summer, although thermal gradients were slight. Late summer surface temperatures exceeded 20°C while bottom temperatures were as high as 15°C. Fall circulation began in early September, and the lake froze over in mid-November.

The specific conductance was about 65 μmhos/cm during fall circulation (Figure 3b); values as high as 150 μmhos/cm were observed in the anoxic deep water during the winter and summer.

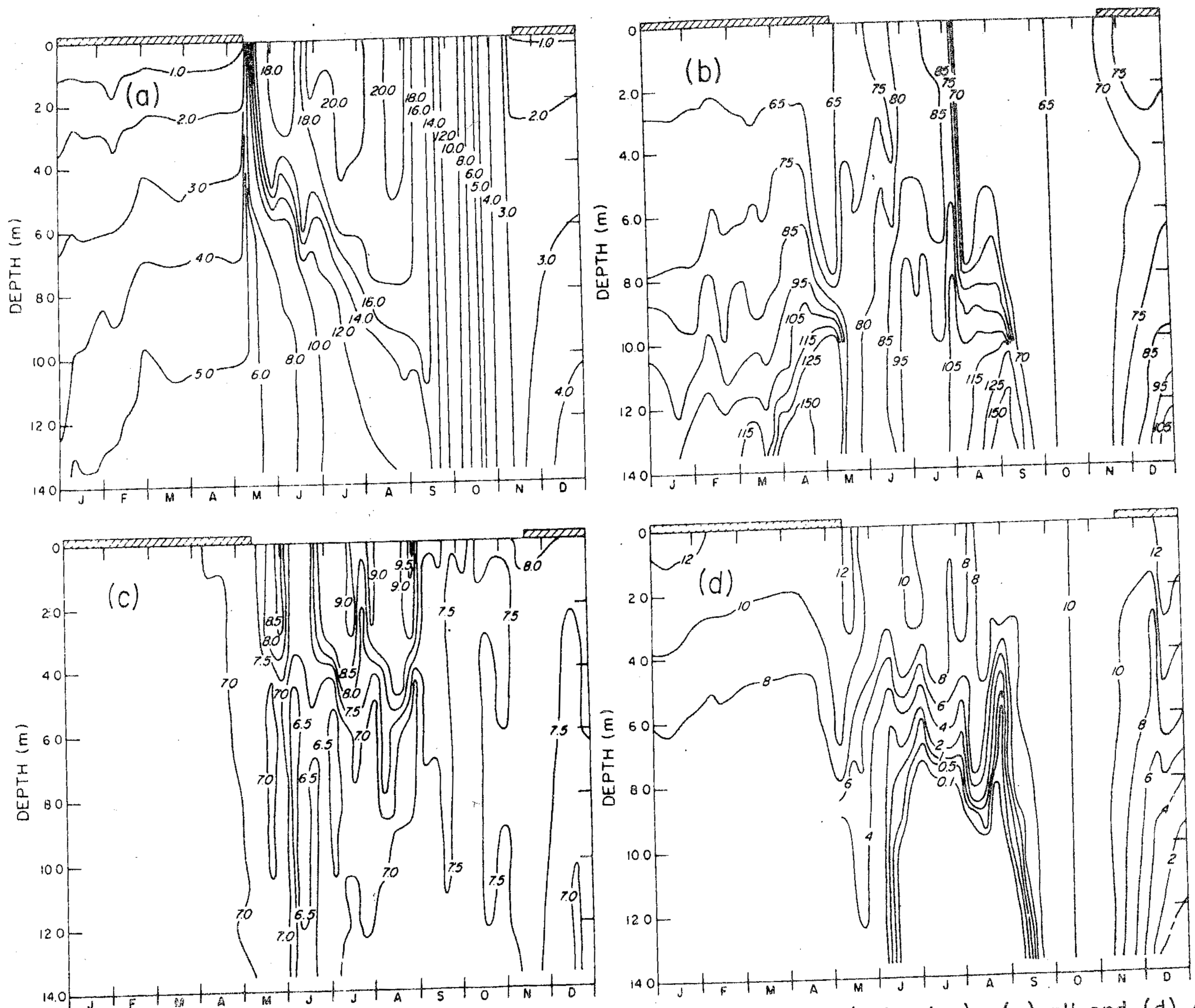


Figure 3. Isopleths of (a)temperature (°C), (b) specific conductance (μmhos/cm), (c) pH and (d) dissolved oxygen (mg/l) for 1972. Cross-hatching indicates ice cover (from Larsen and Malueg, 1975).

Maximum pH values of slightly greater than 9.5 occurred in surface waters during algal blooms; low values of 6.5 were observed in deep water (Figure 3c). Under ice cover and during fall circulation, pH values were generally 7.0-7.5.

During summer stratification anoxic conditions developed below 8 m (Figure 3d). In other years anoxia was observed in the bottom waters during February and March. Oxygen supersaturation often existed in surface waters during summer algal blooms.

The average total alkalinity concentration (as $CaCO_3$) was about 22 mg/l during the fall circulation (Figure 4a). Summer values ranged from 17 to more than 20 mg/l in surface waters, slightly exceeding 40 mg/l in deeper anoxic waters.

Total phosphorus concentration in surface waters ranged between 0.025-0.05 mg/l during most of the year but increased to 0.075 mg/l prior to fall circulation (Figure 4b). Bottom concentrations exceeded 1.0 mg/l during anoxic periods. Soluble reactive phosphorus concentrations reached 0.6 mg/l in the bottom waters during anoxic periods but were depleted in surface waters during most of the summer months (Figure 4c). Winter concentrations in surface waters often were greater than 0.020 mg/l.

Inorganic nitrogen concentrations were high during the ice-covered interval, sometimes exceeding 0.20 mg/l, but depletion occurred during summer months. Nitrate and nitrite (Figure 4d) were undetectable during summer throughout the lake, but ammonia (Figure 5) increased slightly in the surface waters and to more than 1.0 mg/l in the anoxic bottom waters prior to fall circulation.

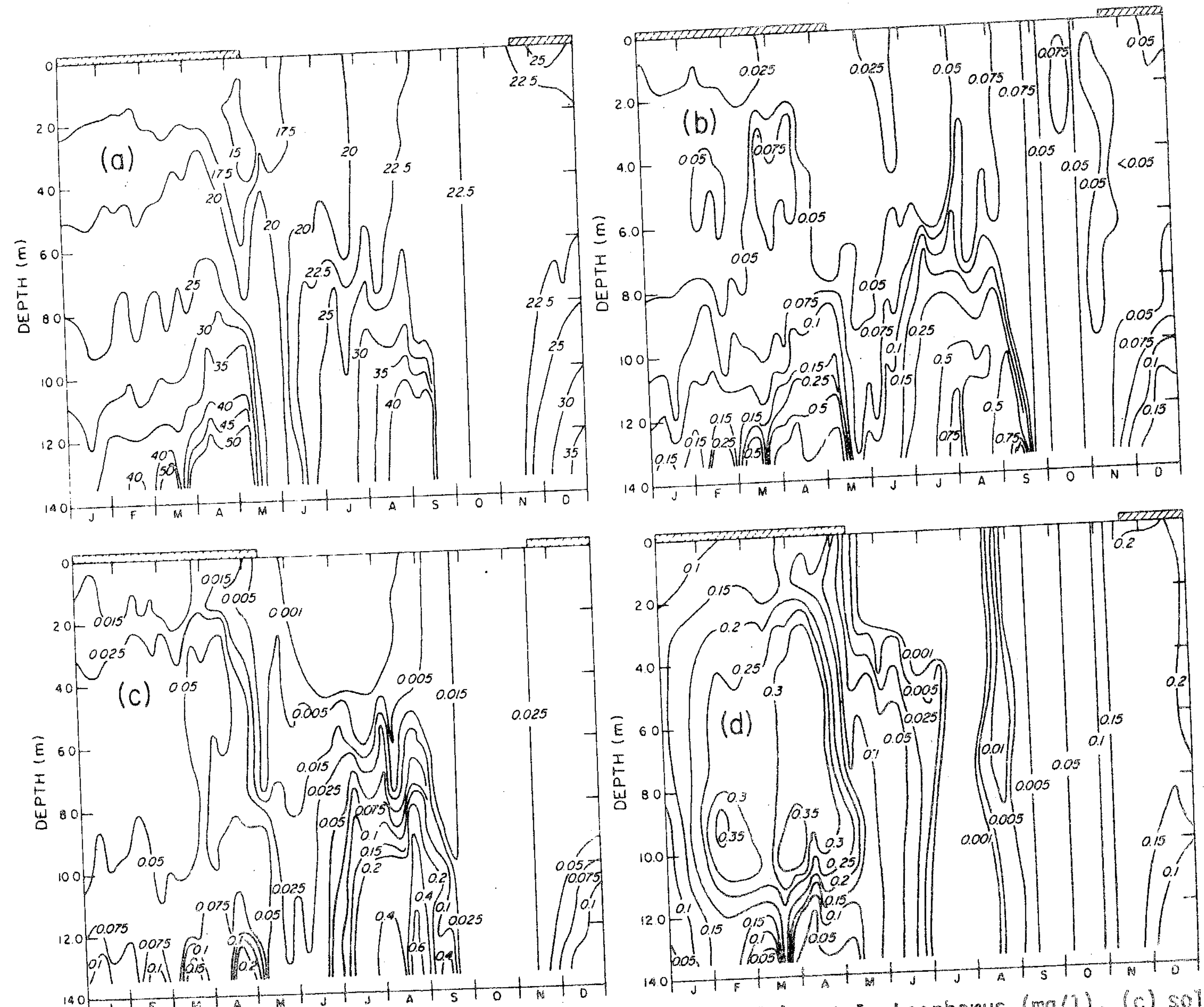
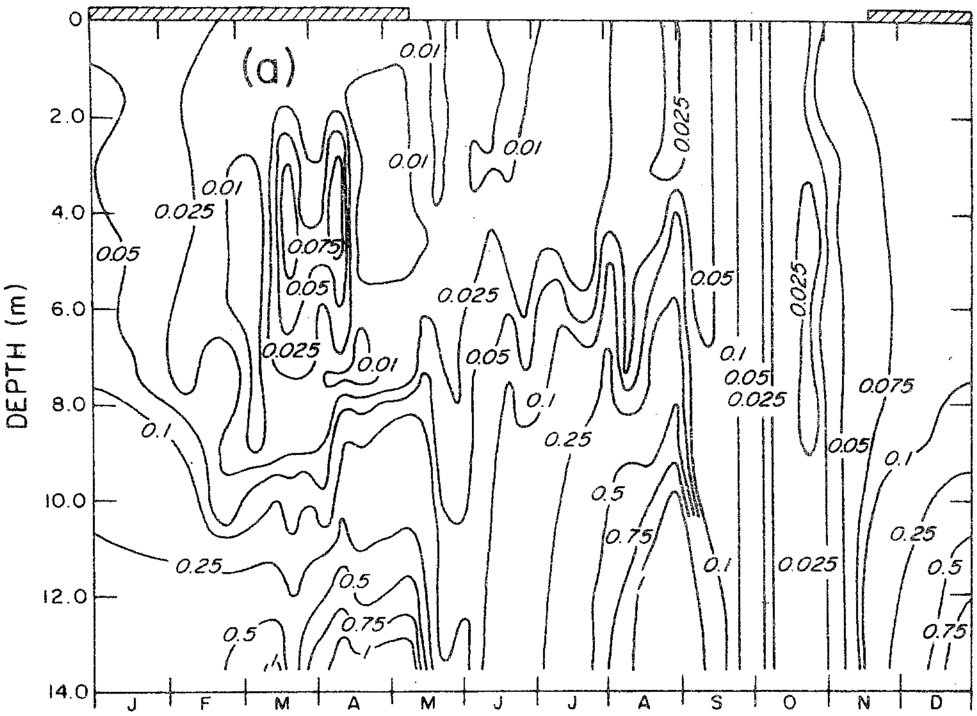


Figure 4. Isopleths of (a) total alkalinity as $CaCO_3$ (mg/l), (b) total phosphorus (mg/l), (c) soluble reactive phosphorus (mg/l), (d) nitrate plus nitrite (mg/l) for 1972. Cross-hatching indicates ice cover (from Larsen and Malueg, 1975).



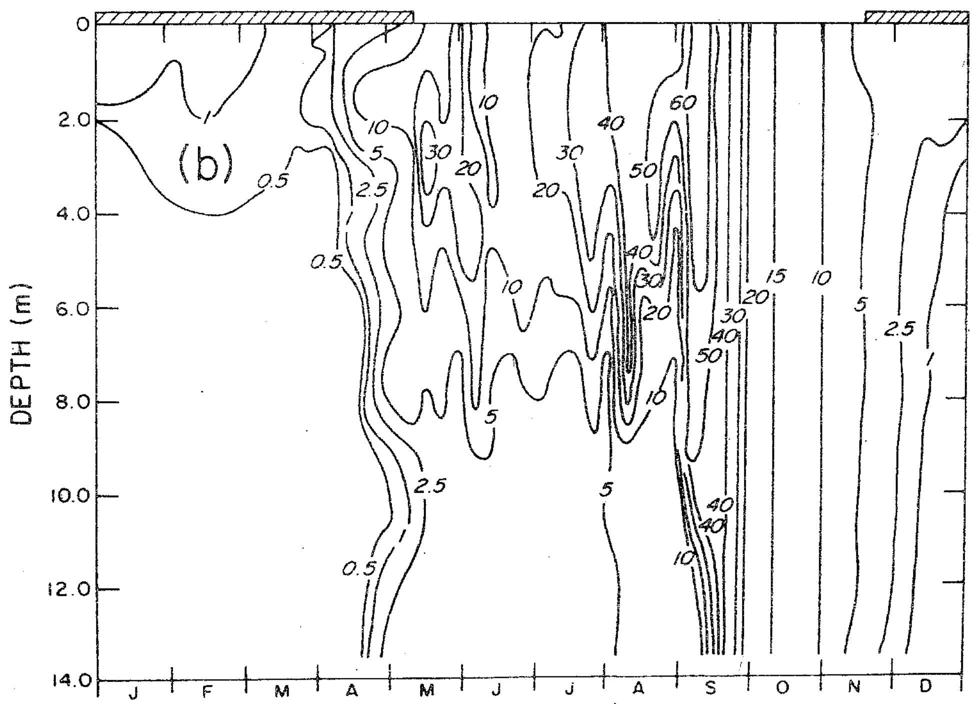


Figure 5 (top) Isopleth of ammonia nitrogen (mg/l) for 1972. Cross-hatching indicates ice cover (from Larsen and Malueg, 1975).

Figure 6 (bottom) Isopleth of chlorophyll a (μg/l) for 1972. Cross-hatching indicates ice cover (from Larsen and Malueg, 1975).

Major cation concentrations (Ca^{++}, Mg^{++}, K^{+}, and Na^{+}) are summarized in Table 1. Table 2 summarizes trace element concentrations.

Average chlorophyll a concentrations (Figure 6) increased in surface waters from 1 μg/l under ice-cover to 30 μg/l during the spring maxima, then decreased to about 10 μg/l. A summer bloom developed later raising the average chlorophyll value to about 60 μg/l. Following fall-overturn, values declined to 1 μg/l under ice-cover. Minimum secchi disc depth was 1.1 m, maximum was 6.1 m, and the average for the open water season was 2.3 m.

Primary productivity profiles were obtained weekly or bi-weekly at one station (E) using the dissolved oxygen, light/dark bottle technique. Measurements were conducted over a four-hour interval from 1000-1400 hrs (CST). Maximum areal productivity values of 260 mg $C/m^2/hr$ were attained in late summer (Figure 7) corresponding to peak chlorophyll a concentrations. Extrapolation of four-hour values to daily values -- assuming productivity proportional to incident solar radiation and integrating over time -- suggests that approximately 220 g C/m^2 were fixed during the ice free season. Solar radiation weekly averages as measured with a pyranometer are summarized in Figure 8.

Laboratory algal assays (National Eutrophication Research Program, 1971) conducted quarterly during 1972 indicated phosphorus limitation in surface samples; however, nitrogen limitation was observed by assays in other years during the ice free season.

During 1972 the pattern of algal succession was a spring pulse of greens followed by diatoms in early summer and blue-greens during mid to late summer. The greens were dominated by *Chlamydomonas sp.*; the diatoms by *Synedra spp.*; and the blue-greens by *Anabaena circinalis*, *Anabaena spiroides* and *Coelosphaerium naegelianum* in early July, late July, and

TABLE 1. Summary of Major Cation Concentrations (mg/l) for 1972

		3/7	6/6	8/21	11/14
Calcium	OM	-	7.0	8.3	-
	6M	14.2	8.0	8.9	9.1
	12M	25.0	8.0	11.1	9.9
	Bright (1968)	10.2			
Potassium	OM	-	0.5	0.6	0.7
	6M	0.6	0.6	0.6	0.7
	12M	0.9	0.6	0.7	
	Bright (1968)	0.78			
Magnesium	OM	-	1.4	1.4	1.7
	6M	2.0	2.1	2.1	2.1
	12M	2.8	2.1	2.5	2.1
	Bright (1968)	3.04			
Sodium	OM	-	1.4	1.4	1.7
	6M	1.5	1.5	1.5	1.6
	12M	2.3	1.4	1.6	1.6
	Bright (1968)	1.6			

TABLE 2. Summary of Trace Element Concentrations (mg/l) for 1972

Element	Depth	7 March	6 June	21 Aug.	14 Nov.
Aluminum	0M	0.05	0.10	0.21	0.11
	6M	0.05	0.20	0.26	0.16
	12M	0.05	0.20	0.27	0.10
Cobalt	0M	0.002	0.001	0.002	0.001
	6M	0.002	0.001	0.002	0.001
	12M	0.004	0.001	0.004	0.001
Copper	0M	0.0035	0.004	0.004	0.003
	6M	0.003	0.002	0.004	0.004
	12M	0.0035	0.002	0.0035	0.003
	Bright (1968)	0.0062			
Iron	0M	0.180	0.070	0.225	0.083
	6M	0.145	0.280	0.375	0.112
	12M	0.732	0.420	10.3	0.076
	Bright (1968)	0.094			
Lead	0M	0.005	0.005	0.010	0.005
	6M	0.005	0.008	0.005	0.005
	12M	0.005	0.005	0.0075	0.005

Element	Depth	7 March	6 June	21 Aug.	14 Nov.
Manganese	0M	0.0225	0.020	0.067	0.006
	6M	0.040	0.085	0.129	0.006
	12M	1.1	0.135	2.22	0.005
	Bright (1968)	0.0168			
Mercury	0M	<0.0005	<0.0005	-	-
	6M	<0.0005	<0.0005	-	-
	12M	<0.0005	<0.0005	-	-
Molybdenum	0M	<0.01	<0.01	<0.01	<0.01
	6M	<0.01	<0.01	<0.01	<0.01
	12M	<0.01	<0.01	<0.01	<0.01
	Bright (1968)	0.002			
Silicate	0M	4.25	-	0.35	2.05
	6M	5.02	-	1.15	2.07
	12M	8.70	-	3.25	2.07
	Bright (1968)	5.57			
Zinc	0M	0.009	0.012	0.012	-
	6M	0.008	0.007	0.013	-
	12M	0.010	0.008	0.010	-
	Bright (1968)	0.0116			

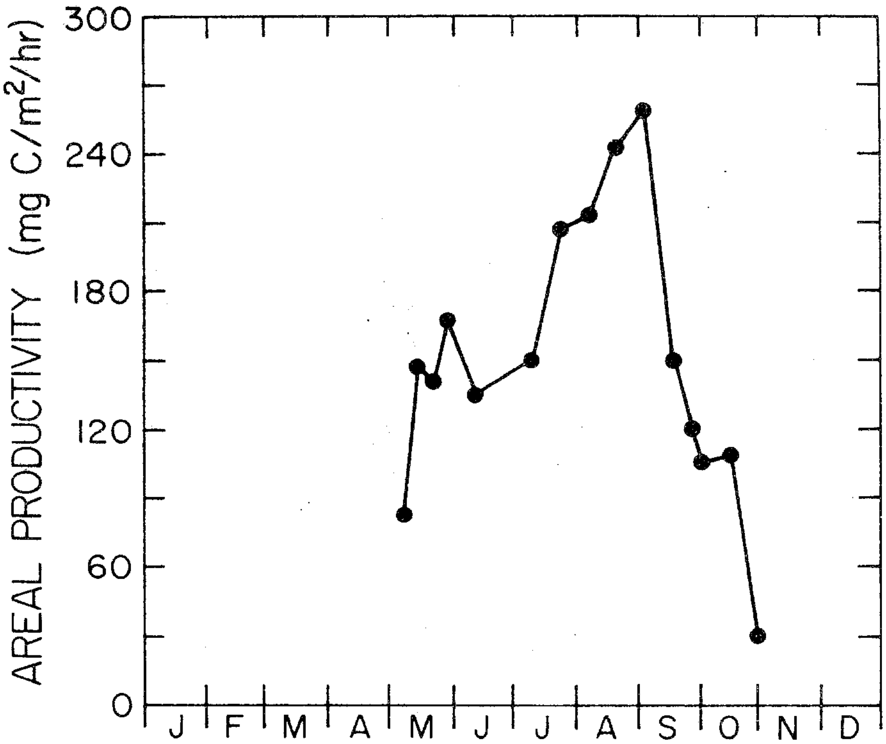


Figure 7 Areal productivity (mg C/m^2/hr) for 1972.

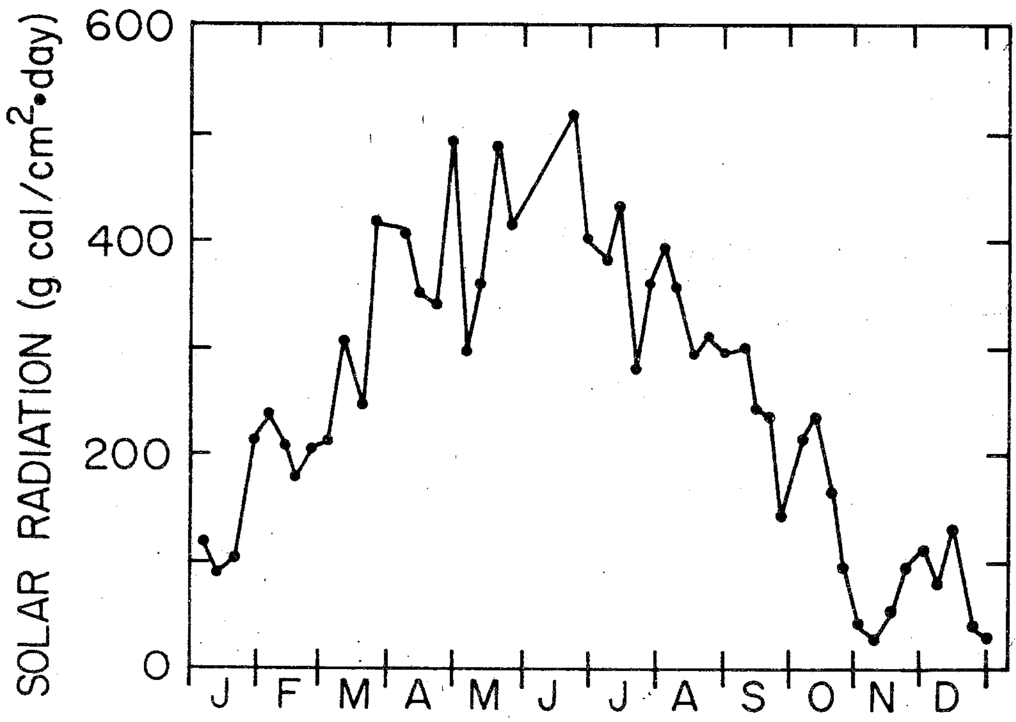


Figure 8 Solar radiation, weekly averages, (g cal/cm^{2}sec) for 1972.

late August, respectively (Schults et al. 1975). Protozoans and rotifers were the most numerous zooplankton in Shagawa Lake, followed by copepods and cladocerans (Figure 9). Maximum production of benthos, dominated by Chaoborus spp., occurred in October when total organisms reached $20 \times 10^3/m^2$ (Figure 10).

Macrophytes covered less than 1% of the lake surface. Dominant species of emergent vegetation included the bur-reed (*Sparganium eurycarpum*), waterlily (*Nuphar sp.*), pondweed (*Potamogeton richardsonii*), water weed (*Elodea sp.*), and bushy pondweed (*Najas sp.*).

Dominant fish included cisco (*Coregonus artedii*), walleye (*Stizostedion vitreum*), yellow perch (*Perca flavescens*) and the rock bass (*Ambloplites rupestris*).

Schults et al. (1975), present further discussion of the phytoplankton, zooplankton, benthos, fish, and macrophyte communities.

Table 3 summarizes nitrogen and phosphorus budgets (see Malueg et al. 1975 for details). Wastewater accounted for about 80% of total phosphorus entering the lake, while tributaries contributed about 15%. On the other hand, waste discharges accounted for only about 27% of the nitrogen entering the lake while tributaries supplied 60%. About 50% of the phosphorus and 16% of the nitrogen were retained by the lake-sediment system. The amount of nitrogen gain or loss by nitrogen fixation or denitrification was not determined.

Discussion

Based on the multitude of measured limnological variables, Shagawa Lake was classified as eutrophic. For example, during the summer it exhibited high pH in the epilimnion, anaerobic conditions in the hypolimnion, low

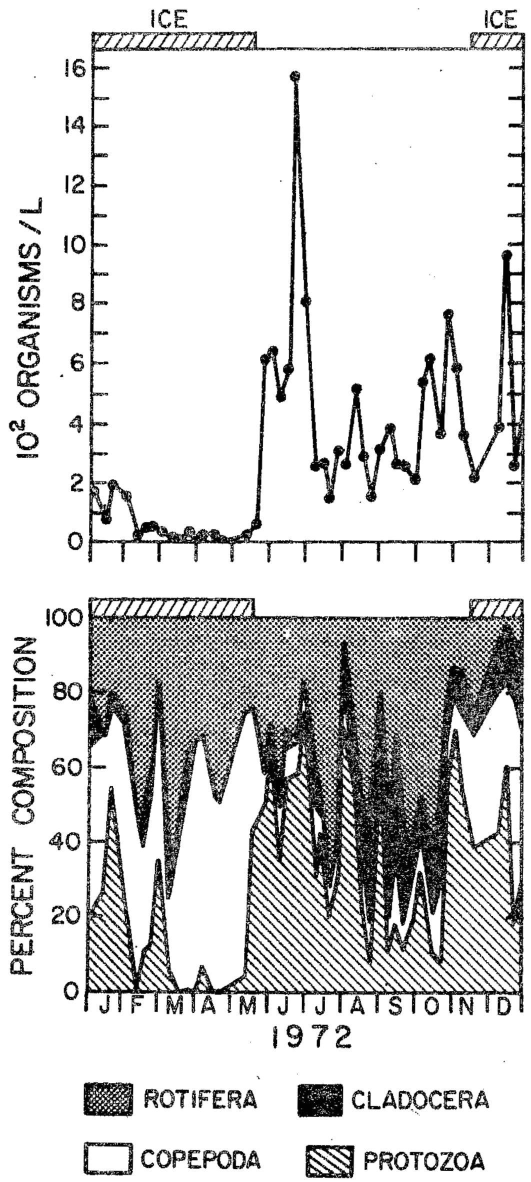


Figure 9 Shagawa Lake zooplankton and group composition for 1972, vertical haul (from Schults, et. al., 1975).

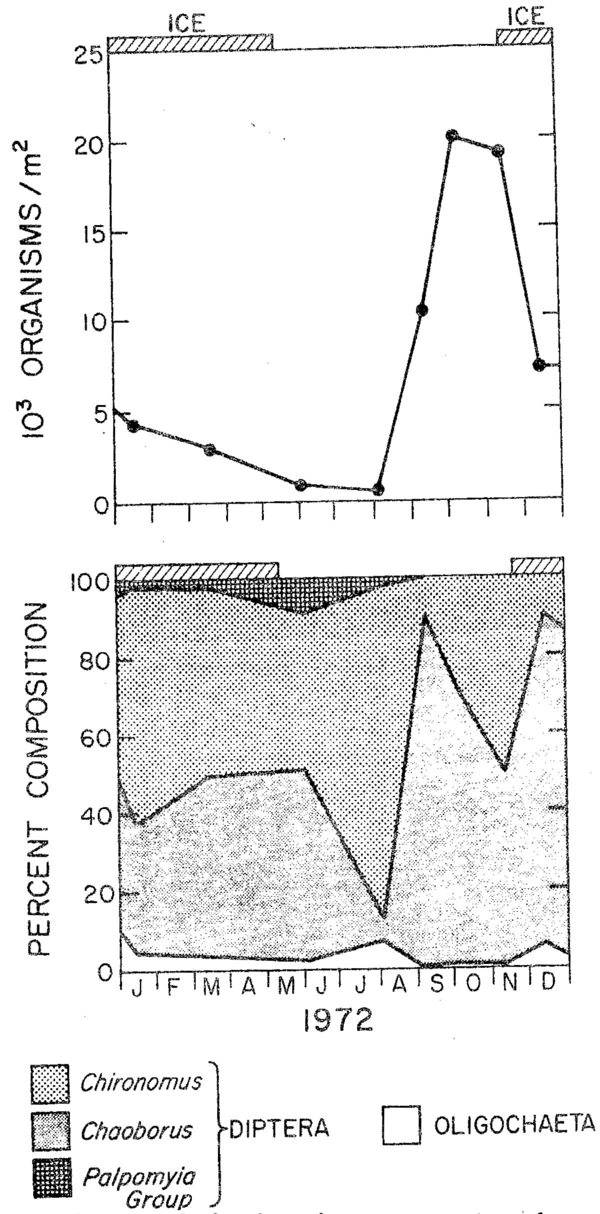


Figure 10 Shagawa Lake benthos concentration and group composition for 1972, Ekman dredge sample.

Table 3: Summary of P and N Budgets for 1972

PHOSPHORUS

Source		kg/yr
Wastewater discharges		5180
Tributary runoff		910
Precipitation		80
Groundwater		--
Other (direct runoff - 60; excess drinking water - 10)		70
	Total	6240
Sink (outflow)		3140
	% retention	50

NITROGEN

Source		kg/yr
Wastewater discharges		19300
Tributary runoff		43300
Precipitation		5400
Groundwater		--
Other (Direct runoff - 2800; excess drinking water - 1100)		3900
	Total	71900
Sink (outflow)		60400
	% retention	16

Secchi disc values and large variations of nitrogen and phosphorus with time and depth. Summer phytoplankton were predominantly blue-green algae, and chlorophyll values reached about 60 μg/l.

On the basis of data from 17 Wisconsin lakes, Sawyer (1947) indicated that 0.01 mg/l of inorganic phosphorus and 0.3 mg/l of inorganic nitrogen at the time of spring overturn are critical values, above which blooms can be expected. In Shagawa Lake the springtime values of soluble reactive phosphorus and inorganic nitrogen were generally at or above these levels.

Vollenweider (1968) proposed criteria for classifying the trophic status of lakes on the basis of specific loading rates of nitrogen and phosphorus normalized to mean depth and surface area. The observed phosphorus and nitrogen loadings to Shagawa Lake were 0.68 $g/m^2/yr$ and 7.82 $g/m^2/yr$ respectively, well above the threshold values for eutrophic lakes.

Vollenweider (1974) later refined his relationship to take into consideration the mean hydraulic retention time of the body of water as well as the mean depth. Figure 11 shows 1972 phosphorus loading for Shagawa Lake plotted in this manner.

Approximately 80% of the phosphorus entering Shagawa Lake and 27% of the nitrogen were attributed to the municipal wastewater. In early 1973, processing of the wastewater by tertiary treatment was initiated, reducing the phosphorus loading to the lake by 80%. The trophic condition of Shagawa Lake is changing from eutrophic to mesotrophic as a result of the greatly decreased loading of this critical nutrient.

During periods of anoxic conditions in the bottom waters, phosphorus is released from the sediments. This phosphorus can be transported throughout the lake as the thermocline breaks up during passing storms and thus

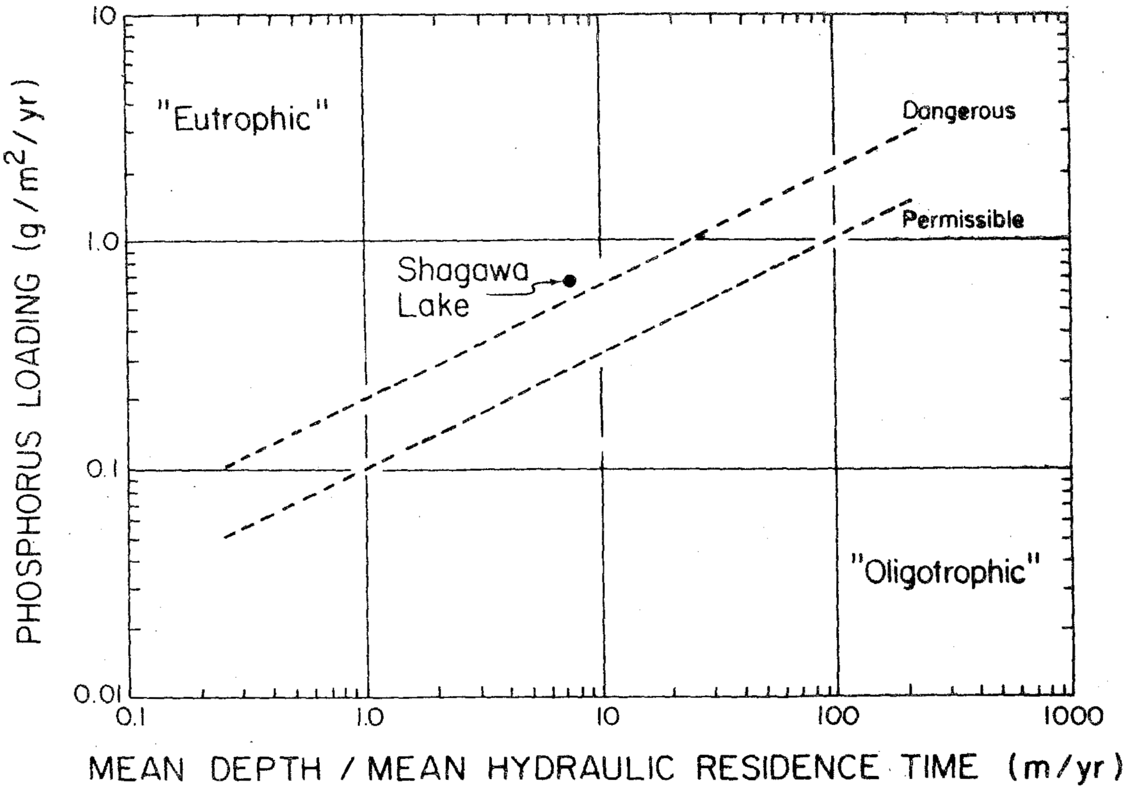


Figure 11 Phosphorus loading versus mean depth/mean hydraulic residence time for 1972 (Vollenweider, 1974).

can be made available for algal growth. Mass balance estimates demonstrate this during July-August when internal loading of phosphorus was of a considerable magnitude and did significantly increase the concentration of phosphorus in the upper waters during that time (Larsen et al., 1975). This nutrient transport process has also been reported for Lake Mendota by Stauffer and Lee (1973).

SUMMARY

This description of Shagawa Lake, Minnesota includes limnological data obtained during 1972. Because the lake has received municipal wastewater for 75 years, it is culturally eutrophic, a condition extremely rare for a lake in this region of Minnesota. During the past eight years, the Environmental Protection Agency has intensively studied Shagawa Lake to evaluate lake restoration by wastewater phosphorus removal. A data summary of Shagawa Lake and its drainage basin is presented in Table 4.

Acknowledgements

The authors are indebted to many individuals of the Eutrophication and Lake Restoration Branch, both at Corvallis, Oregon, and Ely, Minnesota, who gave freely of their time and data, and without whose assistance this report would not have been possible.

Table 4

1972 DATA SUMMARY FOR NORTH AMERICAN PROJECT

Lake name	Shagawa Lake, Minnesota, USA	
Trophic state	Eutrophic	(oligo., meso., eutro)
Lake type	Lake	(lake impound., estuary)
Drainage area	270×10^6	(square meters)
Lake surface area	9.2×10^6	(square meters)
Mean depth	5.7	(meters)
Retention time	∿1	(years)
Mean alkalinity	22 (fall circulation)	(mg/l)
Mean conductivity	60 (fall circulation)	(umhos/cm)
Mean Secchi disk	2.3 (ice-free period)	(meters)
Mean dissolved phosphorus	0.021	(mg/l)
Mean total phos.	0.055	(mg/l)
Mean inorgan. nitrogen	0.160	(mg/l)
Mean chlorophyll a	15 (annual value) 24 (ice-free period)	(ug/l) (uncorrected)
Annual productivity	220	(gr/meter^2/year)
Phosphorus loading point source	5,100	(kg/year)
non-point source	1,150	(kg/year)
surface area loading	0.68	(gr/meter^2/year)
Nitrogen loading point sources	20,000	(kg/year)
non-point sources	52,000	(kg/year)
surface area loading	7.8	(gr/meter^2/year)
Degree of oxygen depletion in hypolimnion	0.0	(mg/l)

Any other data you feel important

Above mean phosphorus and nitrogen values are volume weighted means based on samples taken approximately weekly at 3 stations at 1.5 m depth intervals.

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LAKE SALLIE, MINNESOTA

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I. INTRODUCTION

Lake Sallie has received nutrients originating in the City of Detroit Lakes sewage for more than 70 years. Wastewater discharges, treated and untreated, have passed through a natural lake and an impoundment enroute to Lake Sallie. In recent years sewage treatment has been very effective for such parameters as BOD and coliforms. This lake also has 168 septic tanks draining toward it.

II. GEOGRAPHIC DESCRIPTION

A. Latitude 46°36'00"N
   Longitude 95°54'12"W
B. Altitude 399 meters (1,309 feet) above mean sea level
C. Catchment Area 1,543.6 km^2 (382,399 acres)

D. General Climatic Data

Mean monthly temperatures have ranged from -18.7C (January) to 21.2C (July). Precipitation since 1968 has varied from 49.53 - 70+ cm per year. Lake Sallie is usually ice-covered from mid November until April. Ice has varied from 15-60 cm in thickness, and has been covered with up to 35 cm of snow. Prevailing wind direction during open water seasons is usually NNW, but strong south winds occur from time to time. Evapotranspiration usually exceeds precipitation by about 25 dm (10 inches) per year, but in the 1973-74 water year it exceeded precipitation by only 5 cm.

E. General Geological Characteristics

Topography of this area was primarily formed by Pleistocene glaciation. Four lobes of the Wisconsin ice sheet advanced into Minnesota, and two, the Wadena and Des Moines lobes, formed this watershed. The Wadena lobe, moving westward, formed a hilly region to the east which was later overridden by the Des Moines lobe moving east and carrying grey till, which was deposited on the moraine when the Des Moines lobe withdrew. Outwash areas were formed to either side.

Soil in the morainic eastern 1/3 of the watershed is medium textured sandy loam which developed from calcareous glacial till; that in the central part consists of coarse to medium textured well drained materials formed from glacial outwash; and that in the western 1/3 is dark, well drained glacial till. Outwash deposits are from 1 to 24 m thick, and the glacial till exceeds 91 m.

Large ice blocks broken off the Des Moines lobe were covered or partially covered by outwash and their melt formed Lake Sallie and others in the watershed.

These lakes are circular or elliptical of the type called kettle holes.

F. Vegetation

The catchment area has the following cover:

1. Forests 23%
2. Water areas and marshlands 29%
3. Pastures and croplands 45%
4. Urban and residential areas 3%

Forest cover is largely deciduous, containing oak, maple, aspen, birch, basswood, cottonwood, ash, and scattered conifers. Marshes are largely covered with cattails and bulrushes, but some wild rice is present. Agricultural lands are devoted to small grains, hay, and pasture.

P. Population

This watershed is a popular vacation area and a large share of its population is transient during open water seasons. The City of Detroit Lakes had 7,000 residents in 1970 and the permanent population of suburban dwellings was around 2,000. Lake Sallie has 168 cottages on its shores that house mostly temporary residents.

H. Land Usage

Fifty two percent of the land (forest, water, and marshlands) is used for recreation. Forty five percent is used for farming - small grains, hay, livestock pasture, and turkey rearing; 3% is residential and urban.

I. Use of Water

Groundwater supplies all domestic and industrial water, and surface waters are used almost exclusively for recreation - swimming, boating, water skiing, fishing, etc. Known groundwater consumption is about 3,800 m^3 (1 million gallons) per day; many residences have private wells and withdraw unknown quantities. A limited amount of commercial fishing is intermittently carried on in Lake Sallie in autumn.

J. Sewage and Effluent Discharge

As previously indicated, 168 cottages have septic tanks draining toward Lake Sallie. Municipal sewage from the City of Detroit Lakes goes to a conventional treatment plant (settling, sludge digestion, biofiltration). This plant effluent goes to an aeration pond whose effluent passes into a stabilization pond, which overflows to a natural peat area that discharges to a natural lake, Lake St. Clair. This lake overflows into a ditch that joins the Pelican River above Muskrat Lake (see Figure 1).

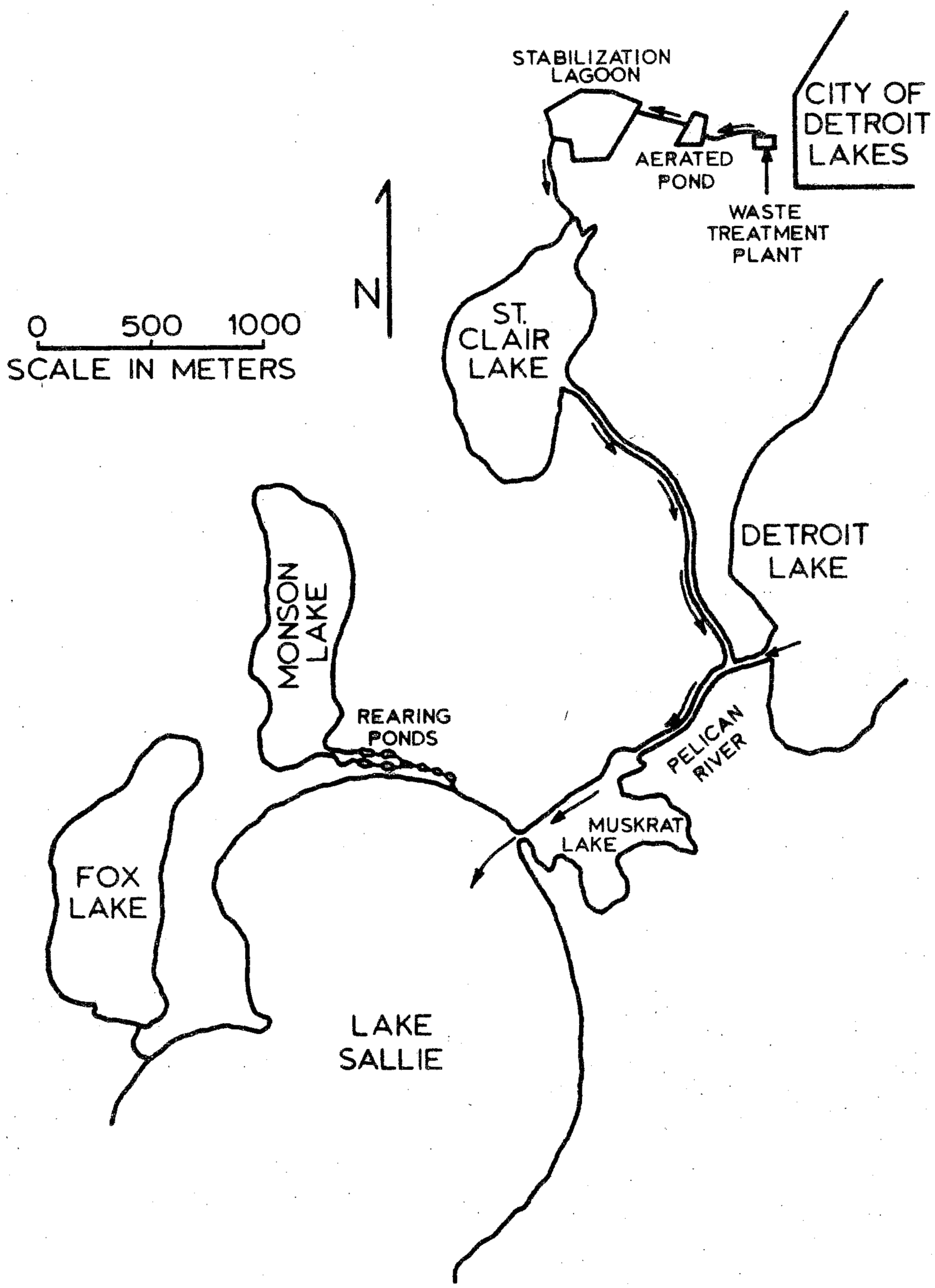

Figure 1. Flow route of wastewater effluent from Detroit lakes to Lake Sallie.

III. MORPHOMETRIC AND HYDROLOGIC CHARACTERISTICS

A. Surface Are 5.3 km^3 (1,310 acres)
Maximum Length 3.32 km (2.06 mi.)
Maximum Width 2.01 km (1.25 mi.)

B. Water Volume 33,700,262 m^3 (27,318 acre feet)

Regulating structures across the Muskrat Lake outlet (see Figure 1) permit control of Pelican River inflow into Lake Sallie during all but extremely high runoff periods. Operation usually strives to maintain the lake at the "normal" overflow level, but inflow is sometimes insufficient and many times in excess of manageable quantities.

C. Maximum Depth 16.5 m (55 ft)
Average Depth 6.35 m (21 ft.)

D. Exceptional Depths

Deep pockets underlie very small portions of the surface area of Lake Sallie (Figures 2). The area north of the hilus in the bean shaped lake contains most deep water, and the area south of this point is practically flat between shoreline slopes. Percentages of surface area lying between selected depths are:

Depths	% of area
0-1.52 m (0-5 ft)	15.85
1.52-3.05 m (5-10 ft)	19.81
3.05-4.57 m (10-15 ft)	7.36
4.57-6.09 m (15-20 ft)	6.80
6.09-9.14 m (20-30 ft)	44.71
9.14-12.19 m (30-40 ft)	4.90
12.19-15.24 m (40-50 ft)	0.38
15.24-16.50 m (50-55 ft)	0.19

E. Ratio of Epi- over Hypolimnion

This varies from year to year and during any one year. The thermocline has disappeared and reappeared during some summers, and it generally tends to sink as summer progresses. Epi-/hypolimnion quotients have ranged from less than 1 to about 18 at the onset of stratification and have usually increased to 250 or more before the disappearance of the thermocline.

F. Duration of Stratification

1969 - July 15 to September 14
1970 - June 6 to 26
1971 - June 3 to 10
June 15 to July 21
August 12 to 20
1972 - June 8 to August 4
August 14 to 22

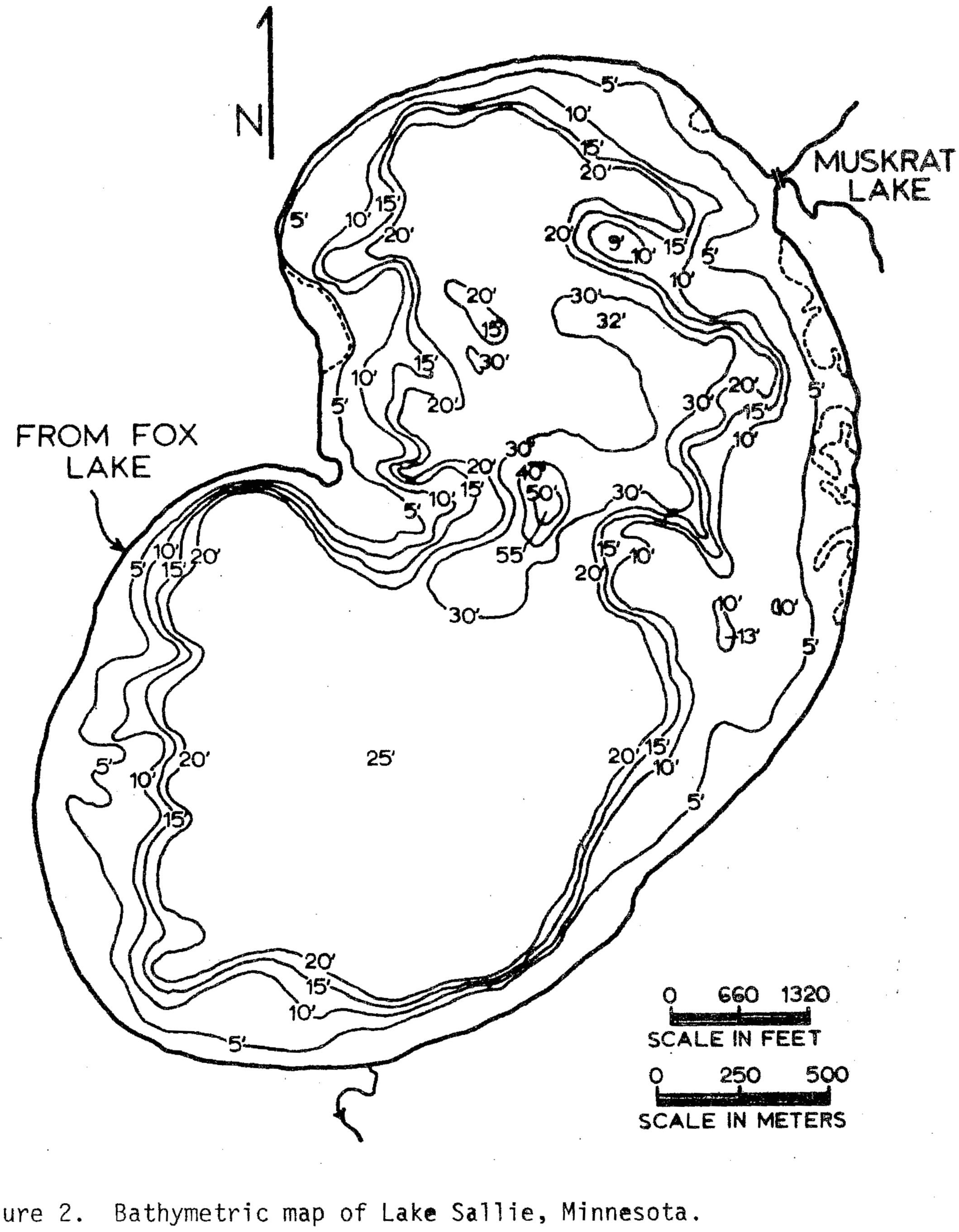

Figure 2. Bathymetric map of Lake Sallie, Minnesota.

1973 - June 6 to September 7
1974 - July 10 to July 31
August 21 to 28

G. Nature of Lake Sediments

Shoal areas, about 45% of the bottom, are largely sand, and deeper regions are mostly covered with silt and clay. Particulate organic matter overlies sand in shoal pockets and entire bottoms in deepest areas. Shallows have bottoms consisting of 75% sand, but deeper regions have but 25%.

H. Seasonal Variation of Monthly Precipitation

Generally, highest precipitation has occurred in May, June, July and August, and lowest in December-February. During some years (1971, 1972, 1973) precipitation was high in late summer and early autumn.

I. Inflow and Outflow of Water

	1969	1970	1971	1972	1973
Inflow (m^3 x 10^6)					
Surface inflow	20.22	17.84	17.85	17.39	26.47
Ground inflow	2.49	1.24	2.96	3.10	4.18
Total inflow	22.71	19.08	20.81	20.49	30.65
Outflow, surface	22.71	19.08	20.81	20.49	30.65*

* 3 mos. estimated

J. Water Currents

Other than in the immediate vicinity of the Pelican River inlet, all currents are wind generated during open water seasons. Maximum waves are produced by southern and NNE winds. Pelican River and other inflows are more readily detected under ice when they escape wind mixing.

K. Water Renewal (Retention) Time

Retention time (all inflow) has ranged from 1.09 to 1.76 years as shown below.

Water Year	Years Detention All Inflow	Years Detention Surface Inflow
1969	1.48	1.66
1970	1.76	1.88
1971	1.61	1.88
1972	1.64	1.94
1973	1.09	1.27

IV. LIMNOLOGICAL CHARACTERIZATION

A. Physical

1. Temperature

Water has responded rather quickly to seasonal air temperature changes, and surface and bottom waters have usually differed in summer and always in winter. In winter water has usually been about 3°C warmer at bottom than at surface, but during periods of summer stagnation it has been as much as 10°C cooler near the bottom. Temperature has ranged from 0-27°C. For data on stratification see F under III above.

2. Conductivity

This parameter has varied by as much as 50 umhos/cm in different lake surface areas on the same date. It has generally been lowest at the outlet (240-280 umhos/cm) and highest (290-360) at the Pelican River inlet. It also increases with depth with advanced thermal stratification (as much as 40 umhos/cm between surface and 9 m).

3. Light

Surface light intensity measurements have varied from less than 400 to 7,720 foot candles. Declines to less than 5% of surface intensity have usually occurred at 3 m. No light has been observed to penetrate ice cover, even the minimum (25 cm) considered safe for observers, but there is indirect evidence that this has occurred. Light penetration was commonly restricted by plankton, especially blue-green algae, during open water seasons. The 1% incident radiation level usually occurred at 3-3.5 m but in autumn sometimes went as deeply as 8.5 m; very faint light was occasionally detected at 10 m. Red and green wave lengths usually had greater intensity and range than blue, but their penetration was often controlled by dominant phytoplankton pigments. Red penetrated more deeply when blue-greens and diatoms or diatoms alone were predominant, but green reached greater depths when greens and blue-greens or greens alone were dominant.

4. Color

This measurement has not been conducted at Lake Sallie.

5. Solar Radiation

This feature has been recorded since June 1971 with few interruptions. It has been most intense in July and August (daily means of 500-560 ly) and least in December (daily mean 100 ly). The maximum daily figure has been 708 ly in July and the minimum 42 ly in December.

B. Chemical

1. pH

Surface waters have had pH above 8.0 at all seasons, but deepest water has fallen below that level with summer and winter stagnation. Some littoral areas have ranged below and above 8.0 depending on the nature of ground and surface inflow. These data indicate virtual isolation of upper waters from deeper areas with significant decomposition most of the year, widespread photosynthetic dominance in upper waters, and a relatively minor water volume noticeably affected by decomposition. Upon a few occasions pH increases in surface water under ice suggested photosynthesis although light could not be detected there. These elevations were accompanied by oxygen increases.

2. Dissolved Oxygen

Oxygen was never deficient in surface water and its concentration was frequently determined by photosynthesis which often produced supersaturation. Maximum levels were usually produced by littoral macrophytes and attached algae which were often responsible for oxygen pulses. Thermal stratification often occasioned depletion in deeper waters, summer and winter, but oxygen always occurred in limnetic surface water, even under thickest ice and snow covers. Photosynthetic oxygen production occurred under ice cover which has already been mentioned with reference to pH. An increase of 3 mgl* was once noted over a 7-day winter period.

3. Phosphorus

Soluble reactive phosphorus (SRP) was concentrated in deeper waters during stagnation periods, but it tended to disappear from limnetic surface waters during growing seasons, although this rarely occurred in 1969. It was missing in surface water from mid-September 1972 until early March 1973. It was usually rather low in surface water in most areas from (0-0.5 mgl) but it became rather heavily concentrated in some littoral areas, especially under ice cover (maximum 4.8 mgl). Concentration in the Pelican River inlet was invariably greater than that in the outlet.

Total phosphorus variation in general resembled that of soluble reactive phosphorus, but concentration was usually higher. It increased with depth, declined from winter maxima during the growing season, and was always more concentrated in incoming than in outgoing water. Total phosphorus as used here is that secured by oxidation with potassium persulfate.

4. Nitrogen

a. Ammonia Nitrogen

This form of nitrogen was present in every water sample taken from Lake Sallie, no matter what depth or season. It was most concentrated in deeper waters during stagnation periods (up to 3.10 mgl in winter and to 2.6 mgl in the summer). Maximum concentrations in limnetic surface waters (up to 1.25 mgl) occurred during periods of full circulation when accumulations built up in deeper waters were reduced. Surface concentrations varied in different lake regions. Values above 4.0 mgl were observed under ice in ground water inflow. The Pelican River inlet and three other littoral areas had ammonia concentrations

*In this report mgl means milligrams per liter above those of the general lake surface. All such areas had inflowing surface or ground water. Concentration in surface limnetic water was almost without exception greater than 0.1 mgl and usually more than 0.2 mgl. Ammonia nitrogen was generally more concentrated in the Pelican River inflow (Station 1) than in the lake outlet (Station 8).

b. Nitrite Nitrogen

This form was often, but not invariably, found in samples from all lake regions and depths. It disappeared from all sampled areas in August 1969 and from surface waters during much of summer and autumn 1972. Concentration was usually well below 0.01 mgl at all depths during open water seasons, but it increased to 0.1 in the inlet and limnetic waters briefly in September 1970. During stratification, maximums were found at 6 meters under ice and at intermediate to maximum depths with open water.

c. Nitrate Nitrogen

Nitrate was most concentrated under ice cover. It was most abundant at 6 and 8 m during the 1969-70 winter and at 14 m during the 1970-71 winter. The Pelican River inflow generally had higher values than its outflow. NO_3 was rarely absent from surface water during open water, and never under ice cover. Concentration never quite reached 0.4 mgl and was usually less than 0.1 mgl.

5. Alkalinity

Carbonate alkalinity was present in limnetic surface water at all seasons, was absent from deeper waters during stagnation periods, and from some littoral areas at intervals, e.g., in areas with ground water entering under ice cover. It reached 100 mgl (as $CaCO_3$) in Muskrat Lake discharge, and 48 mgl in the limnetic zone. It increased slightly under ice cover in response to photosynthesis.

Minimum bicarbonate concentrations were recorded in surface water, and its limnetic maxima in deeper waters during periods of stratification. Its maximum in the limnetic zone was 230 mgl in winter. Higher values (up to 428 mgl) were noted where ground water inflow was isolated by ice cover in some littoral areas, and in winter Pelican River inflow. HCO_3 was photosynthetically reduced to 124 mgl in upper limnetic water.

6. Calcium and Magnesium

In limnetic areas Ca ranged from slightly less than 60 to 100 mgl, whereas magnesium has varied from slightly more than 100 to 165 mgl. Highest values for each were in deepest water during stagnation periods. Groundwater entering the lake in littoral areas had higher levels, and in it Ca exceeded Mg. Preponderance of Mg in surface waters fed by such groundwater indicates photosynthetic overshadowing of decomposition. Both ions increased when CO_2 appeared in the hypolimnion, but Mg to a greater extent.

C. Biological

1. Phytoplankton

a. Chlorophyll

No chlorophyll determinations have been made to date as this procedure has not entered into project objectives.

b. Primary Production

This feature has been measured over two hour incubation periods using the light and dark bottle technique. Reproducibility was within ± 10 mg C fixed/m^3/hr.

Phosphorus and nitrogen declined along the path of Pelican River inflow, and this was true of the rate of primary production on many but not all dates of measurement. It was often greater in the mid-limnetic area than near the inlet. Horizontal variation in the limnetic zone appeared normal.

Most measurements were made near the center of the limnetic zone. Surface water there has shown maxima in late summer or early fall, but activity has varied considerably over the seasons. Patterns for the euphotic zone have resembled those for surface water, but generally have had fewer and sharper peaks. Means over the euphotic zone were comparable to those of surface water in 1969 and '71, but were much lower than the surface in 1970. Maximum production levels increased following weed harvest which began in 1970 as follows:

	Maximum gross primary production (mg C fixed/m^3/hr)
1969	435
1970	650
1971	645
1972	780
1973	720
1974	970

This would make it appear that there was considerable competition between phytoplankton and macrophytes.

Productivity varied with depth in the euphotic zone and with time of day, but with no consistent patterns. There was no definite relationship with intensity or amount of light on a daily or seasonal basis other than some suppression of activity at noon on bright summer days. Photosynthetic efficiency (mg C fixed/m^3/ly) was usually greatest in late afternoon, and seasonally in autumn.

c. Algal Assays

Collections through 1974 have yielded 131 algal species: 59 green algae, 38 diatoms, 25 blue-greens, 3 dinoflagellates, 1 cryptophycean, 3 euglenophytes, 1 chrysophyte, and 1 xanthophyte.

Annual succession patterns have varied with location in the lake, and have been unstable at periods with changing dominant groups. In the central limnetic zone the most frequent tendency was dominance by diatoms in spring, blue-greens in summer, diatoms again in autumn, and green algae or cryptophyceae in winter. This was by no means a fixed successional order. Blue-greens were sometimes dominant for short periods in winter, diatoms were noted to temporarily or permanently usurp dominance from blue-greens in summer, and diatoms were at times replaced in dominance by blue-greens for varying periods in autumn. Vacillating dominance between diatoms and blue-greens that often occurred in late spring and early fall suggests that competition between them is rather finely balanced and that swings to either side may result from minor environmental changes.

Phytoplankton composition as larger taxonomic groups differed noticeably at any time in varied limnetic and littoral areas, and at different depths in the central limnetic zone. Variation at the generic level was often noticeable during periods of dominance by a single class or order.

d. Count

Concentration has varied from 12,000 to more than 66,000,000 units per liter. Seasonal influences appear to result in spring, summer, and autumn maxima, and winter, late spring, and late summer minima. Diatoms have been largely responsible for spring and fall elevations and blue-greens for major summer growth, but this has not been a fixed pattern. Diatoms once replaced blue-greens in summer. Green algae never contributed significantly to annual maxima but had about 5 peaks per year, three in summer and one each in spring and winter. They were dominant only with low total phytoplankton concentrations in winter. Greatest total concentrations (more than 50,000,000/l) observed (outside plankton drifts) were during diatom dominance in spring, and usually at some distance below the surface where numbers may have been enhanced by settling. Highest summer concentrations of blue-greens were considerably lower, ca 6,000,000/l.

2. Zooplankton

a. Identification and Count

Zooplankton samples have yielded 27 species of Protozoa, 1 nematode, 1 gastrotrich, 20 rotifers, 1 tardigrade, 6 cladocerans, 3 copepods, 1 ostracod, and larval water mites. The most numerous protozoan (Coleps) attained very great concentrations (500,000/l) in deep water each summer. Bosmina longirostris (O. F. Mull) and Chydorus sphaericus (O. F. Mull) have been the most abundant cladocerans, and Trichocerca similis Ehr., T. multicrinis Kell., and Keratella cochlearis (Gosse) the most numerous rotifers. Halteria, Didinium, Strombidium, and Vorticella were common protozoans.

3. Bottom Fauna

To date benthos has not entered into the Lake Sallie study program.

4. Fish

No detailed examination of nekton has been undertaken, but data on number of fish species is available from the Game and Fish Division of the Minnesota Department of Natural Resources. Species assayed for carbon, nitrogen and phosphorus content, and these values as percent of wet weight were:

	% of wet wt.		
Fish Species	C	N	P
Catostomus commersonii (Lac.)	9.97	1.99	0.49
Esox lucius L.	9.11	2.61	0.57
Ictalurus natalis (LeS.)	9.39	2.25	0.54
I. nebulosus (LeS.)	9.31	2.02	0.55
I. melas (Raf.)	9.18	1.97	0.50
Lepomis gibbosus (L.)	8.68	2.06	0.71
L. macrochirus Raf.	8.52	1.84	0.85
Perca flavescens (Mitch.)	9.49	1.83	0.59
Pomoxis negromaculatus (LeS.)	9.76	1.95	0.73
Stizostedion vitreum (Mitch.)	7.91	1.84	0.60

Over the period 1969-73 Ictalurus melas was the most abundant species and accounted for the greatest weight of fish removed by commercial and sport fishermen. Estimates of total biomass of fishes in the lake made by the Game and Fish Division are:

1969 - 347,545 kg.
1970 - 212,090 kg.
1971 - 594,740 kg.

5. Bacteria

Bacterial observations have been limited to forms microscopically identifiable in plankton samples (Sphaerotilus natans). Under ice this species has comprised up to 70% of the plankton population in some littoral areas.

6. Bottom Flora

In addition to macrophytes the bottom bore growths of Chara sp., Cladophora sp., Rhizoclonium SP., and Nostoc. The two filamentous algae were often attached to macrophytes as was Lemma trisulca L., which was also free floating. Spirodela polyrhiza L. and Wolffia columbiana Karst. also occurred as surface floaters.

7. Macrophytes

This population has to date included the following species:

Najas flexiles (Willd.) Rostk. and Schmidt
Potamogeton amplifolius Tuckerm
P. crispus L.
P. filiformis var. Macounii Marong
P. pectinatus L.
P. praelongus Wulf.
P. Richardsonii (Benn.) Rydb.
Ruppia maritima L.
Alisma gramineum var. Geyeri (Torr.) Sam.
Scirpus actus Muhl.
Heteranthera dubia (Jacq.) MacM.
Elodea canadensis Michx.
Vallisneria americana Michx.
Ceratophyllum demersum L.
Myriophyllum exalbescens Fern.
Nuphar variegatum Engelm.
Nymphae tuberosa Paine

Prior to weed harvest macrophytes covered about 34% of the lake area, all above the 3 m contour except *Potamogeton praelongus* which grew down to 6 m. Northern and southern ends of the lake were then dominated by *Myriophyllum* and *P. pectinatus*. Weed harvest began in 1970 and in 1971 all species listed above were still present but noticeably less abundant. *P. pectinatus* was more prominent than *Myriophyllum* and most luxuriant growth had changed from northwestern to west and southern areas. In 1972, following 2 years of weed harvest, *Myriophyllum* was becoming rare, and a previously rare species, *P. crispus*, dominated wide areas in the northern half of the lake. Weed growth in harvestable areas declined each with harvest as shown by total mass removed each year:

1970 - 428,034 kg wet wt.
1971 - 111,064 kg wet wt.
1972 - 59,487 kg wet wt.
1973 - Practically nil in harvestable areas

V. NUTRIENT BUDGETS SUMMARY

A. Phosphorus

	Kg/water year				
Source	1968-69	'69-70	'70-71	'71-72	'72-73
Waste Discharge	10,020	7,060	15,169	20,081	20,519
Land Runoff	100	410	474	176	5,966
Precipitation	-	-	18	9	-
Ground Water	1,345	620	1,480	1,552	2,063
Total	11,465	8,090	17,141	21,818	28,548

B. Nitrogen

Source	1968-69	'69-70	'70-71	'71-72
Waste Discharge	11,360	5,590	10,568	6,930
Land Runoff	290	260	610	94
Precipitation	-	-	175	240
Ground Water	-	-	3,410	8,752
Total	11,650	5,850	14,763	16,016

VI. DISCUSSION

A. Limnological Character

Prior to cultural enrichment Lake Sallie was probably middle-aged (late mesotrophic or early eutrophic as these terms refer to aging). The large epilimnion/hypolimnion quotient that usually occurs does not produce a major isolation of nutrients, and summer stagnation is usually interrupted. It would therefore appear that naturally occurring nutrients were generally available during the growing season, and that quantities tied up in the hypolimnion were often made available at intervals during summer. The major water mass probably supported photosynthesis as it does today, but intensity and persistence were less.

Lake Sallie is a moderately hard, moderately alkaline lake, and neither calcium nor bicarbonate appears limiting to plant growth or photosynthesis. Its epilimnion retains a photosynthetically imparted character the year around, and is evidently influenced by photosynthesis at times under ice and snow cover. Although containing considerable nutrients and highly productive, the major water mass is remarkably free from decomposition effects. Photosynthesis affects all water during full circulation periods and stratification isolates relatively small volumes.

The lake has a detention period of about 1.5 years and the major surface inflow (Pelican River) sometimes forms interflows under ice cover that may pass on through the lake. Ice cover also permits many lesser inflows to demonstrate their individuality which is never evident in wind driven open water. Some inflow induces oxygenless conditions over a few hundred meters of shoreline under ice, but no such effects are evident during open water.

The plankton population is not dominated by any major group for any great length of time. Diatoms have produced more biomass than any other phytoplankton group, and they have been involved in some of the most intense primary production. They have usually been dominant in cooler open water periods, spring and autumn, but have sometimes replaced blue-greens in predominance in summer. Blue-green phytoplankters generally dominate in the hotter months, but have appeared liable to usurpation by diatoms even then. Blue-green maximums have been about 11% of those achieved by diatoms, but their denser populations have been much more noticeable macroscopically. Dominant groups have varied in different lake regions at all seasons, in both littoral and limnetic zones.

B. Delineation of Trophic State

Lake Sallie today may be classed as a culturally enriched, late mesotrophic or early eutrophic lake. The latter two terms are used as they refer to natural aging.

C. Trophic State vs. Nutrient Budgets

Nutrient budgets vary from year to year but they have been adequate to maintain a high level of plant growth. Attempts to reduce nutrients to less productive levels by weed harvest have been a signal failure, as they have removed only a small percentage of the annual phosphorus increment. Removal of all fish and weeds would make inroads on previously accumulated nitrogen, but would not equal any annual phosphorus increment observed to date. The only practicable method for P reduction now appears to be great reduction, or perhaps virtual elimination, of quantities entering in surface inflow. Phosphorus removal procedures are scheduled to be applied to the municipal waste effluent beginning in autumn 1975. Response of the lake to nutrient reduction will be studied concurrently.

Phosphorus loading has varied from 1.52 to 4.16g./m^3/yr., depending upon character of the wastewater effluent and inflow volume. The second Vollenweider number, depth/detention time, is 4.2. Detention time used (1.51 yrs.) is based on both surface and groundwater inflow; surface inflow alone would give a detention of about 1.80 years, and a smaller Vollenweider number ($\bar{z}$/Tw). If data for Lake Sallie are plotted on the L - $\bar{z}$/Tw curve (Figure 3, Vollenweider and Dillon, 1974) any L figure (gP/m^2/yr.) within the above range would place this lake in the eutrophic range, and the higher value (4.16gP/m^2/yr.) would put it well above the "dangerous" limit. Actually, Lake Sallie has developed nuisance conditions each recent year of record that would place it above the "dangerous" condition with P loadings ranging from 1.52-4.16g/m^2/yr. This would suggest that above certain loadings P is no longer controlling or that the Vollenweider-Dillon curve needs further modification.

VII. SUMMARY

Lake Sallie is a kettle hole lake formed by an ice block left in outwash as the Des Moines lobe of Wisconsin glaciation retreated from northwestern Minnesota. The lake now lies in sand and sand and gravel. Its catchment area (1,544 km^2) is covered by forest (23%), water bodies and marshlands (29%), pastures and croplands (45%), and urban and residential areas (3%).

The lake area is a summer vacationland with a large transient population. The nearby City of Detroit Lakes had a 1970 population of 7,000, and 2,000 more reside in suburban areas along lake shores. Lake Sallie has 168 cottages along its shores that house mainly transient residents.

On the basis of age classification Lake Sallie is in a late mesotrophic or early eutrophic state and is culturally enriched by wastewater effluent from the City of Detroit Lakes, septic tanks along its shores, and ground water inflow from agricultural lands and distant residential areas. Waste discharges have put in 7,000-20,000 kg P and 5,000-11,000 kg N per year; surface land runoff has contributed up to 474 kg P and 610 Kg N; direct precipitation has added up to 18 Kg P and 24 kg N; and ground water inflow has supplied up to 1,552 Kg P and 8,750 kg N annually.

Thermal stratification usually comes and goes during summer and early fall, but has endured continuously for about two months. When present the hypolimnion occupies a relatively small volume. During open water seasons and most of the time under ice the chemical nature of the major water mass is of the sort imparted by photosynthesis, and it rarely changes during periods with complete circulation.

The lake is mainly fed by surface inflow (mostly down the Pelican River) but ground water has significant contributions (up to 3.10×10^6 m^3/yr.).

Weed and fish harvest has not removed more than a fraction of annual N and P increments. Complete removal of the biota would eliminate only part of the annual P load, and significant removal of nutrients from inflowing water appears the only practicable method to insure lake recovery.

Profuse weed and phytoplankton populations compete for nutrients; when harvesting reduced weed populations phytoplankton mass and photosynthesis increased. Weed harvest in 1971 and 1972 yielded but 26% and 14%, respectively, of the 1970 crop, and in 1973 weeds in harvestable locations did not merit harvester operation.

Phytoplankton was generally dominated by diatoms in spring and fall and by blue-green algae in summer. Green algae were frequently dominant in winter when total numbers were low. Greatest densities were achieved by diatoms. Blue-greens have not attained more than 11% of maxima reached by diatoms, but have been much more noticeable along shore lines than diatoms, since they are more prone to drift. Before harvesting began macrophytes occupied about 34% of the lake area (down to 3 m except for P. praelongus which grew down to 6 m). The fish population is now dominated by bullheads; the lake was formerly known for its walleye populations.

Phosphorus loading has varied from 1.52 to 4.16g/m^2/yr. and nuisance conditions have occurred each year regardless of loading rate in this range. This suggests that phosphorus is not limiting above a certain loading or that further modifications or factors must be incorporated in loading-detention models.

DATA SUMMARY
FOR
LAKE SALLIE

Trophic state	Eutrophic	(oligo., meso., eutro.)
Lake type	Lake	(lake, impound., estuary)
Drainage area	1,543,600,000	(square meters)
Lake surface area	5,300,000	(square meters)
Mean depth	6.35	(meters)
Retention time	1.51	(years)
Mean alkalinity	162	(mg/l) (1973-74)
Mean conductivity	(280-360) mean 310	(umhos) (1973-74)
Mean dissolved phosphorus	0.130	(mg/l)
Mean total phosphorus	0.34955	(mg/l)
Mean inorgan. nitrogen	0.4437	(mg/l)
Phosphorus loading		
point source	7,060 - 20,081	(kg/year)
non-point source	1,030 - 1,972	(kg/year)
surface area loading	1.52 - 4.16	(gr/meter^2/year)
Nitrogen loading		
point sources	5,590 - 11,360	(kg/year)
non-point sources	4,195 - 9,086	(kg/year)
surface area loading	2.78 - 3.02	(gr/meter^2/year)

THREE OLIGOTROPHIC LAKES IN NORTHERN MINNESOTA

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I. INTRODUCTION

A. Past History

The three lakes (Dogfish L., Meander L. and Lamb L.) described in this report are located in the Superior National Forest near the southern border of the Boundary Waters Canoe Area in northeastern Minnesota. The area in the vicinity of the lakes in uninhabited.

The general area consists primarily of virgin deciduous-coniferous forest. The drainage basin of Dogfish L. has remained virtually undisturbed, and the only significant disturbance in the watershed of Lamb L. has been the construction of a dirt road bypassing the lake. The watershed of Meander L. has been subjected to minor disturbances; the construction of Echo Trail in 1926, the building of a CCC Camp in 1934, selective cutting of pines in the northwestern portion of the basin in 1945, and further cutting about 200 m from shore in the southwestern basin in 1969-1970.

On May 14, 1971, the Little Sioux fire began and in the course of three days burned 5900 ha. The fire killed about 70% of the overstory in the watershed of Meander L. and about 65% of the overstory in the drainage basin of Lamb L. Dogfish L., similar in water chemistry to Meander L. and located 2 km west of the fire perimeter, was selected as a "control" lake.

A cooperative effort was undertaken to investigate the effect of the fire on the watersheds and the lakes. In addition to the results presented here, studies on internal nutrient cycling in the watersheds have been conducted by Dr. H. E. Wright and J. P. Bradbury and J. C. B. Waddington of the Limnological Research Center, University of Minnesota.

Studies on water chemistry and phytoplankton were initiated in May/June 1971 and continued into the winter of 1972/73. Investigations on the hydrology and nutrient budgets of the watersheds and lakes were conducted during the period January 1 - December 31, 1972. Some of the results have been published by Wright (1974), Wright (1975), Bradbury et al. (1974), and Tarapchak (1975).

II. GEOGRAPHIC DESCRIPTION

A. Latitude and Longitude

The lakes are located in northeastern Minnesota and are within 10 km of one another (Table 1 contains latitude and longitude).

B. Altitude of the Lakes

The altitude of the lakes at their surface is approximately 400 m above sea level (Table 1).

C. Catchment Area

The catchment areas (A_d) of Lamb L. and Meander L. are small and nearly equivalent. A_d of Dogfish L. is about one-third the size of the other lakes (Table 1).

D. General Climatic Data

Northern Minnesota has a typical mid-continental climate with cold winters and warm, moist summers. Average monthly temperatures and precipitation records from two weather stations located within 30 km of the lakes are given in Table 2 (data from the Environmental Science Services Administration,

Table 1. Geographic location and description and morphometric and hydrologic characteristics of the three lakes.

	Meander	Dogfish	Lamb
Latitude	48°08'N	48°11.5'N	48°10'N
Longitude	92°8.5'W	92°11'W	92°6.5'W
Altitude Above Sea Level (m)	423	393	376
Drainage Basin Area Ad (ha)	133	59.1	156.2
Lake Surface Area A_o (ha)	36	29.1	39.7
Total $A_d + A_o$ (ha)	169	88.2	195.9
Ratio $A_d + A_o$	2.03	3.70	3.94
Length (m) maximum	1158	885	965
Width (m) maximum	1006	805	708
average	210	117	402
Shore Length ($x10^3$ m)	3.117	2.966	3.318
Volume (10^6 m^3)	1.80	1.164	1.588
Depth (m) maximum	7.0	5.5	5.5
average	5.0	4.0	4.0
Ratio Epilimnion to Hypolimnion	1.55	--	1.68
Retention Time (yrs)	2.7	3.5	2.3

Table 2. A summary of average monthly air temperatures and average monthly total precipitation. First row of values from the Winton Power Plant; second row of values from Crane Lake Ranger Station, Minnesota.

	Temp. (°F) 1971	Precip. (inches)	Temp. (°F) 1972	Precip. (inches)
January	- 4.0 - 4.0	0.61 0.59	- 3.8 - 3.8	1.03 1.18
February	10.2 10.7	1.29 1.31	3.7 2.4	0.61 0.44
March	19.3 20.4	1.02 0.61	18.3 18.9	1.34 1.26
April	37.8 39.2	0.88 1.22	34.8 36.2	1.16 1.25
May	48.6 50.6	2.32 3.11	57.8 58.5	1.46 0.98
June	64.3 65.1	3.97 3.39	62.3 62.5	2.07 3.01
July	63.2 63.1	2.29 1.71	63.8 63.9	6.15 6.47
August	63.0 64.9	2.18 1.97	64.6 64.7	4.62 2.34
September	57.0 57.1	4.28 4.38	50.7 52.3	2.34 3.20
October	47.0 48.2	4.39 6.65	39.6 41.1	1.18 1.62
November	26.2 26.1	1.84 2.09	25.2 25.8	0.75 0.52
December	11.2 10.9	0.87 0.71	4.5 4.3	1.39 0.95
Annual Mean	37.0 37.7	25.9 27.4	35.1 35.6	24.0 23.2

U.S. Department of Commerce). Mean annual precipitation is about 25 inches, and annual temperatures average about 36°F. The lakes are ice-covered from November through early April. The prevailing winds are from the southwest during the open-water season. Mean annual evaporation just exceeds mean annual precipitation in the region.

E. General Geologic Characteristics

The lakes lie within the Vermilion granite batholith. This massive Precambrian intrusion is composed of 25% quartz, 50% potassium feldspar, 20% oligoclase, and 2% biotite, and minor amounts of zircon, allanite, muscovite, and magnetite (Grout 1925). Granite outcrops are visible along ridges and knobs, and intervening low spots often are mantled by a thin layer of ground moraine deposited by the Rainy Lobe of the Wisconsin ice sheet (Wright and Watts 1969). The till is a brown sandy material and is highly permeable. Thick soils that occur on the glacial-lacustrine clays are poorly developed, and acid soils that are poorly-to-well drained have developed on the sandy ground moraine (Nordin 1974). The soils generally are less than 25 cm thick but can range in spots to 2 m. Granite outcrops are covered by a thin mat of moss and organic matter. The soils in the drainage basins of Dogfish L. and Meander L. are similar. Those in the watershed of Lamb L., however, contain deposits of gray calcareous clays.

F. Vegetation

The lakes are located in a mixed deciduous-coniferous forest dominated by pine, spruce, fir, aspen, and birch (Wright and Watts 1969). The watersheds of the three lakes consist primarily of undisturbed virgin forest.

G. Population (The watersheds are uninhabited)

H. Land Use (None)

I. Use of Water (None)

J. Sewage and Effluent Discharge (None)

III. DESCRIPTION OF MORPHOMETRIC AND HYDROLOGIC CHARACTERISTICS

A. Surface Area of Water

The lakes have relatively small surface areas (A_o). A_o of Dogfish L. is the smallest of the three lakes. Meander L. has the greatest length and maximum width, and Lamb L. has the greatest average width and shore length (Table 1, Fig. 1).

B. Volume of Water

Bathymetric maps were used to compute lake volume for Meander L. and Lamb L. (Fig. 1). The volume of Dogfish L. was computed from an estimate of mean depth (obtained from line soundings) and surface area. Meander L. has the largest and Dogfish L. the smallest volume (Table 1).

C. Maximum and Average Depth

The maximum and average depths of the lakes are similar. Meander L. has the greatest maximum depth, and Lamb L. and Dogfish L. are essentially identical (Table 1, Fig. 1).

D. Location of Exceptional Depths

Bathymetric maps for Meander L. and Lamb L. (Fig. 1) show that the basins are steeply sloped near the shore and have gently sloping to flat bottoms

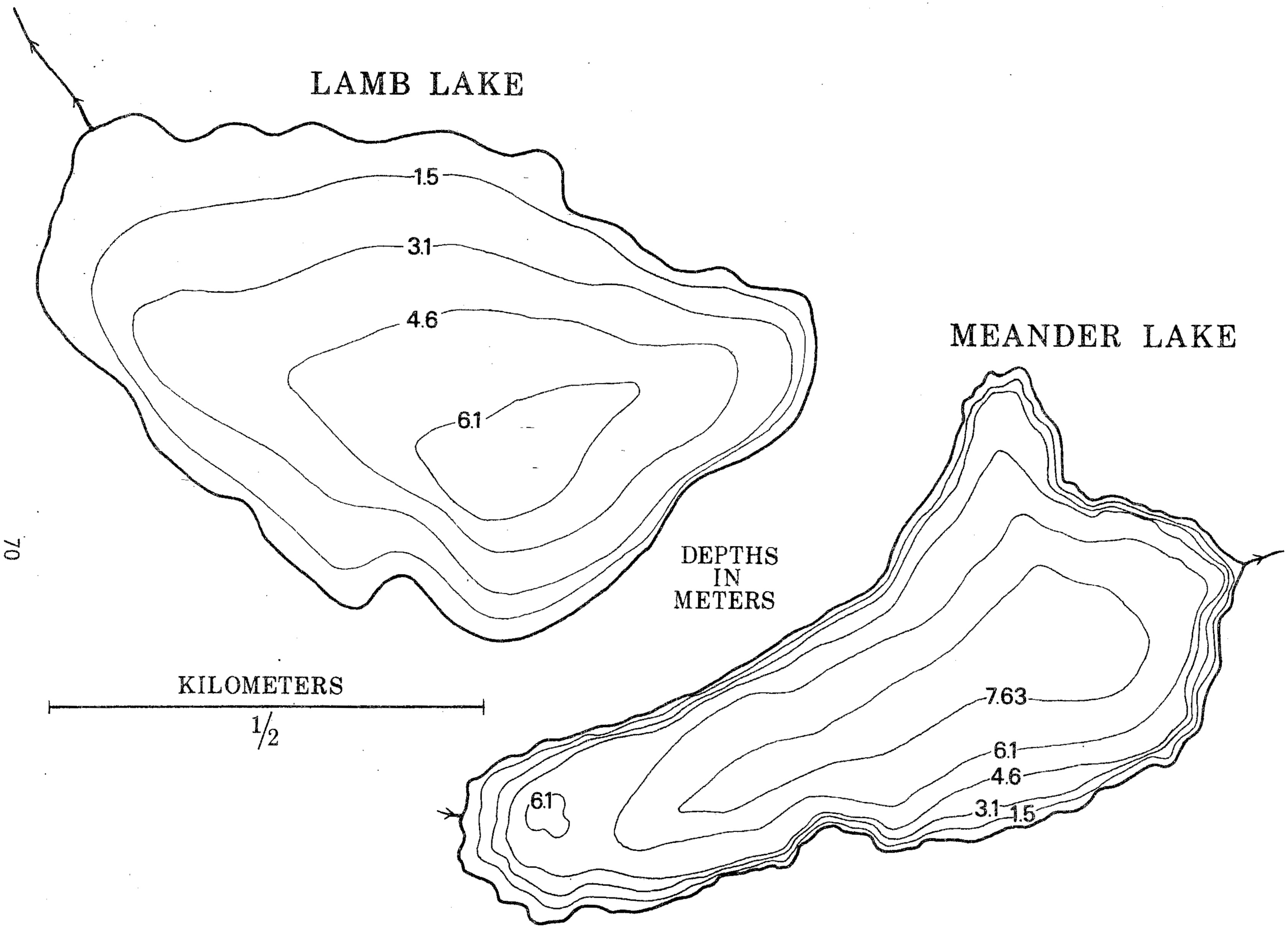

Fig. 1. Bathymetric maps of Lamb Lake and Meander Lake, Minnesota.

offshore. Line soundings in Dogfish L. indicate a similar basin configuration.

E. Ratio of Epilimnion to Hypolimnion

The ratio volume of the epilimnion/volume of the hypolimnion in Meander L. and in Lamb L. was computed from bathymetric maps and from the location of the thermocline in mid-July (3.5 m and 2.5 m in Meander L. and Lamb L., respectively). Ratios of 1.55 and 1.68 (Table 1) for Meander L. and Lamb L. indicate large euphotic zones relative to volume of bottom waters.

F. Duration of Stratification

Temperature profiles indicate that permanent summer stratification is established by mid-June and is disrupted by mid-September.

G. Nature of Lake Sediment

The nature of the lake sediments and the results of paleolimnologic analyses are given in Wright (1974) and Bradbury et al. (1974). A 60-cm core of surface sediment from Meander L. was analyzed for per cent dry weight; carbon, nitrogen, hydrogen content; total phosphorus and major cations; and pollen and diatom distribution. The major sediment constituents are biopel, clastics, and biogenic opal. Historical variations in sediment chemistry, pollen, and diatom composition have been detected. They are attributed to increases in air-born dust resulting from agricultural activities in northwestern and western Minnesota in the late 1880's and early 1900's and to recent disturbances in the watersheds themselves (see I.A.).

H. Seasonal Variation of Monthly Precipitation

Average monthly precipitation is given in Table 2. Additional data are available in Wright (1974).

I. Inflow and Outflow of Water

A hydrologic budget, including direct measurements on surface runoff, stream flow, precipitation falling in the basins, and lake outflows, is given by Wright (1974).

J. Water Currents (Not investigated)

K. Water Renewal Time

Retention time for each lake was computed from estimates of lake volume and components of the hydrologic budgets. Dogfish L. has the longest and Lamb L. has the shortest retention time (Table 1).

IV. LIMNOLOGICAL CHARACTERIZATION SUMMARY

A. Physical

1. Temperature

The lakes are dimictic, exhibiting thermal stratification in summer and winter. Spring overturn occurs just after ice-out in mid- or late-April, and the lakes stratify in June. Temperatures of the surface waters in summer range between 20° and 23°C; the bottom waters are at least 10°C in each lake during summer. Fall overturn occurs in September, and the lakes stratify inversely after an ice cover develops in November.

2. Conductivity

Specific conductance measurements show that Meander L. and Dogfish are similar, but that the waters of Lamb L. have much higher conductivity levels (Tables 3-5).

3. Light

Secchi-disc transparency ranges between 1.5 and 4.0 m in the lakes, but generally tends to be lower in Lamb L. (Tables 6-8).

4. Color

Color was not measured directly. Lamb L. is distinctly yellow-brown (difficult to filter 150 ml through a 0.5 millipore filter). The other two lakes are clear to very slightly stained with "humics."

5. Solar Radiation

Not investigated. Measurements are available from the Environmental Protection Agency Laboratory, Shagawa Lake, Ely, Minnesota.

B. Chemical

1. pH

Values of pH in the lakes are comparable (Tables 3-5).

2. Dissolved Oxygen

Oxygen measurements indicate that the surface waters of the three lakes are saturated during most seasons but that depletion to 1.0-2.0 mg/l occurs in the hypolimnion during late summer.

Table 3. A summary of selected measurements made in Meander Lake during the open water seasons of 1971 and 1972. Specific conductance (μmhos/cm @ 25°C), pH, dissolved silica (SiO_2), total organic nitrogen (TON), ammonia nitrogen (NH_4-N), nitrate-nitrogen (NO_3-N), potassium (K^+), sodium (Na^+), sulfate ($SO_4^=$), chloride (Cl^-). ND represents undetectable concentrations.

	1971 5/71 through 10/71				1972 5/72 through 10/72			
	No.	Range	$\bar{x}$	σ	No.	Range	$\bar{x}$	σ
μmhos/cm^2 @ 25°C	7	15.0–28.0	19.6	5.88	2	15.6–17.7	16.7	1.49
	1		25.8					
pH	7	6.0–7.6	6.3	0.58	3	5.6–6.0	5.77	0.21
	6	6.0–6.3	6.1	0.15	1		6.00	
SiO_2 (mg/1)	6	0.84–1.1	1.16	0.38	3	1.97–2.5	2.15	0.30
	9	0.86–1.28	1.45	0.67	3	2.08–2.53	2.25	0.24
TON (mg/1)*	6	0.0–0.9	0.48	0.29				
NH_4-N (mg/1)*	6	0.3–0.5	0.37	0.082				
NO_3-N (mg/1)*	6	0.0–0.5	0.08	0.20				
NO_2-N (mg/1)*	6	ND						
K^+ (mg/1)	9	0.3–0.65	0.45	0.15	3	0.58–0.60	0.59	0.01
Na^+ (mg/1)	9	0.5–1.1	0.65	0.20	4	0.61–0.87	0.71	0.13
$SO_4^=$ (mg/1)	7	1.6–2.0	1.86	0.18				
Cl^- (mg/1)	7	0.3–3.2	1.24	1.03				

*Measurements made on water samples filtered through a 0.45μ millipore filter.

Table 4 A summary of selected measurements made in Dogfish Lake during the open water seasons of 1971 and 1972. Specific conductance (μmhos/cm @ 25°C), pH, dissolved silica (SiO_2), total organic nitrogen (TON), ammonia nitrogen (NH_4-N), nitrate nitrogen (NO_3-N), potassium (K^+), sodium (Na^+), sulfate ($SO_4^=$), chloride (Cl^-). ND represents undetectable concentrations.

	1971 5/71 through 10/71				1972 5/72 through 10/72			
	No.	Range	$\bar{x}$	σ	No.	Range	$\bar{x}$	σ
μmhos/cm @ 25°C	7	14.0–23.0	17.3	4.19	3	14.6–17.7	16.0	1.56
pH	7	6.0–7.1	6.25	0.41	3	5.8–6.5	6.10	0.33
	7	6.0–6.7	6.19	0.26	1	–	6.0	
SiO_2 (mg/l)	7	0 –0.84	0.32	0.34	3	0.05–0.85	0.44	0.40
	6	0 –0.62	0.30	0.30	3	0.03–0.86	0.43	0.42
TON (mg/l)*	6	0.0–0.6	0.33	0.27				
NH_4-N (mg/l)*	6	0.0–0.4	0.22	0.13				
NO_3-N (mg/l)*	6	0.0–0.5	0.17	0.26				
NO_2-N) (mg/l)*	6	ND						
K^+ (mg/l)	9	0.3–0.7	0.42	0.14	4	0.38–0.57	0.45	.08
Na^+ (mg/l)	9	0.6–0.8	0.67	0.08	5	0.59–0.70	0.62	.05
$SO_4^=$ (mg/l)	7	0.9–1.4	1.17	0.21				
Cl^- (mg/l)	7	0.2–2.7	0.80	0.87				

*Measurements made on water samples filtered through a 0.45μ millipore filter.

Table 5. A summary of selected measurements made in Lamb Lake during the open water seasons of 1971 and 1972. Specific conductance (µmhos/cm @ 25°C), pH, dissolved silica (SiO_2), total organic nitrogen (TON), ammonia nitrogen (NH_4-N), nitrate nitrogen (NO_3-N), potassium (K^+), sodium (Na^+), sulfate ($SO_4^=$), chloride (Cl^-). ND represents undetectable concentrations.

	1971 5/71 through 10/71				1972 5/72 through 10/72			
	No.	Range	$\bar{x}$	σ	No.	Range	$\bar{x}$	σ
µmhos/cm @ 25°C	7	37.0-63.0	46.6	12.3	4	43.3-50.1	47.3	3.38
pH	6	6.0-6.3	6.12	0.13	6	6.0-6.9	6.34	0.30
	6	6.0-6.3	6.15	0.15	2	6.05-6.1	6.08	0.04
SiO_2 (mg/1)	6	0.35-1.08	0.75	0.26	3	1.2-2.85	1.88	0.86
	10	0.2-2.70	0.99	0.72	2	1.23-2.83	2.03	1.13
TON (mg/1)*	7	0.0-1.0	0.60	0.42				
NH_4-N (mg/1)*	7	0.0-0.5	0.34	0.19				
NO_3-N (mg/1)*	7	0.0-0.6	0.17	0.29				
NO_2-N (mg/1)*	7	0.0-0.0	ND					
K^+ (mg/1)	10	0.8-1.0	0.86	0.67	6	0.68-1.6	0.98	0.32
Na^+ (mg/1)	10	1.3-1.6	1.46	0.081	7	1.42-1.57	1.52	0.05
$SO_4^=$ (mg/1)	8	1.3-2.0	1.79	0.22				
Cl^- (mg/1)	8	0.4-1.3	0.89	0.39				

*Measurements made on water samples filtered through a 0.45µ millipore filter.

Table 6. A summary of selected measurements made in Meander Lake during the open-water seasons of 1971 and 1972. Total alkalinity (HCO_3^-), calcium (Ca^{++}), magnesium (Mg^{++}), secchi-disc transparency (SD), total phosphorus (TP), chlorophyll *a*, (Chl. *a*), phytoplankton biomass (Phyto Biomass, $10^9\mu^3$ = 1.0 mg/l fresh weight).

	1971 5/71 through 10/71				1972 5/72 through 10/72			
	No.	Range	$\bar{x}$	σ	No.	Range	$\bar{x}$	σ
HCO_3^- (mg/l)	9	3.8-11.0	6.5	2.2	9	3.5-8.0	5.9	1.5
	7	5.1- 9.2	6.8	1.6	7	4.9-8.6	6.7	1.4
Ca^{++} (mg/l)	8	.9- 2.3	1.9	.4	10	1.8-3.0	2.5	.4
	2	.9- 2.3	1.6	1.0	6	2.2-3.0	2.8	.3
Mg^{++} (mg/l)	8	.6- 1.9	1.0	.5	10	.24-.71	.6	.2
	1	-	1.4	0.0	6	.24-.61	.4	.1
SD	11	2.0- 4.0	3.1	.7	13	1.5-3.5	3.0	.6
TP (ug/l)	8	4.0-56.0	13.3	17.6	13	4.0-14.2	8.7	2.8
	7	4.8-27.0	10.9	7.5	12	6.5-18.0	9.8	3.2
Chl. *a* (ug/l)	11	.9- 5.1	3.0	1.2	6	.4-2.9	1.4	1.0
	12	2.0-10.7	4.7	2.4	7	.7-3.8	1.8	1.2
Phyto Biomass (mg/l)	10	0.61-2.98	1.8	0.83	11	.085-6.3	2.3	1.8

Table 7. A summary of selected measurements made in Dogfish Lake during the open water seasons of 1971 and 1972. Total alkalinity (HCO_3^-), calcium (Ca^{++}), magnesium (Mg^{++}), secchi-disc transparency (SD), total phosphorus (TP), chlorophyll *a* (Chl. *a*), phytoplankton biomass (Phyto Biomass, $10^9 \mu^3$ = 1.0 mg/l fresh weight).

	1971 5/71 through 10/71				1972 5/72 through 10/72			
	No.	Range	$\bar{x}$	σ	No.	Range	$\bar{x}$	σ
HCO_3^- (mg/l)	8	5.0-8.3	6.9	1.4	9	7.6-10.5	8.9	1.3
	6	5.0-11.0	7.8	1.9	7	7.2-11.0	8.9	1.4
Ca^{++} (mg/l)	8	1.2-2.6	2.0	.4	6	1.8-3.4	2.4	.7
	2	2.1-2.5	2.3	.2	1	-	3.2	0.0
Mg^{++} (mg/l)	8	0.7-0.76	.7	.02	5	.64-.73	.7	.04
	1	-	.8	0.0	-	-	-	-
SD (m)	11	1.5-4.0	2.7	.9	13	1.5-3.5	2.5	.6
TP (μg/l)	10	5.0-15.3	9.1	3.5	13	6.0-13.6	9.4	1.9
	7	7.3-13.8	10.5	2.4	12	8.0-11.7	10.1	1.3
Chl. *a* (μg/l)	9	1.2-8.4	4.2	2.5	7	1.4-2.7	2.5	.9
	9	3.7-12.0	6.2	2.5	6	2.3-7.2	4.5	2.0
Phyto Biomass (mg/l)	10	0.37-9.8	3.3	2.6	12	0.59-4.1	2.2	1.1

Table 8. A summary of selected measurements made in Lamb Lake during the open water seasons of 1971 and 1972. Total alkalinity (HCO_3^-), calcium (Ca^{++}), magnesium (Ma^{++}), secchi-disc transparency (SD), total phosphorus (TP), chlorophyll _a_, (Chl. _a_), phytoplankton biomass (Phyto Biomass, $10^9\mu^3$ = 1.0 mg/l fresh weight).

	1971 5/71 through 10/71				1972 5/72 through 10/72			
	No.	Range	$\bar{x}$	σ	No.	Range	$\bar{x}$	σ
HCO_3^- (mg/l)	8	22.2-29.0	25.7	2.4	11	24.2-45.3	30.6	6.0
	5	23.2-29.2	27.2	2.5	6	25.7-46.8	32.2	8.3
Ca^{++} (mg/l)	10	5.5-8.8	7.5	.8	9	7.5-10.5	8.9	1.1
	5	7.4-9.2	8.0	.7	4	9.6-10.4	10.0	.4
Mg^{++} (mg/l)	9	.8-2.3	2.0	.5	9	1.2-2.7	1.9	.5
	4	2.1-2.7	2.4	.3	4	.9-2.0	1.2	.3
SD (m)	12	1.5-2.0	1.8	.3	13	2.0-4.0	2.2	.6
TP (μg/l)	10	7.0-25.0	14.2	6.7	12	8.0-15.0	11.6	2.2
	7	7.3-19.9	12.3	4.5	10	8.3-17.0	12.1	2.6
Chl. _a_ (μg/l)	10	1.5-11.9	5.1	3.2	7	1.2-4.4	2.6	1.3
	11	1.8-11.2	5.6	2.5	4	1.4-4.6	2.9	1.3
Phyto Biomass (mg/l)	13	1.4-85.0	12.4	22.8	10	1.4-11.5	5.2	3.6

3. Total Phosphorus and Fractions

Total PO_4-P, total soluble PO_4-P, and particulate PO_4-P (by difference) were measured by the phosphomolybdate/stannous chloride method after digestion with $K_2S_2O_8$. Average values of total PO_4-P for the euphotic zone and the water column are given in Tables 9-11 and summarized in Tables 6-8. Total PO_4-P values generally are higher in Lamb L. than in the other two lakes.

4. Total Nitrogen and Fractions

Measurements of total nitrogen, NH_4N, NO_2-N and NO_3-N are given in Tables 3-5. The concentrations are higher in Lamb L. and Meander L.

5. Alkalinity

Bicarbonate alkalinity values are summarized in Tables 6-8. Levels in Meander L. and Dogfish L. are similar, but concentrations in Lamb L. are significantly higher.

6. Ca, Mg, Na, K, SO_4, Cl

Concentrations of major cations generally are similar in Meander L. and Dogfish L., but they are significantly higher in Lamb L. SO_4 concentrations are lower in Dogfish L., and Cl concentrations appear to be higher in Meander L.

7. Trace Metals (Not investigated)

C. Biological

Table 9. Average concentrations of chlorophyll a and total phosphorus in Meander Lake during open water seasons of 1971 and 1972. Mean values are given for the epilimnion ($\bar{X}e$) and for the water column ($\bar{X}c$). Values for $\bar{X}c$ were not calculated when measurements for either the epilimnion or hypolimnion were not available.

	Chlorophyll a (ug/1)		Total Phosphorus (ug/1)	
1971	$\bar{X}e$	$\bar{X}c$	$\bar{X}e$	$\bar{X}c$
5-21	4.1	4.9	--	--
5-28	3.2	3.6	--	--
6-20	0.9*	2.6*	4.0*	27.0*
7-5	2.8*	4.7	56.0*	--
7-21	--	10.7	--	--
8-5	1.5	2.7	5.8	4.8
8-15	4.2	5.6	5.0*	7.0*
9-3	2.8	7.5	5.7	10.6
9-18	5.1	5.6	6.0*	6.5
10-2	2.0	2.0	15.3	11.9
10-16	3.1*	3.5	8.5*	8.4
10-30	2.8*	3.3	--	--
1972				
5-7	1.8*	1.5	6.0*	7.3
5-20	0.4	0.7	9.8	9.3
6-3	--	--	6.6	6.5*
6-17	2.9	2.3	10.5	18.0
7-1	1.9	2.5	8.7	9.2
7-15	0.6	0.7	14.2	13.1
7-31	--	0.8*	5.5	7.9
8-13	0.4	3.8	10.5	12.4
8-20	--	--	4.0*	--
9-5	--	--	7.3	7.7*
9-20	--	--	8.5	8.3
10-7	--	--	10.0*	9.5*
10-21	--	--	11.1	8.8

*Indicates that only one measurement was made in the epilimnion or hypolimnion.

Table 10. Average concentrations of chlorophyll _a_ and total phosphorus in Dogfish Lake during open water seasons of 1971 and 1972. Mean values are given for the epilimnion ($\bar{X}e$) and for the water column ($\bar{X}c$). Values for $\bar{X}c$ were not calculated when measurements for either the epilimnion or hypolimnion were not available.

	Chlorophyll _a_ (ug/l)		Total Phosphorus (ug/l)	
1971	$\bar{X}e$	$\bar{X}c$	$\bar{X}e$	$\bar{X}c$
5-27	--	--	10.0*	--
6-20	3.1*	3.7*	5.0*	--
7-5	2.1	4.0	12.0*	13.8*
7-21	1.2*	12.0*	5.6	8.2*
8-10	2.4	7.4*	5.8	--
8-16	2.9	5.2	6.0	7.3*
9-4	--	--	11.0	10.2
9-19	8.4	7.1	15.3	12.4
10-2	6.7	6.3*	11.9	12.1*
10-16	5.8*	5.3	8.0*	9.4
10-30	5.5	1.7	--	--
1972				
5-5	2.7	2.7	9.5	8.0
5-20	2.1	2.3	9.8	10.6
6-3	--	--	7.3	8.9
6-17	2.3	7.2*	10.3	11.3*
7-1	2.1	4.1	8.0	11.4
7-15	1.4	3.8	8.0	8.0
7-31	2.7	--	11.8	9.9
8-12	4.4	6.7	9.5	10.0
8-19	--	--	6.0*	--
9-5	--	--	9.5	10.3
9-20	--	--	9.8	11.7
10-7	--	--	13.6*	11.3
10-22	--	--	9.6	9.9

*Indicates that only one measurement was made in the epilimnion or hypolimnion.

Table 11. Average concentrations of chlorophyll a and total phosphorus in Lamb Lake during open water seasons of 1971 and 1972. Mean values are given for the epilimnion ($\bar{X}e$) and for the water column ($\bar{X}c$). Values for $\bar{X}c$ were not calculated when measurements for either the epilimnion or hypolimnion were not available.

	Chlorophyll a (ug/l)		Total Phosphorus (ug/l)	
1971	$\bar{X}e$	$\bar{X}c$	$\bar{X}e$	$\bar{X}c$
5-20	--	--	10.0*	--
5-28	3.5	5.1*	--	--
6-5	--	--	25.0*	--
6-20	1.5*	4.8*	25.0*	--
7-5	3.0*	3.9	17.5*	--
7-21	--	1.8	--	7.5
8-4	2.3	3.4	8.0	10.4
8-15	2.4	5.2	8.5	20.0
9-5	6.9	7.4	11.8	11.8
9-18	6.3	4.6	12.0	13.0
10-2	6.7	6.9	17.5	15.9*
10-17	7.0	6.7	7.0	7.3*
10-31	11.9	11.2*	--	--
1972				
5-5	2.0	1.5	11.5	10.8
5-22	4.4*	--	8.0*	--
6-3	--	--	8.5	17.0
6-18	4.2	4.6	10.5	8.3
7-2	1.2	2.3	13.8	11.9
7-15	1.4	--	12.0	14.1*
7-31	2.3	--	14.0	--
8-12	3.0	3.1	--	13.5
8-20	--	--	12.0*	--
9-6	--	--	13.2	12.4
9-21	--	--	15.0*	12.8
10-8	--	--	9.3	8.3*
10-21	--	--	11.5	12.3

*Indicates that only one measurement was made in the epilimnion or hypolimnion.

1. Phytoplankton

a. Chlorophyll a

Chlorophyll a was measured spectrophotometrically (Strickland and Parsons 1968) until May 1972 and with a fluorometer thereafter. The latter values, corrected for phaeophytin, are suspect because of time delays in analysis. Average values of chlorophyll a for the euphotic zone and the water column are given in Tables 9-11 and summarized in Tables 6-8. Although there are marked seasonal differences among the lakes, the mean values for each of the lakes are roughly comparable.

b. Primary Production (Not investigated)

c. Algal Assays (Not investigated)

d. Identification and Count

Phytoplankton samples were analyzed by the Utermöhl (1958) technique. Approximately 400 taxa were identified to species. The counts, expressed as biomass estimates, indicate that standing crop levels in Dogfish L. and Meander L. are comparable but they are significantly higher in Lamb L. Further information on seasonal cycles is available in Bradbury et al. (1974) and Tarapchak (1975).

2. Zooplankton (Not investigated)

3. Bottom Fauna (Not investigated)

4. Fish (Not investigated)

5. Bacteria (Not investigated)

6. Bottom Flora

There is abundant diatom growth on the littoral sediments (cf. Bradbury et al. 1974).

7. Macrophytes

Not investigated extensively. Visual observations indicate sparse development in the lakes.

V. NUTRIENT BUDGETS SUMMARY

The details of nutrient budgets for major cations and total phosphorus in 1972 are available in Wright (1974, 1975) and Bradbury et al. (1974). Direct measurements were made on inputs from the atmosphere, from streams, and from overland flow. Quantities of nutrients leaving the lakes via stream outflow were measured and permit calculations of the amounts retained in the lakes. In order to determine the increase in nutrient loadings due to the fire, nutrient export from the watershed of Dogfish L. (the control lake) was considered to be representative of nutrient export from the watersheds of Lamb L. and Meander L. prior to the fire.

A. Phosphorus

The total input of phosphorus to the lakes and specific surface loadings are given in Table 12. Lamb L. has the highest specific loadings and Dogfish L. the lowest. Atmospheric loading is a major source of phosphorus for each of the lakes. Using the measured quantity of phosphorus export from the

Table 12. A summary of phosphorus loading rate computations used to predict the trophic state of Lakes Dogfish, Meander, and Lamb for 1972.

	Dogfish L.	Meander L.	Lamb L.
Total Phosphorus Loading (kgm)	4.888	9.936	12.14
Specific Surface Loading of Phosphorus ($mg/m^2/yr$)			
Runoff Input	3.1	13.2	16.6
Precipitation Input	13.7	14.4	14.0
Total	16.8	27.6	30.6
Per Cent Phosphorus Loading Retained in the Lake	74	72	71
Vollenweider's (1968) Trophic Limits ($mg/m^2/yr$)			
Admissible	57.5	66.1	57.5
Dangerous	114.8	131.8	114.8
Ratio of Actual Specific Surface Loading of Phosphorus to Vollenweider's (1968) Trophic Limits			
Admissible	0.29	0.42	0.53
Dangerous	0.15	0.21	0.27
Specific Surface Loading ($gm/m^2/yr$) divided by mean depth (m)/Retention time (yrs.), from Vollenweider 1973	0.015	0.015	0.018

watershed of Dogfish L. as the supply from a natural, undisturbed system, the fire apparently increased the loadings to Meander L. and Dogfish L. by 38 and 53 per cent, respectively. Phosphorus retention in the lakes is high and ranges between 71 and 74 per cent.

B. Nitrogen (Not investigated)

C. Other Nutrient Budgets

Major cation loadings are given in Table 13. The fire increased potassium export from the burned watersheds. Loadings of other cations, to the lakes, however, apparently were not increased by the fire.

VI. DISCUSSION

A., B. Limnological Characteristics and Delineation of Trophic State

The three lakes (with the possible exception of Lamb L.) are similar to other undisturbed wilderness lakes located on the Precambrium Shield in northeastern Minnesota (Tarapchak 1973), and generally can be considered members of the same population of lakes located in the Experimental Lakes Region (ELA) in northwestern Ontario, Canada. The lakes are low in salinity, with concentrations and ionic proportions of major anions and cations that are similar to those reported for representative ELA lakes by Armstrong and Schindler (1971). On the basis of the trophic scale for north-temperate lakes presented by Vollenweider (1968), the annual average biomass of phytoplankton would rank Dogfish L. and Meander L. in an oligotrophic-mesotrophic lake grouping (1.5-5.0 mg/l). Lamb L., however, would be considered eutrophic. Chlorophyll a and total phosphorus concentrations would place Dogfish L. and Lamb L. either in an oligotrophic or mesotrophic category; Lamb L,

Table 13. A summary of calcium, magnesium, potassium, and sodium loading rates for Lakes Dogfish, Meander, and Lamb for 1972. Symbols Runoff Input Mr, Precipitation Input Mp, Total (Mr + Mp). All values are mg/m^2 of lake surface.

	Dogfish	Meander	Lamb
Calcium			
Mr	370	1090	6168
Mp	207	223.4	222
Total	577	1313.4	6390
Magnesium			
Mr	162	450	1830
Mp	29.5	33.5	32.8
Total	191.5	483.5	1862.8
Potassium			
Mr	42.2	600	561
Mp	48	49.4	49.3
Total	90.2	649.4	610.3
Sodium			
Mr	222	556	1745
Mp	58.5	53.4	59.3
Total	280.5	609.4	1804.3

would be ranked as a eutrophic lake.

C. Trophic State vs. Nutrient Budgets

Each of the three lakes receive tolerable phosphorus loadings using Vollenweider's (1968) original relationship between the specific surface loading and mean depth $\bar{z}$ of a lake. The measured loadings for each of the three lakes are in fact well below "admissible" levels (Table 12).

A recent model proposed by Vollenweider incorporates flushing time to improve the expected relationship between phosphorus loading and lake response (cf. Vollenweider 1973 and Vollenweider and Dillon 1974). The expression specific loading divided by $\bar{z}$/detention time was computed for the lakes. These computations place each lake well below "admissible" phosphorus loadings, and suggest that the lakes are subjected to loadings that can be tolerated by their existing morphometry and hydrology.

VII. SUMMARY

An investigation on water chemistry and phytoplankton, coupled with studies on nutrient budgets and hydrology of three wilderness lakes in northeastern Minnesota, was undertaken to assess the effects of terrestrial nutrient release on wilderness lakes on the Precambrian Shield. The lakes are similar chemically and biologically to other lakes in northeastern Minnesota and appear to be members of the same population of lakes in northwestern Ontario. Phosphorus export from the watersheds of two lakes increased substantially after the fire. The loadings, however, were not high enough to drive the lakes from oligotrophy into a state of mesotrophy as judged by Vollenweider's nutrient loading/trophic state model.

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PHYTOPLANKTON, PHOSPHORUS, AND SEWAGE EFFLUENTS IN LAKE MINNETONKA

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INTRODUCTION

Lake Minnetonka occupies a group of basins in eastern Minnesota, near Minneapolis. The city of Minneapolis developed after the middle of the 19th century at St. Anthony Falls on the Mississippi River, the region became accessible by railroads after 1870, and Lake Minnetonka began to attract tourists from throughout the United States. The lake became an important vacation area, with several large hotels, elegant summer homes, and commercial steamboat service for transportation across the lake.

The villages near the lake have become residential suburbs of Minneapolis since 1950, and most residences are now occupied permanently. The population in the watershed was about 10,000 in 1930, and it increased at the rate of 4% per year to 46,000 in 1970. The villages began to construct secondary sewage treatment plants in 1927. By 1963, effluents from six municipal treatment plants were entering the lake and its tributaries. The quality of water in the lake decreased as the population increased. The minimum secchi-disc transparency of Lower Lake Minnetonka, the largest basin, decreased from 2.5 m in 1937 to only 0.9 m in 1969, and maximum population densities of planktonic algae increased about four-fold.

Sewage effluents have now been diverted from the lake in an effort to decrease algal abundance. Effluents were diverted from Lower Lake Minnetonka, the largest basin, during the summer of 1971-1972. Concentrations of phosphorus and population densities of planktonic algae in this basin have both decreased since diversion. Most of the phosphorus that was lost must have been deposited in the sediments, because losses through the outlet stream are trivial.

Lake Minnetonka is a complex of lake basins with a total surface area of 58.6 km^2 (22 $mi.^2$) located in east-central Minnesota (44° 55' N lat, 93° 37' W long) (Fig. 1). The area of the watershed is 312 km^2 (123 $mi.^2$), which includes marshes and other lakes with an area of 48 km^2 (18.9 $mi.^2$), and uplands with an area of 212 km^2 (82.6 mi^2). Thus the total catchment area is 371 km^2. The water level is controlled by a small dam at the outlet (Minnehaha Creek), which has a crest of 283 m (929 ft.) above sea level.

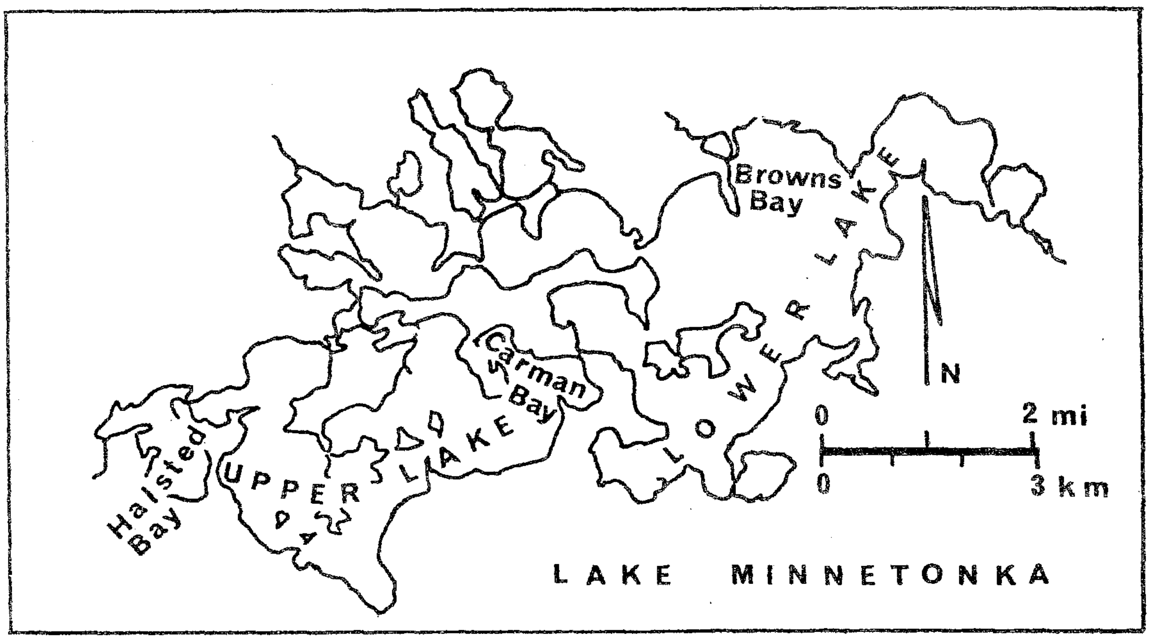

Figure 1. Map of Lake Minnetonka. The localities studied most intensively are Browns Bay in the Lower Lake, Carman Bay in the Upper Lake, and Halsted Bay.

Climatic, hydrologic, and geographic data about the lake and its watershed have been compiled in engineering reports.[1, 2, 3] The mean annual temperature between 1891 and 1966 was 7.8° C (45° F). The minimum temperature in winter was -38° C (-37° F), and the maximum in summer was 44° C (112° F). Average air temperatures are below 0° C (32° F) for five months and below -7° C (20° F) for three months each year. Ice covers the lake approximately five months each year, from December to April.

The average precipitation during the period from 1915 to 1968 was 73 cm/yr (28.9 in./yr). Average evaporation from a free water surface is 76 cm/yr (30.0 in./yr). Precipitation is highest, 10 cm (4.1 in.) during June and lowest, 2 cm (0.76 in.), during January. Mean annual snowfall is 144 cm (55.8 in.). The driest year was 1958, with 41 cm (16.2 in.) of precipitation, and the wettest was 1965, with 101 cm (40 in.).

The lake occupies basins that were occupied by blocks of glacial ice buried in the St. Croix Moraine.[4] The moraine is composed of a young, gray, calcareous drift and an older, red, non-calcareous drift. The red drift was deposited on Lower Paleozoic rocks by a glacial lobe that came from the basin of Lake Superior, 240 km (150 mi.) northeast of Lake Minnetonka. The gray drift was deposited by a glacial lobe that came from the west, moved across the red drift into western Wisconsin, and began to melt about 14,000 years ago.[5] The oldest lake sediments in a lake located in the gray drift 15 km southwest of Lake Minnetonka, contain wood that is 12,000 ± 160 radiocarbon years old,[6] indicating that melting ice blocks may have remained buried in the moraine 2,000 years after the glacier disintegrated.[7]

The glacial drift includes 40-100 m of alternating gravels, sands, and clays. One of the Paleozoic formations beneath the drift, the Jordan Sandstone, is an artesian aquifer. It receives water from Lake Minnetonka, and it is an important source of ground water for metropolitan Minneapolis.

The landscape near Lake Minnetonka is hilly, with maximum elevations of 330 m (880 ft.), or 47 m above the lake. A mixed deciduous fotest composed of <u>Ulmus americana</u> (American elm), <u>Tilia americana</u> (basswood), <u>Acer saccharum</u> (sugar maple), and <u>Quercus ellipsoidalis</u> (pin oak) occurred in the region until the time of european settlement, which began during the middle of the 19th century.

The woodlands were replaced initially by small dairy farms and market farms, but agriculture is now relatively unimportant; only 42 km^2 (16 $mi.^2$) or 15% of the watershed is now agricultural. The area of urban land in the watershed in 1970 was about 52 km^2, which is about 30% of the land that would be suitable for residential or commercial use. The average population density on urban land in the watershed was about 700 persons/km^2 (1,800/$mi.^2$) in 1970.[3] The lake's shoreline, about 175 km (110 mi.) long, is now almost entirely occupied by permanent housing. The lake has been a major incentive for urbanization. The average value of land with lakeshore frontage is approximately ten times greater than the value of land one mile (1.6 km) from the lake. There was 23 rental and mooring facilities for pleasure boats on the lake in 1969, serving about 2,500 boats. About 27,000 kg (60,000 lb.) of rough fish and about 91,000 kg (200,000 lb.) of game fish are caught each year by fishermen, a yield of 20 kg/ha. About 1,500 temporary shelters are erected on the ice by fishermen during some winters.

Effluents from six municipal sewage treatment plants, which served about 20,000 persons or almost one-half the population in the watershed, contributed 21% of the water, 32% of the total nitrogen, and 81% of the total phosphorus that entered the lake from tributaries in 1966-1967. All villages in the watershed will close their sewage treatment installations during this decade and join a sanitary district that operates a large installation on the Minnesota River, 15 km south of the lake. Two villages that discharged effluents into Lower Lake Minnetonka joined the sanitary district in 1972, thereby reducing the annual phosphorus influx to the largest basin almost 80%.

MORPHOMETRY AND HYDROLOGY

Areas, volumes, and depths of the basins of Lake Minnetonka are compiled in Table 1. The basins are connected with each other by natural and artificial navigation channels. The Lower Lake contains 54% of the water (216×10^6 m^3). The maximum depth of the Lower Lake is 27.8 m, and the mean depth is 8.3 m. The Upper Lake contains 29% of the water (115×10^6 m^3); its maximum depth (25.6 m) and mean depth (6.7 m) are both less than those of the Lower Lake. The total volume of the other basins is only 17% of the total lake volume, but their combined area is 26% of the total. The greatest depth, 31 m, is in Crystal Bay, but the maximum depth of Crystal Bay indicated on the map used for these computations is only 24 m.

The largest tributary streams, Sixmile Creek and Painter Creek, both drain agricultural regions and flow through marshes and lakes before

Table 1. MORPHOMETRY OF LAKE MINNETONKA

Data derived from a map prepared by the State of Minnesota Department of Natural Resources.

Basin	Area		Volume		Depth, m	
	10^6 m^2	% of total	10^6 m^3	% of total	max	mean
Lower Lake	26.19	45.0	216.20	53.9	27.8	8.3
Upper Lake	17.32	29.5	114.83	28.6	25.6	6.7
Crystal Bay	3.36	5.7	28.55	7.1	23.8	8.5
Grays Bay	0.76	1.3	2.10	0.5	6.1	2.8
Maxwell Bay	1.20	2.0	5.20	1.3	9.5	4.3
Stubbs Bay	0.80	1.4	3.30	0.8	11.6	4.1
North Arm	1.32	2.3	5.81	1.5	14.0	4.3
West Arm	2.32	4.0	8.98	2.2	9.8	3.9
Jennings Bay	1.20	2.0	3.05	0.8	6.7	2.5
Forest Lake	0.34	0.6	1.45	0.4	12.5	4.2
Harrison Bay	1.02	1.7	2.43	0.6	9.5	2.4
Halsted Bay	2.20	3.8	8.63	2.1	10.1	3.9
Black Lake	0.30	0.5	0.85	0.2	7.6	2.8
Seton Lake	0.16	0.3	0.33	0.1	7.0	2.1
Total	58.58		401.73			

they enter western basins of Lake Minnetonka. The outlet stream is Minnehaha Creek, which flows eastward through south Minneapolis until it enters the Mississippi River, 30 km from Lake Minnetonka.

The hydrologic balance for Lake Minnetonka between 1914 and 1968 was computed in terms of annual additions and losses of water to a unit of lake surface, as follows:[3]

	cm/yr	
Additions		
direct precipitation	73	
tributaries and overland flow	33	
Total Additions		106
Losses		
evaporation	76	
leakage to aquifer	10	
outflowing stream	20	
Total Losses		106

If the mean depth of the lake is 690 cm, then 15.4% (106 cm yr^{-1} ÷ 690 cm x 100) of the water in the lake enters each year. Eleven percent of the lake's volume is lost by evaporation each year, 1.4% is lost to aquifers, and 2.9% is lost to the outlet. If the retention time for water is defined as the ratio of the mean depth to the total annual influx (or losses) to a unit of area, then the detention time is 6.3 yr (690 ÷ 106 cm yr^{-1}).

Water normally flows over the crest of the dam at the outlet during the spring and early summer, but there was no overflow during 12 years of drought between 1930 and 1942. The surface fell to 1.8 m below the crest of the dam by 1937, exposing 18% of the lake bottom. In some areas the shoreline receded 300 m.

LIMNOLOGY

TEMPERATURE

Temperatures and the duration of thermal stratification depend upon the areas and depths of the basins. The variability is indicated by the difference between the Lower Lake, a large, deep basin, and Halsted Bay, a small, shallow basin, which have been studied most intensively (Fig. 2).

The ice usually melts in early or mid-April, and temperatures increase until they reach 8-10° C at all depths in mid-May, when the water becomes thermally stratified. Maximum temperatures of 24-26° C are achieved in July and August. The shallow basins become isothermal again in September, and the large basins in late October.

LIGHT

Coefficients for the attenuation of photosynthetically active radiation (PhAR) by phytoplankton and by the water have been computed from the photosynthetic rates of phytoplankton incubated *in situ* in the Lower Lake and in Halsted Bay.[8] If phytoplankton are distributed uniformly in the mixed layer, then the daily integral photosynthetic rate is given by the equation[8, 9, 10]

$$\pi = \ln\,(I_o/I_{z'})\; p_{max}/(\varepsilon_c c + \varepsilon_w), \qquad (1)$$

where π = daily integral photosynthetic rate, g C m^{-2} day^{-1}

I_o = average irradiance at the surface

$I_{z'}$ = average irradiance at a depth z'

ε_w = coefficient for attenuation of PhAR by water and substances other than phytoplankton in the water, m^{-1}

ε_c = coefficient for attenuation of PhAR by phytoplankton, referred to chlorophyll *a*, m^2 mg Chl^{-1}

c = concentration of chlorophyll *a*, m^2 mg Chl^{-1}

p_{max} = volumetric photosynthetic rate at the average depth where PhAR is saturating, g C m^{-3} day^{-1}

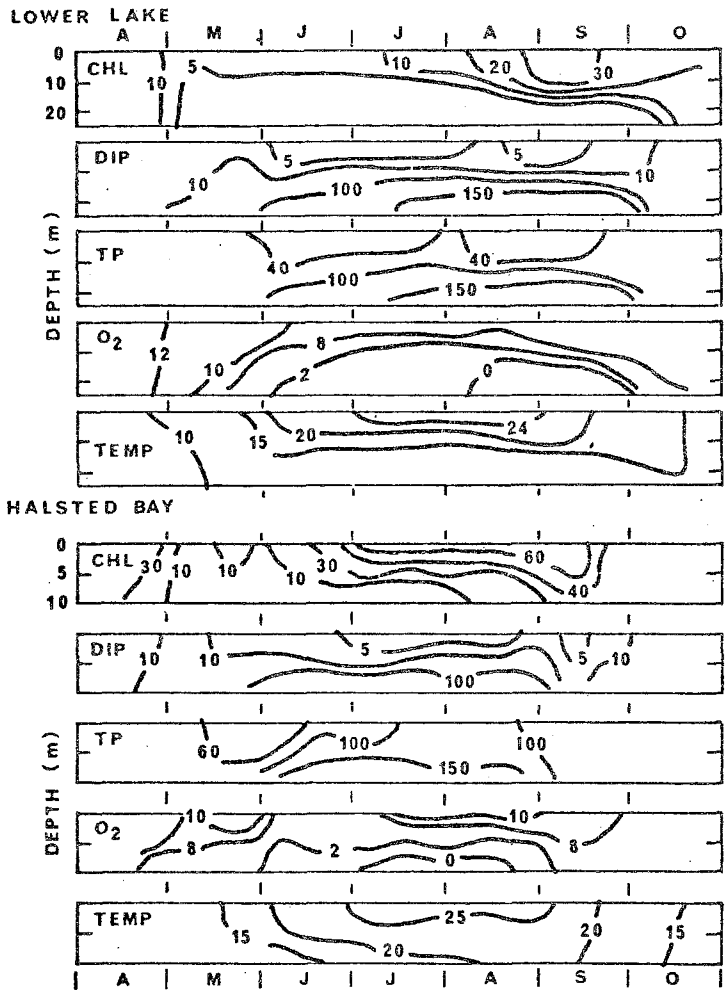

Figure 2. Distribution of chlorophyll a (mg Chl m^{-3}), dissolved inorganic phosphorus (mg DIP m^{-3}), total phosphorus (mg TP m^{-3}), dissolved oxygen (g O_2 m^{-3}) and temperatures (° C) at Browns Bay in Lower Lake Minnetonka (upper panels) and at Halsted Bay (lower panels) during 1973.

The total coefficient for the attenuation of PhAR $\varepsilon = \varepsilon_w + \varepsilon_c c$. Equation (1) may be rearranged as an equation for a straight line of the form y = bx + a:

$$\ln (I_o/I_{z'}) \; p_{max}/\pi = \varepsilon_c c + \varepsilon_w. \qquad (2)$$

The coefficients ε_c and ε_w were evaluated by (1) measuring p_{max}, π, and c periodically, (2) computing $\varepsilon = \ln (I_o/I_{z'}) \; p_{max}/\pi$ for each measurement, and (3) plotting ε against concentrations of chlorophyll. The resulting curve is a straight line with a slope ε_c that intercepts the ordinate at ε_w (Fig. 3). The coefficients were evaluated from the equation for the regression of ε on c computed by least squares.[8]

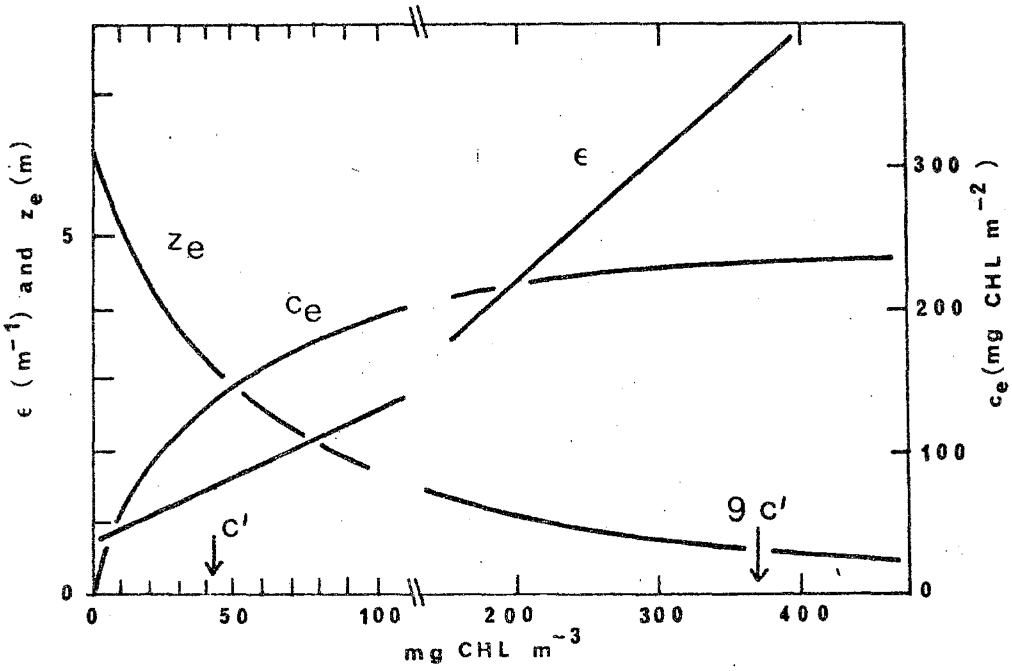

Figure 3. Relationships between the coefficient for the attenuation of photosynthetically active radiation (ε), the thickness of the euphotic zone (z_e), the quantity of chlorophyll a in the euphotic zone (c_e), and concentrations of chlorophyll a in Lower Lake Minnetonka. Phytoplankton attenuate 50% and 90% of subsurface PhAR at the concentrations c' and 9 c' (from Megard et. al. 1975).

The estimate of ε_c in the Lower Lake ($0.018 \pm 0.008\ m^2\ mg\ Chl^{-1}$) during 1968 and 1969 is similar to the estimate of ε_c in Halsted Bay ($0.018 \pm 0.009\ m^2\ mg\ Chl^{-1}$) during 1973 and 1974, in Windermere, England,[11] and in Lake George, Uganda.[12] However, the value of ε_w ($0.74 \pm 0.20\ m^{-1}$) in the Lower Lake is significantly lower ($P < 0.05$) than the value of ε_w ($1.40 \pm 0.38\ m^{-1}$) in Halsted Bay.

The thickness of the euphotic zone and the quantity of chlorophyll in the euphotic zone may be computed as functions of concentrations of chlorophyll from the values of ε_w and ε_c. If the base of the euphotic zone is the depth z_e at which irradiance I_e is reduced to 1% of the irradiance I_o at the surface, then

$$I_e = I_o \exp - z_e (\varepsilon_w + \varepsilon_c c). \tag{3}$$

Therefore,

$$z_e = -\ln (I_e/I_o)/(\varepsilon_w + \varepsilon_c c) \tag{4}$$

$$= 4.6/(\varepsilon_w + \varepsilon_c c)$$

If chlorophyll is dispersed uniformly, then the quantity in the euphotic zone $c_e = c\ z_e$, so that

$$c_e = 4.6\ c/(\varepsilon_w + \varepsilon_c c) \tag{5}$$

Concentrations of chlorophyll a in Lower Lake Minnetonka ranged from 3 to 55 mg m^{-3} during 1968 and 1969. The coefficient for the attenuation of PhAR therefore ranged from 0.8 to 1.5 m^{-1}, the thickness of the euphotic zone ranged from 6 to 2.5 m, and the quantity of chlorophyll a in the euphotic zone ranged from 10 to 130 mg Chl m^{-2} (Fig. 3).

The secchi-disc transparency of Lake Minnetonka depends upon concentrations of chlorophyll a (Fig. 4), decreasing from 3.5 m to 0.5 m as concentrations of chlorophyll increase from 5 to 80 mg m^{-3}.

The minimum transparency in the Lower Lake was 0.9 m during 1969, when the concentration of chlorophyll was 43 mg m^{-3}. However, the minimum transparency increased to 1.2 m and the maximum concentration of chlorophyll decreased to 30 mg m^{-3} in 1974, two years after the phosphorus influx decreased.

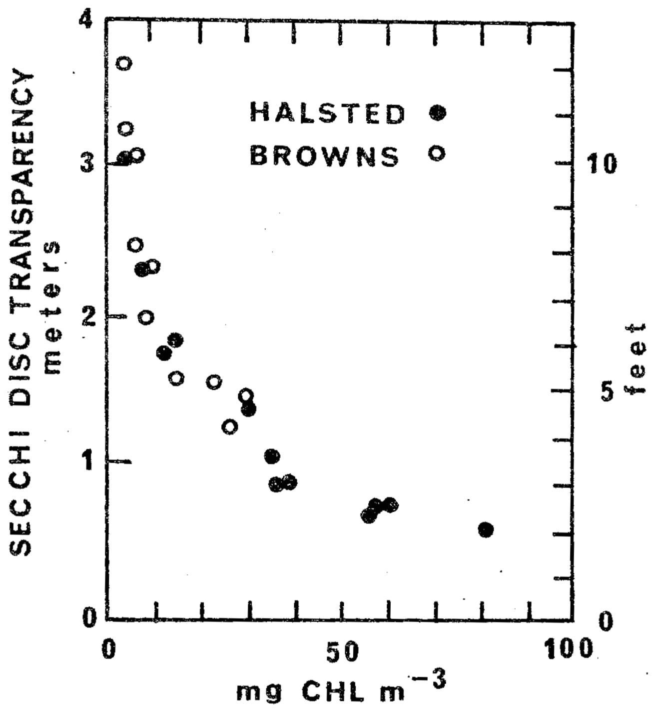

Figure 4. Relationships between secchi-disc transparency and concentrations of chlorophyll a in Lake Minnetonka during 1972 and 1973.

The water in the Lower Lake was more transparent during 1937 than it is now. The transparency ranged from 3.5 m in July to a minimum of 2.5 m in September, 1937.[13] The relationship between transparency and chlorophyll was probably the same in 1937 as now, because ε_w and ε_c probably have not changed significantly. Therefore the minimum transparency during 1937 probably corresponded to a maximum chlorophyll concentration of about 10 mg m^{-3}. Thus maximum concentrations of chlorophyll in Lower Lake Minnetonka were probably four times higher in 1969 and three times higher in 1974 than they were in 1937.

CHEMISTRY

The total salinity of Lake Minnetonka is 6.5 meq $liter^{-1}$, and the specific conductance is 317 μmho at 25° C (Table 2). Calcium (1.35 meq $liter^{-1}$) and magnesium (1.32 meq $liter^{-1}$) are the dominant cations, and bicarbonate (2.5 meq $liter^{-1}$) is the dominant anion. The chemistry of water in Lake Minnetonka is similar to the average for other lakes in the region, except that concentrations of Na and Cl are somewhat high, possibly because large quantities of salt are applied to roads near the lake to melt snow and ice during winter. Alkalinity ranges from 2.1 to 2.9 meq $liter^{-1}$, pH from 7.5 to 8.8, and dissolved inorganic carbon from 28 to 34 mg $liter^{-1}$ in the mixed layers of the Lower Lake and Halsted Bay (Table 3).

Table 2. CONCENTRATIONS OF MAJOR IONS

Data for Lake Minnetonka from State of Minnesota Department of Natural Resources and for other lakes from Bright.[21]

Ion	Minnetonka		Average for other lakes in deciduous forest in Minnesota
	mg $liter^{-1}$	meq $liter^{-1}$	meq $liter^{-1}$
Ca	27	1.35	1.48
Mg	16	1.32	1.71
Na	10	0.44	0.15
K	5.8	0.15	0.17
Σ cations	58.8	3.25	3.51
HCO_3	153	2.51	2.66
CO_3	0	0	
SO_4	9.8	0.20	0.39
Cl	20	0.56	0.09
Σ anions	182.8	3.28	3.14
Σ salinity	341.6	6.53	6.65

Spec. Conductance: 317 μmho @ 25° C

Table 3. ALKALINITY (TA), pH, AND DISSOLVED INORGANIC CARBON (DIC) IN THE MIXED LAYER OF LOWER LAKE MINNETONKA AND OF HALSTED BAY DURING 1972

	LOWER LAKE			HALSTED BAY		
Date	TA[a]	pH	DIC[b]	TA	pH	DIC
May 4	2.6	8.3	2.6			
Jun 1	2.6	8.5	2.6	2.8	8.1	2.8
8				2.8	8.6	2.8
15	2.6	8.8	2.5	2.9	8.7	2.8
Jul 5	2.4	8.2	2.4	2.5	8.4	2.5
18	2.4					
Aug 10	2.5					
27	2.1	8.2	2.1	2.2	7.4	2.3
Sep 8				2.4	7.8	2.5
23	2.4	8.1	2.4	2.5	8.2	2.5

[a]meq $liter^{-1}$
[b]mMol $liter^{-1}$

The distribution of dissolved oxygen, total phosphorus, dissolved phosphorus, and chlorophyll a during 1973 at Browns Bay in the Lower Lake and at Halsted Bay are shown in Fig. 2. Although the mean depth of the Lower Lake (8.3 m) is twice that of Halsted Bay (3.9 m), the basins are similar in that dissolved oxygen disappears from the deep water of both during July; dissolved inorganic phosphorus is 5-10 mg m^{-3} in surface water, and may exceed 150 mg m^{-3} in deep water. However, concentrations of total phosphorus and of chlorophyll a are both higher in Halsted Bay than in the Lower Lake. The mean (± 95% confidence limits) concentration of total phosphorus during 1973 was 88 ± 7 mg m^{-3} in Halsted Bay and 40 ± 4 mg m^{-3} in the Lower Lake, whereas the mean concentration of chlorophyll a was 37 ± 8 mg m^{-3} in Halsted Bay and 14 ± 4 mg m^{-3} in Browns Bay (Table 4).

Concentrations of total phosphorus and of chlorophyll a have both decreased in the Lower Lake since sewage effluents were diverted (Table 4, Fig. 5). The mean concentration of phosphorus during about 80% of the ice-free season decreased 10 ± 7 mg m^{-3}, from 50 ± 6 mg m^{-3} in 1969 to 40 ± 4 mg m^{-3} in 1973, whereas the mean concentration of chlorophyll decreased 8 ± 6 mg m^{-3}, from 22 ± 5 mg m^{-3} in 1969 to 14 ± 4 mg m^{-3} in 1973.

Table 4. CHLOROPHYLL a AND PHOSPHORUS IN LAKE MINNETONKA

Mean (± 95% confidence limits) concentrations (mg m^{-3}) in the mixed layer (0-5 m in Lower Lake, 0-3 m in Halsted Bay). Number of samples in parentheses.

Basin	Year	Chlorophyll	Phosphorus	
			Dissolved Inorganic	Total
Lower Lake	22 Apr - 7 Oct, 1969	22 ± 5 (21)		50 ± 6 (21)
	24 Apr - 18 Oct, 1973	14 ± 4 (26)	8 ± 3	40 ± 4 (28)
	23 May - 13 Sep, 1973	13 ± 3 (18)		36 ± 3 (20)
	18 May - 17 Sep, 1974	19 ± 3 (22)		35 ± 3 (20)
Halsted Bay	24 Apr - 18 Oct, 1973	37 ± 8 (26)	9 ± 4	88 ± 7 (28)

There is a high correlation between chlorophyll and total phosphorus in Lake Minnetonka during summer but not during spring and autumn,[14] and it is therefore notable that chlorophyll has decreased in the Lower Lake only during the summer. The mean (± 95% confidence limits) concentration of chlorophyll in the mixed layer between 10 July and 10 September has decreased from 37 ± 3 mg m^{-3} in 1968 and 1969 to 17 ± 4 mg m^{-3} in 1973 and 1974 at the rate of 1.0 ± 0.3 mg Chl/mg P as mean phosphorus decreased from 48 ± 7 to 34 ± 4 mg m^{-3} (Fig. 6, Table 5). The percentage of the variance of chlorophyll attributable to its regression on total phosphorus (P %) was 62% in 1968-1969

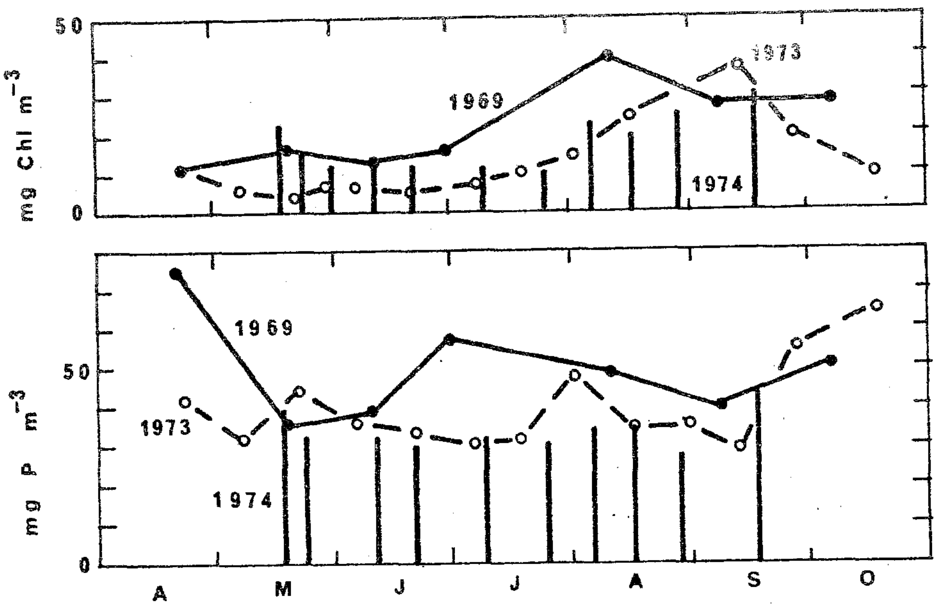

Figure 5. Mean concentrations of chlorophyll a and of total phosphorus in Lower Lake Minnetonka in 1969, before diversion of sewage effluents and in 1973 and 1974, after diversion. Each point for 1969 is the mean concentration of samples from the surface at three localities (Browns Bay, Wayzata Bay, and Gale Island). Each point for 1973 is the mean of concentrations at depths of 0 m and 5 m in the mixed layer at Browns Bay. (See also Table 4).

and 53% during all four summers. However the regression of chlorophyll on phosphorus was not significant ($P > 0.05$) during the summers of 1973 and 1974, indicating that population densities of phytoplankton in Lower Lake Minnetonka are now independent of phosphorus concentrations.

RATES OF PHOTOSYNTHESIS

Volumetric photosynthetic rates at saturating light (p_{max}) depend upon population densities (c) and specific photosynthetic rates at saturating light (P_{max}) according to the equation

$$p_{max} = c\, P_{max}. \qquad (6)$$

Values of P_{max} were computed from the slope of the equation for the linear regression of p_{max} on concentrations of chlorophyll.[8] The high linear correlation between p_{max} and chlorophyll at 6-16° C ($r = 0.84$) and at 18-25° C ($r = 0.93$) over the range of concentrations

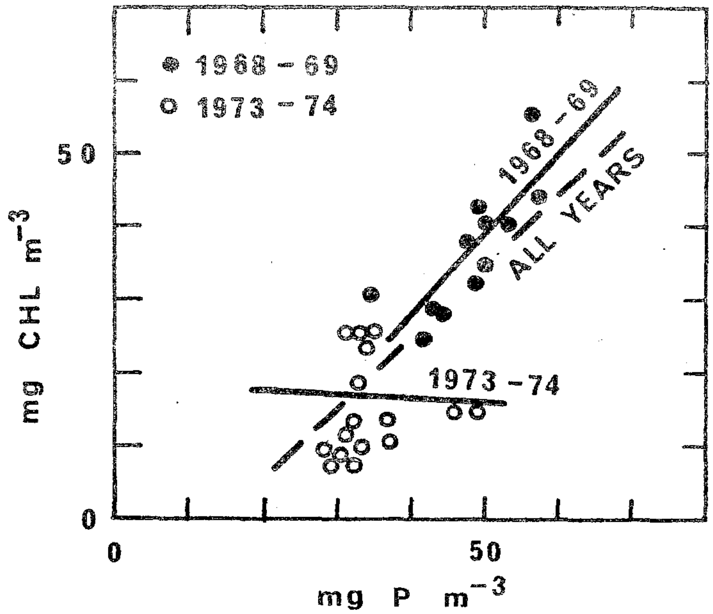

Figure 6. Relationships between concentrations of chlorophyll a and of total phosphorus in the mixed layer (0 m and 5 m) of Lower Lake Minnetonka between July 10 and September 10 of 1968 and 1969 (closed circles) and of 1973 and 1974 (open circles). Regression statistics are summarized in Table 5.

Table 5. RELATIONSHIP BETWEEN CONCENTRATIONS OF CHLOROPHYLL a AND OF TOTAL PHOSPHORUS

Mean (± 95% confidence limits) concentrations (mg m^{-3}) of total phosphorus (TP) and of chlorophyll a (Chl), and the slope (b) of the equation ($y = bx + a$) for the linear regression of chlorophyll on total phosphorus in the mixed layer of Lower Lake Minnetonka during summer (July 10 - September 10) before (1968-1969) and after (1973-1974) diversion of sewage effluents. Also indicated are the proportion (P %) of the variance of chlorophyll attributable to its regression on phosphorus, the significance of the regression (** $P < 0.01$, *** $P < 0.001$, NS not significant $P > 0.05$), the correlation coefficient (r), and the number of samples (n).

	TP	Chl	b	P %	r	n
All years	39 ± 3	25 ± 3	0.98 ± 0.3	53***	0.74	30
1968-1969	48 ± 7	37 ± 3	1.09 ± 0.6	62**	0.79	12
1973-1974	34 ± 4	17 ± 4	-0.04 ± 0.7	NS	0.04	18

up to 120 mg m^{-3} indicates that P_{max} is independent of population densities in Lake Minnetonka. The mean value (± 95% confidence limits) of P_{max} is significantly ($P < 0.05$) higher at 18-25° C (54 ± 5 mg C mg Chl^{-1} day^{-1}) than at 6-16° C (31 ± 7 mg C mg Chl^{-1} day^{-1}). These rates of carbon assimilation per unit chlorophyll at saturating light were computed from rates of oxygen evolution with a photosynthetic quotient of 1.2. The corresponding rates of oxygen evolution were 173 ± 16 mg mg Chl^{-1} day^{-1} at 18-25° C and 99 ± 22 mg mg Chl^{-1} day^{-1} at 6-16° C. Volumetric photosynthetic rates at saturating light also depend upon concentrations of total phosphorus, but only during summer when population densities depend upon phosphorus.[14]

If equation (6) is substituted in equation (1), then it can be shown[8] that

$$\pi = \pi_{max}/[1 + \varepsilon_w/(\varepsilon_c c)] \tag{7}$$

and that

$$\pi_{rel} = \pi/\pi_{max} = \frac{1}{1 + [\varepsilon_w/(\varepsilon_c c)]} \tag{8}$$

where $\pi_{max} = \ln(I_o/I_{z'})\ P_{max}/\varepsilon_c$ is the maximum daily integral photosynthetic rate attained by populations dense enough to attenuate all photosynthetically active radiation.[15] Equation (7) is an equation for a rectangular hyperbola, indicating that integral photosynthetic rates approach an upper assymptote (π_{max}) as population densities increase. If daily average ln $(I_o/I_{z'})$ is 2.7,[16] if P_{max} is 54 mg C mg Chl^{-1} day^{-1}, and if ε_c is 0.02 m^2 mg Chl^{-1}, then π_{max} is 7.3 g C m^{-2} day^{-1} at 20° C in Lower Lake Minnetonka.

The relative integral photosynthetic rate (π_{rel}) is a dimensionless parameter that ranges from 0 to 1, depending upon the proportion of PhAR attenuated by phytoplankton. Phytoplankton attenuate 50% of the light and $\pi_{rel} = 0.5$ at the chlorophyll concentration $c' = \varepsilon_w/\varepsilon_c$, which is 41 mg m^{-3} in the Lower Lake and 76 mg m^{-3} in Halsted Bay. Population densities in Halsted Bay must be almost twice as great as in the Lower Lake in order to attain $\pi_{rel} = 0.5$ because ε_w is higher in Halsted Bay.

Daily integral photosynthetic rates have been computed from changes of oxygen concentrations (PQ = 1.2) in transparent and opaque bottles filled with lake water and incubated at depth intervals for six hours, beginning at noon. The mean (± 95% confidence limits) daily integral photosynthetic rate measured monthly (22 April - 7 October) during 1969 at three localities in the Lower Lake was 2.2 ± 0.4 g C m^{-2} day (computed from Table 4 in Megard 1972[14]), and the mean concentration of chlorophyll was 22 ± 5 mg m^{-3} (Table 4), corresponding to π_{rel} = 0.35. The mean concentration decreased to 14 ± 4 mg m^{-3} during the comparable interval of 1973, and π_{rel} therefore decreased to 0.23. However, the decrease of mean concentrations of chlorophyll during summer (10 July - 10 September) indicate that the mean integral photosynthetic rate decreased from 0.48 π_{max} (1968-1969) to 0.29 π_{max} (1973-1974), or from 3.4 to 2.1 g C m^{-2} day^{-1}.

The dominant phytoplankton of the Lower Lake during 1969 were *Stephanodiscus*, *Cyclotella*, and Cryptophyta during April, May, and June. A diverse assemblage of Cyanophyta including *Aphanizomenon*, *Anabaena*, *Oscillatoria*, *Lyngbya*, and *Microcystis* was dominant during summer and early autumn, when population densities were highest.[14] In contrast the dominant phytoplankton in this basin during 1937 were *Melosira*, *Fragilaria*, and *Lyngbya*.[13]

PHOSPHORUS BUDGET

The phosphorus budget for the Lower Lake is amenable to analysis, but phosphorus enters most other basins from many diffuse sources which are difficult to measure. Exchanges between the Lower Lake and other large basins may be neglected because the exchanges of water between large basins are small and because concentrations of phosphorus in the Lower Lake are similar to the concentrations in adjacent large basins.

The estimated annual influx of total phosphorus to the Lower Lake between 1 June, 1969, and 31 May, 1970, was 12.9 t (= 12.9 x 10^3 kg), of which 69% came from sewage effluents (Table 6). The annual loading was therefore 0.5 g m^{-2} yr^{-1} (12.9 x 10^6 g ÷ 26.2 x 10^6 m^2). The annual loss of water and therefore of phosphorus through the outlet (0.01 t)

Table 6. PHOSPHORUS INFLUX TO LOWER LAKE MINNETONKA, 1 JUNE, 1969 - 31 MAY, 1970[a]

Source	Quantity	
	tonnes	%
Sewage effluents	8.9	69
Tributary streams	0.4	3
Overland runoff[b]	2.0	16
Rainfall on lake[c]	0.4	3
Septic tank drainage	1.2	9
Total	12.9	100

[a]Compiled by Harza Engineering Co.

[b]Estimated as 130 pounds/square mile from rural uplands and 510 pounds/square mile from urban land

[c]Concentrations in rainfall estimated to be 20 mg m^{-3}

was negligible. The quantity of total phosphorus in the Lower Lake fluctuated between 7.6 and 19.3 t, and the mean was 13.4 t (Fig. 7). Virtually all the phosphorus that entered must have been deposited in the sediments if the lake was in a steady state. The influx from sewage effluents and septic tanks was stopped during the winter of 1971-1972, decreasing the annual influx 78%, to 2.8 t yr^{-1} (0.1 g m^{-2} yr^{-1}).

The response to the decreased influx was computed with the following equation for phosphorus in a perfectly-mixed basin (A. G. Fredrickson pers. comm., Vollenweider 1968[17]):

$$V \frac{dC}{dt} = i - (Q_o C + SAC) \tag{9}$$

where V = volume of basin

C = concentration of total phosphorus

i = annual influx from all sources

Q_o = annual loss of water through outlet

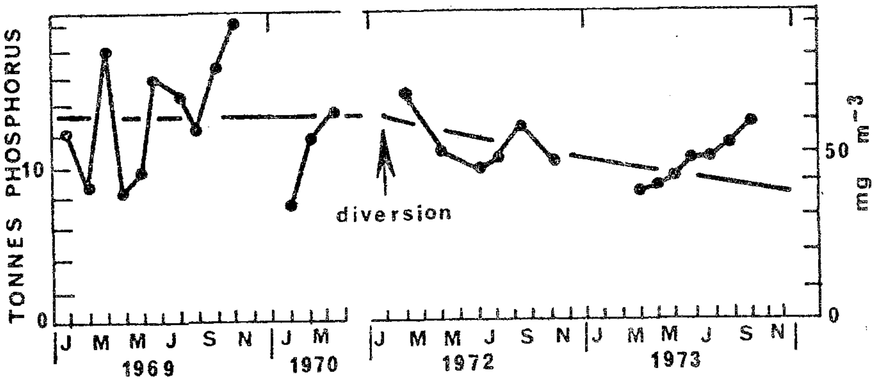

Figure 7. Quantities and concentrations of total phosphorus in Lower Lake Minnetonka before and after diversion of sewage effluents. Dashed line indicates the change computed with equation 11.

A = surface area of basin

S = net sinking velocity of phosphorus lost to sediments.

If $I = i/V$ and $(Q_o + SA)/V = 1/\theta_p$, where θ_p is a residence time for phosphorus, then,

$$dC/dt = I - (1/\theta_p)\ C, \tag{10}$$

and the solution for the initial conditions $C = C_o$ and $t = t_o$ is

$$C_t = I\theta_p + (C_o - I\theta_p)\ \exp - (t/\theta_p). \tag{11}$$

If the average concentration during 1969-1970 (62 mg m^{-3}) is assumed to be a steady state concentration, then the residence time of phosphorus may be computed from equation (10) where $dC/dt = 0$:

$$\theta_p = C/I. \tag{12}$$

The influx (I) was 60 mg m^{-3} yr^{-1}, hence $\theta_p = 1.1$ yr. If θ_p remains constant at the decreased rate of influx (13 mg m^{-3} yr^{-1}), then the concentration at the new steady state is given by equation (11) at $t = \infty$:

$$\lim_{t\to\infty} C_t = I\ \theta_p = 14 \text{ mg m}^{-3}. \tag{13}$$

Concentrations this low do not occur in lakes of the region, which suggests that the residence time of phosphorus is more than 1.1 yr. A more conservative estimate may be obtained by assuming that the residence time is 2 yr, which corresponds to a steady state concentration of 26 mg m^{-3} where I = 13 mg m^{-3}.

The time ($t_{0.5}$) required to achieve 50% of the new steady state if θ_p = 2 yr may be computed by rearranging equation (11):

$$\exp - (t_{0.5}/\theta_p) = C_t - I\,\theta_p/C_o - I\,\theta_p \tag{14}$$

$$= 0.5$$

$$t_{0.5} = 1.5 \text{ yr.}$$

Total quantities and mean concentrations of total phosphorus in the Lower Lake before and after the influx decreased are shown in Fig. 7. The average annual concentration decreased from 62 mg m^{-3} 1969 to 47 mg m^{-3} in 1973, which is 42% of the decrease required to attain the projected steady state concentration. The new steady state will be attained approximately seven years after the rate of influx decreased.

DISCUSSION

Lakes are difficult to classify on the scale from oligotrophic to eutrophic because the criteria for classification are ambiguous and subjective. As an alternative, we have suggested classifying lakes according to relative integral photosynthetic rates.[8] The relative integral photosynthetic rate depends upon how PhAR is partitioned between the phytoplankton and their environment, ranging from 0 (all PhAR attenuated by water) to 1 (all PhAR attenuated by phytoplankton). It is an objective basis for comparison, depending upon (1) concentrations of chlorophyll _a_, (2) the coefficient for attenuation of PhAR by a unit of chlorophyll concentration (ε_c), and (3) the coefficient for attentuation of PhAR by water and substances other than phytoplankton in the water (ε_w) (equation 8).

It is difficult to evaluate ε_w and ε_c, but they appear in equation (8) as a ratio, which equals the ratio $k_{w\ min}/k_{c\ min}$, where $k_{w\ min}$ and $k_{c\ min}$ are coefficients for the attenuation of the most-penetrating spectral region of PhAR.[10, 18] The total attenuation coefficient for the most-penetrating wavelength, $k_{min} = k_{w\ min} + k_{c\ min}\ c$, is easy to measure with a photometer equipped with suitable filters, and $k_{w\ min}$ and $k_{c\ min}$ may be estimated from the intercept ($k_{w\ min}$) and the slope ($k_{c\ min}$) of the equation for the linear regression of k_{min} on concentrations of chlorophyll.[11, 12, 19]

Relative integral photosynthetic rates at maximum population densities of phytoplankton in four Minnesota lakes (Lower Lake Minnetonka, Halsted Bay, Shagawa Lake, and Budd Lake) are compared with those in three British lakes (Windermere, Esthwaite Water, and Loch Leven) and two African lakes (Victoria and George) in Table 7. Most observers would probably consider Windermere (π_{rel} = 0.32) and Victoria (π_{rel} = 0.38) as oligotrophic or mesotrophic and the others (π_{rel} = 0.56-0.88)

Table 7. RELATIVE INTEGRAL PHOTOSYNTHETIC RATES

Computed for maximum concentrations (mg m^{-3}) of chlorophyll _a_ (c). The concentration $c' = \varepsilon_w/\varepsilon_c$ is the concentration of chlorophyll at which phytoplankton attenuate 50% of PhAR.

Lake	c	c'	π_{rel}	reference
Windermere, England	7	15	0.32	11
Victoria, Africa	5	8	0.38	20
Shagawa, Minnesota	108	83	0.56	8
Lower Lake Minnetonka				
1968	56	41	0.58	
1969	43		0.51	
1973	39		0.49	
1974	30		0.43	
Halsted Bay (1974)	89	62	0.59	8
Budd, Minnesota	170	60	0.74	8
Loch Leven, Scotland	250	78	0.76	19
Esthwaite, England	170	50	0.78	18
George, Uganda	1100	159	0.88	12

as eutrophic or polytrophic. The differences between these lakes depend primarily upon differences of maximum concentrations of chlorophyll (8-160 mg m^{-3}) and of ε_w (< 0.16 - 2.5 m^{-1}); ε_c varies only from 0.01 to 0.025 and it is usually about 0.02 m^2 mg Chl^{-1}. Therefore variations of ε_w are responsible for most of the variation of $c' = \varepsilon_w/\varepsilon_c$, the concentration of chlorophyll at which phytoplankton attenuate 50% of PhAR.

The relative integral photosynthetic rate at maximum population densities is misleading however, because the average relative rate for the lakes in temperate continental lakes is lower than the average for lakes in other climates, where there are relatively small seasonal variations of population densities. Thus, although the maximum relative rate in Lower Lake Minnetonka was 0.58 in 1968, the average during the open-water season was 0.35 in 1969 and 0.26 in 1973, less than the maxima for Windermere and Lake Victoria. Computations of mean relative integral photosynthetic rates would therefore be required for comprehensive comparisons; the only data required for such comparisons are periodic measurements of chlorophyll concentrations and of subsurface light.

SUMMARY

Maximum population densities of phytoplankton in Lower Lake Minnetonka were three or four times greater in 1968 and 1969 than in 1937. The linear regression of concentrations of chlorophyll a in the mixed layer on concentrations of total phosphorus was very significant during the summers of 1968 and 1969, indicating that population densities of phytoplankton during summer depended upon concentrations of total phosphorus.

The annual influx of phosphorus decreased from 0.5 to 0.1 g m^{-2} yr^{-1} when sewage effluents were diverted from the Lower Lake during the winter of 1971-1972. The mean concentrations of phosphorus decreased from 48 ± 7 mg m^{-3} during the summers of 1968 and 1969 to 34 ± 4 mg m^{-3} during the summers of 1973 and 1974, whereas mean concentrations of chlorophyll a decreased from 37 ± 3 mg m^{-3} to 17 ± 4 mg m^{-3} at the rate of 1.0 ± 0.3 mg Chl per mg of phosphorus decrease. Concentrations of chlorophyll were independent of phosphorus during the summers of 1973 and 1974.

The annual mean concentration of phosphorus decreased from 62 mg m^{-3} in 1969 to 47 mg m^{-3} in 1973, which is almost 50% of the decrease required to attain the new steady state concentration predicted with a balance equation for phosphorus in a perfectly-mixed basin. Virtually all the phosphorus lost from the lake since the influx decreased has been deposited in the lake sediments, because the quantity lost through the outflowing stream is very small.

The trophic state before and after the influx of phosphorus decreased is described objectively by changes in the relative integral photosynthetic rate, which indicates the fraction of photosynthetically active radiation attenuated by phytoplankton populations on a scale from 0 to 1. The relative integral photosynthetic rate is the integral photosynthetic rate relative to the rate attained by a population dense enough to attenuate all subsurface light. The maximal relative integral photosynthetic rates, attained by the densest populations in Lower Lake Minnetonka, were 0.58 in 1968 and 0.43 in 1974. The latter is somewhat higher than the maxima at highest population densities in Windermere, England, (0.38) and Lake Victoria, Africa (0.32). The mean relative integral photosynthetic rate during the ice-free season decreased 26% from 0.35 in 1969 to 0.26 in 1973, as mean concentrations of chlorophyll decreased from 22 ± 5 mg m^{-3} to 14 ± 4 mg m^{-3}.

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REPORT ON THE MINNEAPOLIS CITY LAKES

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I. INTRODUCTION

A. Past History

The five lakes discussed in this report make up the so-called Minneapolis Chain of Lakes. They are all within the Minneapolis city limits (see Figure 1). Four of them, from north to south, Brownie, Cedar, Isles, and Calhoun, are connected by channels made between 1910 and the early 1920's, but Lake Harriet, the southernmost, is isolated. Lake of the Isles was dredged extensively about 1920. The four upper lakes have had chronic low water problems for several decades because of their connection with the groundwater table, the level of which has not been high enough to sustain them. In order to resolve this low water problem, storm drainage from the surrounding city was directed into the lakes beginning in 1912 and continuing to the present. In addition, groundwater used by nearby companies for cooling purposes has been added to the four upper lakes. In recent dry years, water from the Mississippi River and city drinking water has been pumped in. As a consequence, particularly of the storm drain inputs, the lakes have become increasingly eutrophic in recent years.

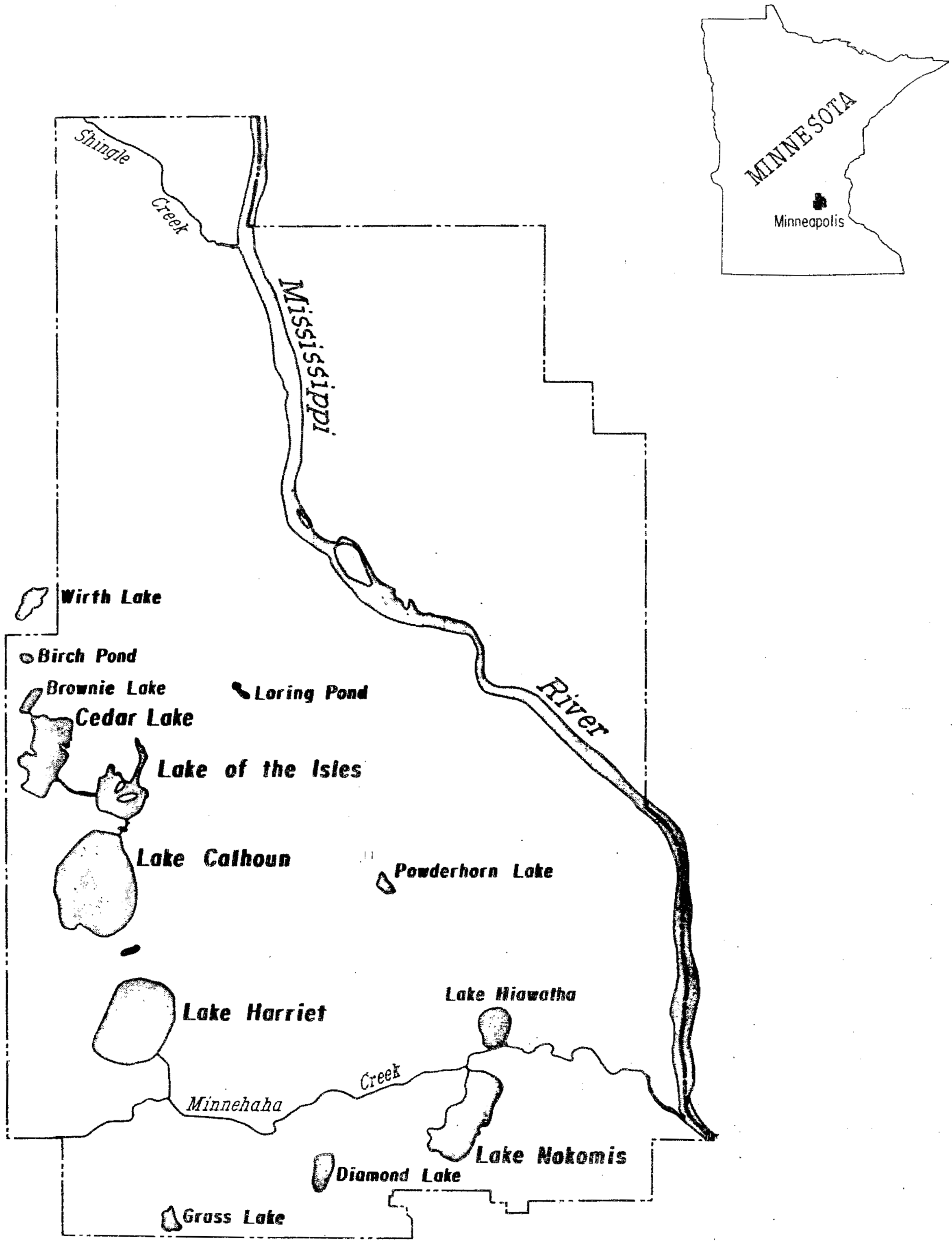

Fig. 1. Location map of study area.

Transparencies in Minneapolis Lakes in 1927 and 1971, m

Lake	July 29, 1927	July-August, 1971
Brownie	4.1	1.9
Cedar	3	1.2
Isles	2.6	.5
Calhoun	3.4	1.7
Harriet	4.1	3

II. Geography

A. Latitude and Longitude - 45°N 93°W

B. Altitude

The surface of the upper lakes is at an altitude of 260 m above mean sea level. Lake Harriet's surface is at an altitude of 258 m.

C. Catchment Area

Lake	Area, ha (inc. lake)
Brownie	47
Cedar	163
Isles	285
Calhoun	761
Harriet	480

D. Climatic data

Mean monthly air temperatures (°F) are approximately as follows:

	J	F	M	A	M	J	J	A	S	O	N	D
High	22	25	35	55	65	75	84	82	70	58	43	27
Low	4	7	20	36	45	58	64	62	52	43	25	14

The lakes begin to freeze over in November and breakup generally occurs in mid-April. Ice thickness may reach 75 cm. Snow may be present from October to May but the peak snowfall is from November to March. Total snowfall may reach 150 cm. Approximate average monthly precipitation as follows (in cm):

J	F	M	A	M	J	J	A	S	O	N	D
2.0	2.0	3.6	6.1	8.4	10.7	8.6	8.6	7.6	4.6	3.6	2.3

Summer winds average 190° (SSE) with a mean velocity from April to October of 14.6 kph. Evaporation exceeds precipitation by 12.7 cm annually and by 17.8 cm between April and October. Total evaporation averages about 99 cm per year.

E. Geology

All of the lakes are ice block lakes formed about 11,000 years ago. They are embedded in sand and gravel with some clay. The bottom sediments are typical deep-water sediments of productive lakes, total thickness unknown. Sediments in Lake of the Isles are only about .5 m deep and are underlaid by undecomposed plant remains as this lake was essentially a swamp until its dredging in the 1920s. Land erosion in the catchment area is probably negligible as the whole area is urban and has long been settled. Pollen analysis of the deep waters of Lake Calhoun and Harriet shows a sedimentation rate of about 3 mm per year.

F. Vegetation

Virtually all of the area not covered by impervious surfaces such as houses, roads, walks, and alleys is planted in lawn grasses. A large number of trees, especially elms and oaks, is present in the catchment area.

G. Population

Population density is high--about 1550/km^2.

H. Land use

Excluding water areas, 83% of the area surrounding the lakes is urban, and includes residential use, commercial and service use, institutional use, and transportation. The remaining 17% is open land, and includes parks, a golf course, a cemetery, islands in Lake of the Isles, and grassy areas immediately surrounding the lakes.

I. Water use

The lakes are used primarily for fishing, swimming, sailing, and canoeing. No water is taken for drinking purposes.

J. Sewage and effluent discharge

No sanitary sewage or industrial effluents reach the lakes.

III. Morphometric and hydrologic characteristics

A., B., C., D., and E. See Table below.

Lake	Area (ha)	Max. depth (m)	Mean depth (m)	Volume $10^6 m^3$	Est. % area less than 3 m deep	Epilimnion thickness, m July - Aug
Brownie	7.3	15	6.8	0.5	--	3
Cedar	69	15	6.1	4.22	15	4
Isles	42	11	2.7	1.12	80	4
Calhoun	170	27	10.6	18.0	5	6
Harriet	143	26	8.8	12.5	15	6

F. Duration of stratification

The lakes are stratified for approximately six months from late April to late October.

G. Sediments

The sediments are highly organic. Lake Calhoun has 2000 micrograms P/gram dry weight in the surface sediments and 800 micrograms P/gram dry weight below 10 cm. Lake Harriet has 3000 micrograms P/gram dry weight at the surface, falling to 800 at 25 cm, then rising again to high and variable concentrations of between 2000 and 6000 grams P/gram dry weight.

H. Variation of precipitation

See IID above.

I. Inflow and outflow of water

Lake	Total inflow $10^3 m^3$	% lost to groundwater	% lost to evaporation
Brownie	252	79	21
Cedar	1281	61	39
Isles	1812	82	18
Calhoun	5012	75	25
Harriet	5157	81	19

There are no functional surface outlets. The percentage of groundwater loss was determined by resolving the hydrologic budget, i.e. by accounting for all inputs and outputs except groundwater. Because the upper lakes are connected, this procedure may not be entirely correct. If, for example, Lake of the Isles has a completely impervious basin, its excess water will run to Lake Calhoun which would then have a higher loss to groundwater than was calculated. This would not change the water residence time for Isles but Lake Calhoun would appear to have a longer residence time than it really has.

J. Currents

No currents are known.

K. Water retention time

With the possible error noted in I above, retention times are as follows:

Lake	Water retention times (yrs)	
Brownie	1.98	(probably less for the mixolimnion because of meromixis)
Cedar	3.30	
Isles	.62	
Calhoun	3.59	
Harriet	2.43	

IV. Limnological characterization

A. Physical

1. Temperature

All of the lakes stratify thermally. Surface temperatures range up to 26°C in summer but typically are 20-22°C. Bottom temperatures are 5-7°C except in Brownie Lake which is meromictic from road salt. Its bottom temperatures are higher than those at 6-8 m which are at 4°C. The mixolimnion extends down to 4 m in Brownie Lake.

2. Conductivity

Lake	1974 range in surface specific conductance (micromhos/cm)
Brownie	400-475
Cedar	400
Isles	380-470
Calhoun	400-500
Harriet	360-425

3. Light

No light measurements were made other than Secchi disk transparencies.

4. Color

The lakes have no apparent color.

5. Solar radiation

No measurements were made.

B. Chemical

1. pH

Epilimnetic pH values are high throughout the growing season.

Lake	1971-72 pH maximum in surface waters
Brownie	8.91
Cedar	9.30
Isles	9.49
Calhoun	9.10
Harriet	8.81

2. Dissolved oxygen

All of the lakes have anoxic hypolimnia from late May until turnover in October-November. Anoxia in Isles begins somewhat earlier. Brownie Lake has an anoxic hypolimnion year round because of its meromixis.

3. Phosphorus

a. Orthophosphorus-P

Calhoun, Harriet, and Cedar surface waters contain 5 ppb or less PO_4-P during the summer. Isles and Brownie surface waters contain 10 ppb or less during the summer. Bottom water concentrations are as follows:

Lake	PO_4-P, micrograms/l
Brownie	>1600
Cedar	738
Isles	601
Calhoun	379
Harriet	255

b. Total P

Lake	Surface range, ppb P	Mean concentration whole lake
Brownie	30-40	--
Cedar	30-40	55
Isles	70-100	110
Calhoun	40-50	106
Harriet	40	62

4. Nitrogen

No total nitrogen figures are available. Surface NO_3-N is less than 5 ppb in the summer. Surface NH_3-N is less than 50 ppb in the summer.

5. Alkalinity

Lake	surface ranges 1971-1972 meq/l
Brownie	2.47-2.72
Cedar	1.41-2.18
Isles	1.36-2.62
Calhoun	1.59-2.27
Harriet	1.84-2.47

6. Major ions

	Surface values 11/16/71						
Lake	Na	K	Ca	Mg	SO_4	Cl	HCO_3
Brownie	3.11	.06	1.99	1.04	.28	3.64	2.65
Cedar	1.38	.08	1.63	1.03	.42	1.61	2.26
Isles	1.64	.09	1.63	1.01	.33	1.91	2.31
Calhoun	1.64	.10	1.55	1.22	.31	1.93	2.36
Harriet	1.21	.10	1.63	1.02	.21	1.54	2.41

Total iron concentrations in all the lakes averaged about 20 ppb.

7. Trace metals

No determinations were made.

C. Biological characteristics

1. Phytoplankton

a. chlorophyll a

Lake	1971 surface values ppb	mean surface concentration July-August 1971
Brownie	4.3-24	5.6
Cedar	2.4-27	20
Isles	15-72	53
Calhoun	3.4-37	6.0
Harriet	1.2-27	3.5

b. Primary production

No measurements.

c. Algal assays

No algal assays as such. Determinations of alkaline phosphatase activity show low values until late July and high values through September.

d. Identification and count

The five dominant algae in 1971-72 are listed below:

Lake	Algae
Brownie	Fragilaria crotonensis
	Mougeotia sp.
	Asterionella formosa
	Cryptomonad sp. 3
	Oocystis spp.
Cedar	Scourfeldia cordiformis
	Anabaena planctonica
	Cryptomonad sp. 3
	Oscillatoria agardhii
	Aphanizomenon elenkinii

Lake	Algae
Isles	Scourfeldia cordiformis Aphanizomenon flos-aquae Anabaenopsis raciborskii Oscillatoria agardhii Asterionella formosa
Calhoun	Aphanizomenon flos-aquae Anabaena planctonica Stephanodiscus niagarae Stephanodiscus-Cyclotella spp. Cryptomonad sp. 3
Harriet	Aphanizomenon flos-aquae Ceratium hirundinella Oocystis spp. Cryptomonad sp. 3 Stephanodiscus niagarae

Lake	Volume % of blue-greens during July and August 1971
Brownie	< 5
Cedar	99
Isles	99
Calhoun	95
Harriet	78

2. Zooplankton

a. Identification and count

	Numbers of species in each lake			
Lake	Cyclops	Diaptomus	Daphnia	Chydorus sphaericus
Cedar	1	1	1	present
Isles	1	1	4	present
Calhoun	3	2	5	present
Harriet	1	1	3	present

The numbers are very variable.

3. Bottom fauna

Bottom fauna was very sparse in all the lakes.

4. Fish

Fish in the lakes are mostly yellow perch, blue-gill sunfish, and black crappies. Some northern pike are present and bass are abundant in Cedar and Isles. Isles has many carp.

5. Bacteria

Unknown.

6. Bottom flora

Unknown.

7. Macrophytes

Brownie	A ring of *Nuphar variegatum* to a depth of 1.8 m.
Cedar	*Nuphar* and *Nymphaea* with *Potamogeton* and *Ceratophyllum* in significant quantities.
Isles	*Ceratophyllum* and *Potamogeton* in water of less than 1.8 m.
Calhoun	*Potamogeton* and *Ceratophyllum* in less than 1.8 m.
Harriet	*Ceratophyllum* and *Elodea* in less than 2.5 m.

Cedar Lake is the only lake in which the weeds are very abundant.

V. Nutrient budget summary

A. Phosphorus

	kg/year/lake in 1971				
Sources	Brownie	Cedar	Isles	Calhoun	Harriet
Waste discharges (includes city water and air conditioning water)	24.6	0.2	0	13.3	0
Land runoff (via storm drain and direct)	57.5	205	828	1357	890
Estimated precipitation	3.8	36	23	91	72
Estimated ground-water input	0	0	0	0	54
Total	85.9	241	851	1461	101

B. Nitrogen

No data are available.

C. Other budgets

None.

VI. Discussion

A. Limnological characteristics

B. Trophic state

By most criteria all of these lakes would be classed as eutrophic. Data from 1933 show considerably lower concentrations of algae, lower pH, higher dissolved oxygen, and higher transparency. There is no question that the addition of storm runoff has been responsible for the changes.

C. Trophic state vs. nutrient budgets

Lake	grams $P/m^2/yr$	Mean depth/detention time
Brownie	1.18	3.43 (2.02 mixolimnion)
Cedar	0.35	1.85
Isles	2.06	4.35
Calhoun	0.88	2.95
Harriet	0.71	3.62

Plotting the results from the above table would suggest that the lakes should form a series with Isles being most eutrophic and Cedar least so. While some indicators of trophic state would corroborate this, e.g. total P, others, such as chlorophyll, epilimnetic pH, and transparency would not, i.e. Isles is most eutrophic on any basis but Cedar is least so with some, and not least eutrophic with others. Furthermore, the situation changes from year to year. Thus, if summer chlorophyll concentrations are used as the index, Lake of the Isles appears to be becoming less eutrophic in recent years while Calhoun and Harriet, after several years of lessened eutrophy, appear to be becoming more eutrophic (Figure 2). These chlorophyll data are substantiated by transparency measurements and measurements of algal abundance as shown in the table. So far as is known, the nutrient budgets of the lakes have not changed in recent years. Therefore, the question arises, why have the lakes undergone these changes? A variety of hypotheses have been tested and discarded. For example, neither Lake Harriet nor Lake Calhoun appear to have more nitrogen fixing algae in 1974 than in 1971-72, as judged by heterocyst frequencies. Neither has there been a related change in either total rainfall or the seasonal pattern of rainfall that brings the nutrients into the lake. It appears rather that changes within the lakes themselves are responsible for the changes in the manifestation of eutrophy, i.e. changes have occurred despite the fact that the total phosphorus has remained constant (see table for Lake Harriet data).

One explanation that appears likely is that the algal abundance is being affected by the grazing of zooplankton and that the higher chlorophyll concentrations occur as a result of less grazing. Substantiation for this is shown in Figure 3 where the transparency in Lake Calhoun during 1973 appears to correlate very well with the abundance of Daphnia. If this is a correct explanation then it implies that zooplankton grazing pressure in the lakes has been changing. This in turn could be

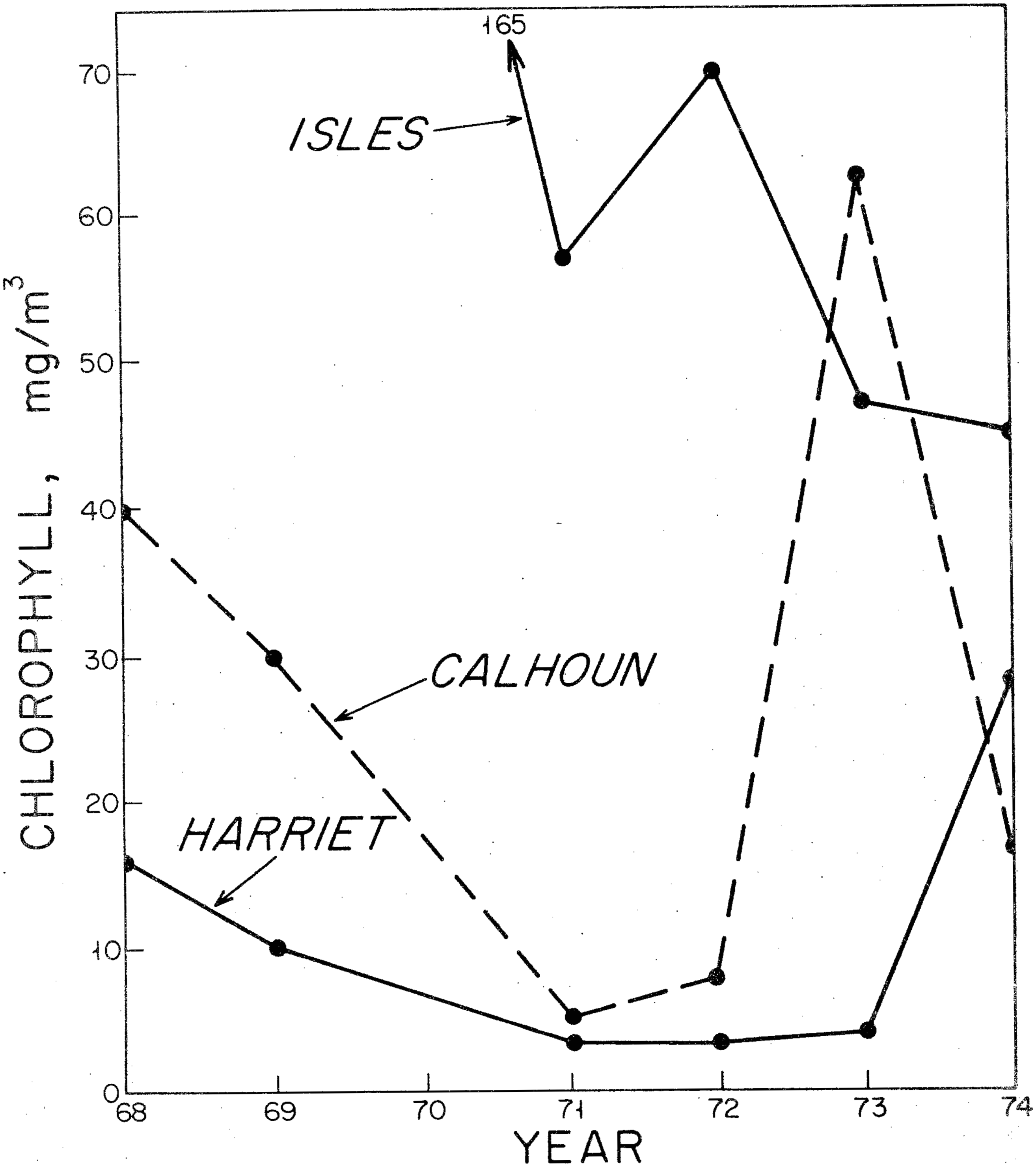

Fig. 2. Mean surface chlorophyll concentrations for the period July-September.

Lake Harriet

Date		Secchi Disk (feet)	Surface Chlorophyll (ppb)	Surface Algae (mg/l)	Surface Total P (ppb)
1968	7/23	-	16.0	-	-
1969	7/29	5.0	10.0	-	-
1971	7/19	11.3	1.7	4.4	41
	8/2	5.2	4.5	-	43
	8/24	5.6	2.8	1.1	41
	9/13	5.9	3.2	0.61	42
1972	7/6	14.0	2.4	0.61	37
	7/24	11.0	2.8	0.33	37
	8/9	9.3	4.2	1.1	39
	8/22	8.4	3.4	1.5	38
	9/13	10.0	4.0	0.70	38
1973	7/11	10.0	(3.9)	-	35
	7/25	7.0	3.9	-	40
	8/23	9.0	-	-	35
	9/19	8.8	-	-	22
1974	7/22	5.5	24.0	10.2	64
	8/20	3.5	47.4	12.0	45
	9/16	6.6	14.8	10.2	42

( ) = values from 2.5 m, not used in average.

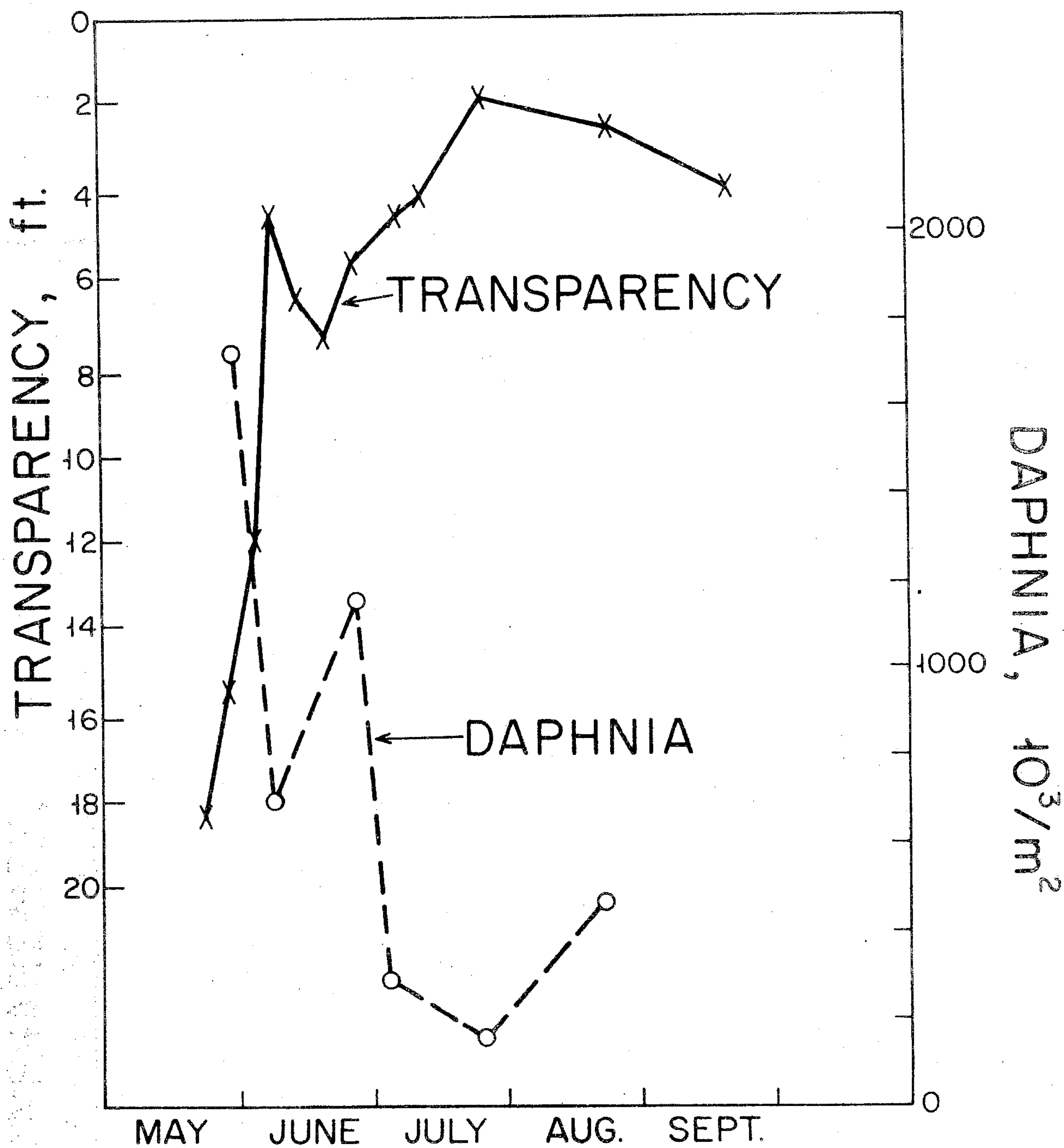

Fig. 3. Relationship between Secchi Disk Transparency and Daphnia abundance in Lake Calhoun during 1973.

a result of changes in the populations of such fish as yellow perch which are zooplanktivorous. In fact, we have recently begun a program to test the possibility of using carnivorous fish to control zooplanktivorous fish, so that zooplankton grazing could increase and so help control algae.

Because of such biological effects as suggested above on the manifestations of eutrophication, it appears extremely unlikely that the loading concept will ever be linked in a precise fashion with trophic state.

SECTION III - NEW YORK

A DESCRIPTION OF THE TROPHIC STATUS AND NUTRIENT LOADING FOR LAKE GEORGE, NEW YORK

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I. INTRODUCTION

Lake George is located in the eastern Adirondack Mountains of New York State (Fig. 1), and has been under investigation by scientists and engineers of the Rensselaer Fresh Water Institute as well as by other educational and governmental bodies within the region. The lake has served as an aquatic site for the Eastern Deciduous Forest Biome of the International Biological Program. Much of the data presented was collected as part of that multidisciplinary, ecosystem-wide study.

Lake George lies in a glacial-scoured basin of Precambrian metamorphic, plutonic and igneous rock, with small patches of Cambrian deposits mainly at the southern end of the basin. Most of the drainage basin is covered with shallow soil from glacial debris with numerous outcroppings present.

Prior to the colonization of the New World, Lake George was part of a natural trail, and the site of numerous Indian conflicts. Its strategic location between the Hudson River and Lake Champlain made it an area of battle in both the French and Indian Wars and the Revolutionary War.

During the latter part of the nineteenth century, mining operations in the region produced a representative supply of the nation's high-grade graphite as well as some iron ore. An active logging industry was also present at this time which supported several mills in the Village of Ticonderoga, located on the extreme north end of Lake George. Virtually all this industry, however, ceased within the first three decades of the twentieth century.

It has been replaced by a flourishing tourist trade, drawn by the beauty of the lake and its scenery. The resort aspects of the area were enhanced by the construction of the Adirondack Northway in 1967, which made the lake far more accessible to the large urban areas to the south and north.

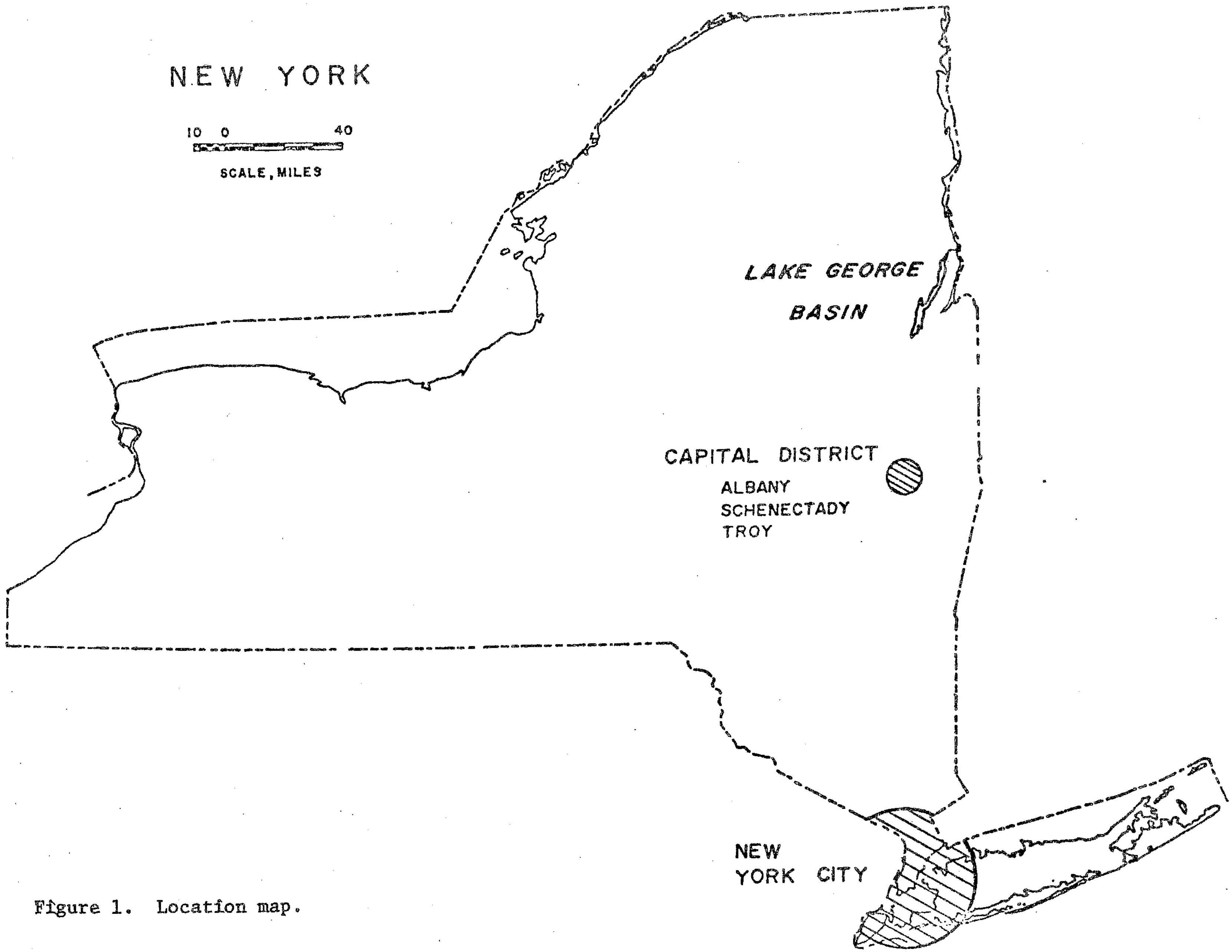


Figure 1. Location map.

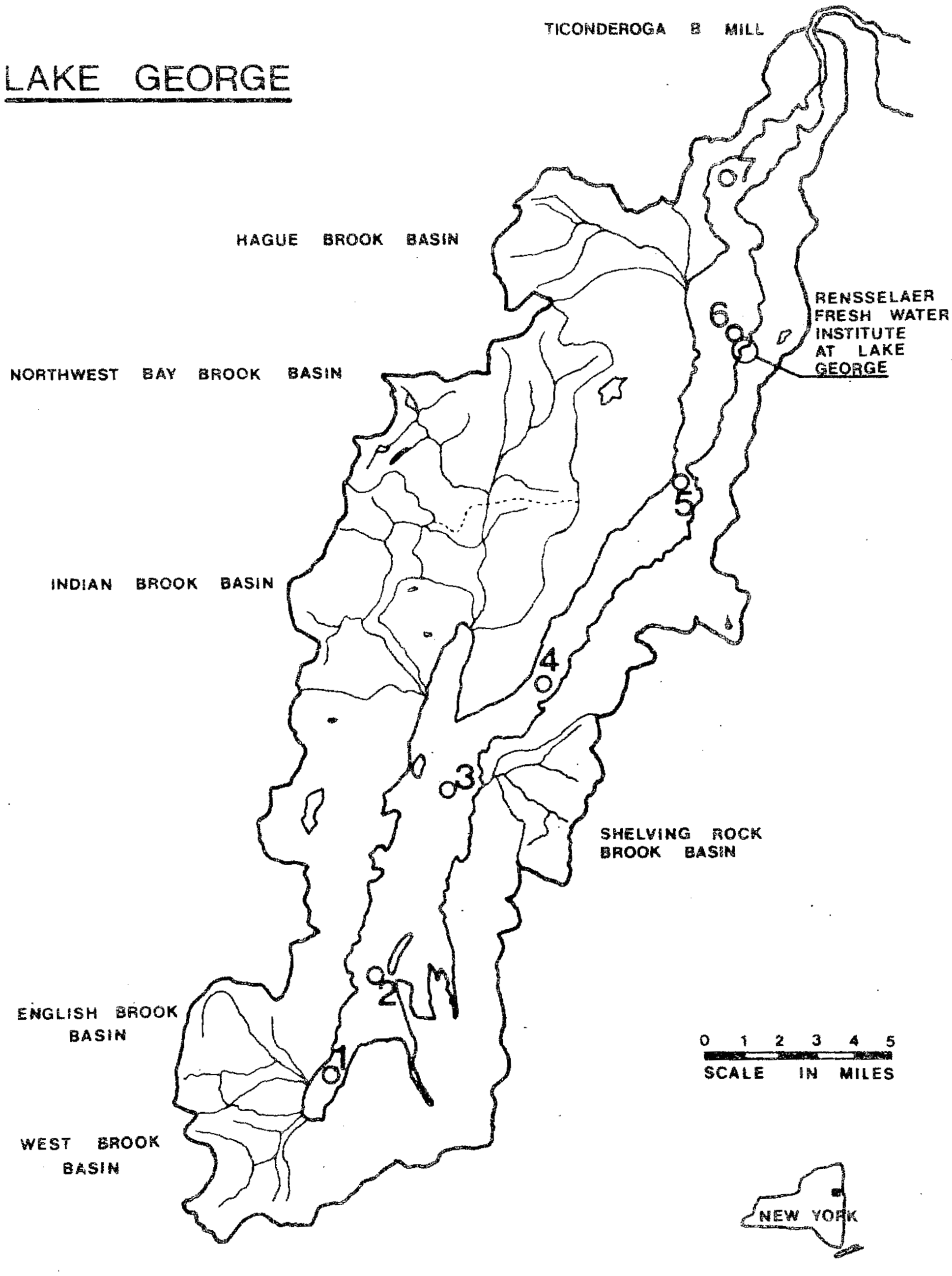


Figure 2. Location of Lake George Sampling Stations.

Not located on the map are the following stations:

1. Smith Bay and Burnt Point are located immediately east of Station 6.
2. Lake George Village is located in the extreme southwestern corner of Lake George (in the West Brook drainage basin).
3. Tea Island is located immediately to the west of Station 1.
4. Diamond Island is located immediately to the south of Station 2.

II. BRIEF GEOGRAPHIC DESCRIPTION OF WATER BODY

Lake George is a relatively large lake located in the south-eastern Adirondack Mountain region of New York State. It lies within the basin boundaries of latitude 43°22' and 43°51' North and longitudes 73°24' and 73°47' West. The lake surface stands at 97 meters above sea level, and encompasses 114 km^2. The drainage basin surface area is 492 km^2, giving a total catchment area of 606 km^2. Thus, the tributary watershed to lake surface ratio is only 4.3.

In general, the climatology of Lake George is typical for the humid continental climatic region of the Northeastern United States (Stewart, 1971, 1972). 1970-71 monthly air temperatures are presented in Table 1 for the Lake George basin. Long term averages at Glens Falls, located approximately seven miles south of Lake George are -7.2°C in January and 21°C in July.

Wind pattern analysis by Stewart (1972) show that for the period of September 1971 through August 1972, wind speed averaged 5.65 ± 0.65 knots (based on monthly averages). Wind direction is from the south or southwest during the warmer months, but shifts to the north or northwest in November, December, February, March and April.

Lake evaporation and evapotranspiration figures for the Lake George catchment area are presented in Table 2. Evaporation has been calculated using the Penman Method and the evapotranspiration from a water balance of the active soil zone.

General Geologic Characteristics

Lake George occupies a graben in Precambrian bedrock. This bedrock consists of plutonic, metamorphic and igneous rock, for example, gneisses and schists, syenite, granite and gabbro. At a few places along the shore of the southern Lake George basin are exposures of Cambrian sandstones (Potsdam sandstone) and dolostones (Little Falls dolomite).

The linear straight shorelines and sheer slopes are the combined effect of erosion following prominent faults and a deepening of the fault-controlled valleys by the sweep of the Pleistocene glaciers which deepened the rock channels. Prior to glaciation, two rivers drained the Lake George basin. One stream originated in the narrow trench now occupied by Northwest Bay Brook and flowed into the southern Lake George basin; the second river flowed from the Narrows northward. A preglacial divide existed where the Narrows are now located. When the glaciers plowed their way through the deep narrow Lake George Valley they

TABLE 1

AVERAGE MONTHLY AIR TEMPERATURE (°C) FOR LAKE GEORGE, N.Y.

	South Basin: Lake George Village			North Basin: Burnt Point		
Month	1969	1970	1971	1969	1970	1971
January			-11.7			- 9.1
February		- 5.3	- 5.6		- 5.1	- 4.6
March	- 0.2	- 0.7	- 2.9		- 0.8	- 2.1
April		7.3	4.3			
May		13.5	13.1			12.9
June		18.3				
July		20.0				
August		20.6			20.3	
September		16.4			15.7	
October		11.3			10.9	
November		5.4			5.2	
December		- 7.5		- 4.0	- 5.8	

(From Colon,1972)

Table 2. EVAPORATION AND EVAPOTRANSPIRATION (1971 WATER YEAR) FOR LAKE GEORGE, N.Y.*

Date	Lake Evaporation		Evapotranspiration	
Month	Inches/day	Inches/month	Inches/day	Inches/month
October	0.06	1.91	0.06	1.74
November	0.04	1.18	0.04	1.11
December	0.03	0.80	0.03	0.78
January	0.02	0.67	0.02	0.63
February	0.03	0.81	0.03	0.74
March	0.05	1.42	0.04	1.26
April	0.07	2.19	0.06	1.91
May	0.11	3.51	0.10	3.06

*(Colon, 1972)

deepened the Narrows by ice erosion. The waters of Lake George are now held in place by Pleistocene glacial sediments which block the river outlets at the north and south end of the lake. At the south end of the lake glacial sand and gravel deposits rise 500 feet above lake level. After the retreat of the glaciers Lake George was a glacial lake as evidenced by the presence of varved clay flooring the bottom of the lake in the Narrows; this varved clay also occurs above the present lake level at elevations up to 750 to 800 feet.

Surficial sediments of the Champlain basin of which Lake George forms a part have been mapped. Sand and gravel are abundant in the delta and ice-contact gravels southwest of Lake George Village (Schoettle and Friedman, 1971).

Vegetation

Hemlock (72% of stands), sugar maple (69%), white pine (64%), red maple and northern oak (57%) are the most frequently encountered of 35 tree species occurring in 75 randomly selected stands in the Lake George drainage basin. Hemlock leads in density (32% stands), white pine (13%), beech (12%), northern red oak (9%), and red/sugar maple

(8%). Distribution patterns of hemlock and pine shows the former is most abundant in sloping stands at the lowest elevation (100 m) and generally prevail on the east side of the basin, while white pine is best represented in level stands about 200 m, but uncommon on the east side. Forest composition of our random sample for the drainage basin differs slightly from 1970 estimates by Northeast Forest Experiment Station in that pine-hemlock stands are more common (42%-18%) and elm-ash-red maple and spruce-fir less common (3%-17% and 0%-7%) (Nicholson and Scott, 1972).

Population - See Tables 3 and 4.

Land Usage - Data are not available.

Use of Water

Primarily drinking, aesthetics, sport (i.e., boating, fishing, SCUBA diving, swimming, etc.), and all other recreational purposes.

Sewage and Effluent Discharges

The types of wastewater discharges in the Lake George drainage

Table 3. POPULATION DISTRIBUTION IN THE LAKE GEORGE, N.Y. BASIN*

Population Type	South Lake Basin Number Sewered	South Lake Basin Total Number	North Lake Basin Number Sewered	North Lake Basin Total Number
Permanent, Year-Round	2,930	4,445	0	1,130
Summer Camp	1,750	8,775	0	3,205
Resort Hotel and Motel	9,111	12,558	0	47
Total Avg. Summer	13,791	25,778	0	4,382

* Compiled from 1970 Census data.

Table 4. TOTAL POPULATIONS AND THOSE SERVICED BY SEWERS IN THE LAKE GEORGE DRAINAGE BASIN.[3]

	Permanent Year Round[1] Population		Summer Camp Population[2]		Motel & Hotel Resort Population		Total Average Summer Population	
	Total	Sewered	Total	Sewered	Total	Sewered	Total	Sewered
Warren County								
Lake George Town	2630	2130	2000	1500	10215	8661	14845	12291
Bolton "	1165	800	2400	250	2343	450	5908	1500
Hague "	640	0	1425	0	47	0	2112	0
Queensbury "	410	0	2375	0	0	0	2785	0
Lake Luzerne "	0	0	0	0	0	0	0	0
Warrensburg "	0	0	0	0	0	0	0	0
Horicon "	10	0	0	0	0	0	10	0
Washington County								
Fort Ann Town	230	0	2000	0	0	0	2230	0
Putnam "	150	0	700	0	0	0	850	0
Dresden "	190	0	1050	0	0	0	1240	0
Essex County								
Ticonderoga Town	150	0	30	0	0	0	180	0
Total	5575	2930	11980	1750	12605	9111	30160	13791

1. Data were adjusted to conform to drainage basin lines by the Env. Quality Research and Development Unit, New York State Dept. of Environmental Control.

2. A normal summer occupancy of 5 persons per camp was assumed.

3. Aulenbach & Clesceri 1972.

basin are: 1) secondary treated (trickling filter plant) from the Village of Lake George Sewage Treatment Plant onto natural sand beds, 2) primary treated (Imhoff tank) discharged onto natural sand beds from the Town of Bolton facility, 3) septic tank-leach field effluent, and 4) pit privy discharge. There is no industrial discharge. Population data relative to this are seen in Table 4.

III. MORPHOMETRIC AND HYDROLOGIC DESCRIPTION OF WATER BODY (at 97.25 m or 319 ft. amsl)

Surface Area of Water - 114 sq. km (44 sq. mi.)

1. Length - 51 km (32 mi.)
2. Width - Maximum = 4.0 km (2.4 mi.)
   Average = 2.3 km (1.4 mi.)
3. Shoreline Length - 209.6 km (131 mi.)

Volume of Water - 2.1 km^3 (0.5 $mi.^3$)

Regulation - Lake George Water Levels (as described in Section 38 of the New York State Navigation Law)

Any dam or other similar structure so located in the outlet of Lake George as to affect the water levels of the lake shall, with due allowance for fluctuations due to natural causes or to emergencies and for a reasonable use of water for power and for sanitary purposes, be operated in such a manner as to maintain the waters of the lake from the first day of June to the thirtieth day of September in each year as nearly as may be at an average level of three and five-tenths feet on the gage of the United States Geological Survey at Rogers Rock on Lake George, known as Rogers Rock gage, and in such a manner as to maintain the waters of the lake from the first day of October to the first day of December at a level which shall not fall below two and five-tenths feet on said gage; and, consistent with the above mentioned fluctuations and reasonable use, the waste gates of any such dam or other structure shall be operated so that, to the extent possible, the waters of the lake will not be permitted to rise above a level of four feet on such gage at any time during the year or to fall below a level of two and five-tenths feet on said gage at any time after the first day of June and prior to the first day of December in any year. If at any time during the year the waters of the lake shall rise above such level of four feet any person owning or operating such dam or other structure shall immediately open the waste gates thereof and take such other appropriate action as in the judgment of the superintendent of public works may be necessary to lower the waters of the lake with the least practicable delay to a level not higher than four feet on said gage. If at any time after the first day of June and prior to the first day of December in any year the waters of the lake shall fall below such level of two and five-tenths feet such person shall immediately close the waste gates of such dam or other structure; and no person shall withdraw water from the lake for the purpose of generating power during any period of time between the first day of June and the first day of October in any year when the level of the waters of the lake is below two and five-tenths feet on said gage. The superintendent of public works or his duly authorized representative shall at all times have access to such dam or other structure and is hereby authorized and directed to operate the waste gates thereof whenever necessary for the purpose of carrying out the provisions of this section. The superintendent of public works shall establish such rules and regulations as in his judgment may be necessary for the enforcement of the provisions of this section, and he is hereby authorized to enter into such agreement or agreements with any person or persons owning or operating any such dam or other structure as in his judgment may be necessary in order to carry into effect the provisions of this section and of such rules and regulations. In addition, the superintendent of public works shall, once in each year during the first week in July, cause to be published in at least three daily newspapers serving the area the reading on the Rogers Rock gage on the first day of July in that year. Any person violating any provision of this section or of any rule or regulation established or of any agreement entered into pursuant thereto shall for every such violation forfeit to the people of the state the sum of not to exceed two hundred and fifty dollars to be recovered in a civil action.

Maximum and Average Depths - See Table 5 (Colon, 1972; Langmuir, et al., 1966).

Table 5. MAXIMUM AND AVERAGE DEPTHS FOR LAKE GEORGE, N.Y.

Basin	Maximum Depth	Average Depth
North	53.3 m (175 ft.)	20.5 m (67.3 ft.)
South	58 m (191 ft.)	15.5 m (50.9 ft.)
Total Lake	58 m (191 ft.)	18 m (59 ft.)

Location of Exceptional Depths and the Surface Area Ratio of Deep to Shallow Waters - These data are not available.

Ratio of Epilimnion over Hypolimnion - These calculations are not available.

Duration of Stratification - This phenomenon occurs in Lake George for approximately 150 to 180 days (i.e., from May 1 through October 31).

Nature of Lake Sediments

Most of the sediments of Lake George consist of silty clay; pure sand lies mostly near the shore, yet most sand also contains silt and clay in nearly equal amounts. In the south basin sediments containing more than 50 percent clay occur near the east shore and underlie the large central expanse of the lake. Sediments with less than 25 percent clay (hence mostly sandy) are restricted to the west shore of the south basin, although in two places a tongue of sandy sediment is present in the central area of the south basin. Sediments underlying the eastern Narrows are rich in clay, whereas those beneath the western Narrows are generally rich in sand. The southern part of the north basin is underlain by clay-rich sediments. In the central part of this basin clay floors the middle of the lake and sand is found closer to shore. In the northernmost part of the north basin, near Ticonderoga, the sediment consists mostly of sand (See Figure 3).

In the south basin most of the bottom sediments contain between 5 and 10 percent organic carbon. However, close to and in bays of the east shore the organic carbon content exceeds 10 percent. By contrast, near the west shore and in two tongues in the central part of the south basin the organic carbon content is $<$ 5 percent. The sediments of the Narrows are mostly depleted in organic carbon, whereas the sediments of the north basin contain between 5 and 10 percent organic carbon in the center, but $<$ 5 percent near the shore. Near Ticonderoga the sediments of the northernmost part of Lake George contain $<$ 5 percent organic carbon. The muddy bottom sediments of Lake Champlain, contiguous to Lake George, contain 5 to 20 percent organic carbon; organic mud covers about three-quarters of its bottom (See Figure 4).

Many values of organic carbon exceed 10 percent and most sediments contain between 5 and 10 percent organic carbon. These high values indicate that a large part of the clay-size fraction consists of organic matter. To compute organic matter from organic carbon a factor of 1.72 is used, so that in most sediments between 8.6 and 17.2 percent organic matter is present. Examination under the binocular microscope shows that the organic matter in the nearshore sediments consists largely of leaves, needles, tree bark, and spore capsules. In deeper water sediments, however, the fabric of organic matter usually cannot be identified because of advanced decomposition. In the clay-size fraction quartz and clay minerals including illite and chlorite with traces of kaolinite are found. In the cores studied the same clay-mineral suite occurs unchanged throughout the cores. The clay is derived from the local metamorphic and igneous bedrock and the glacial sediments.

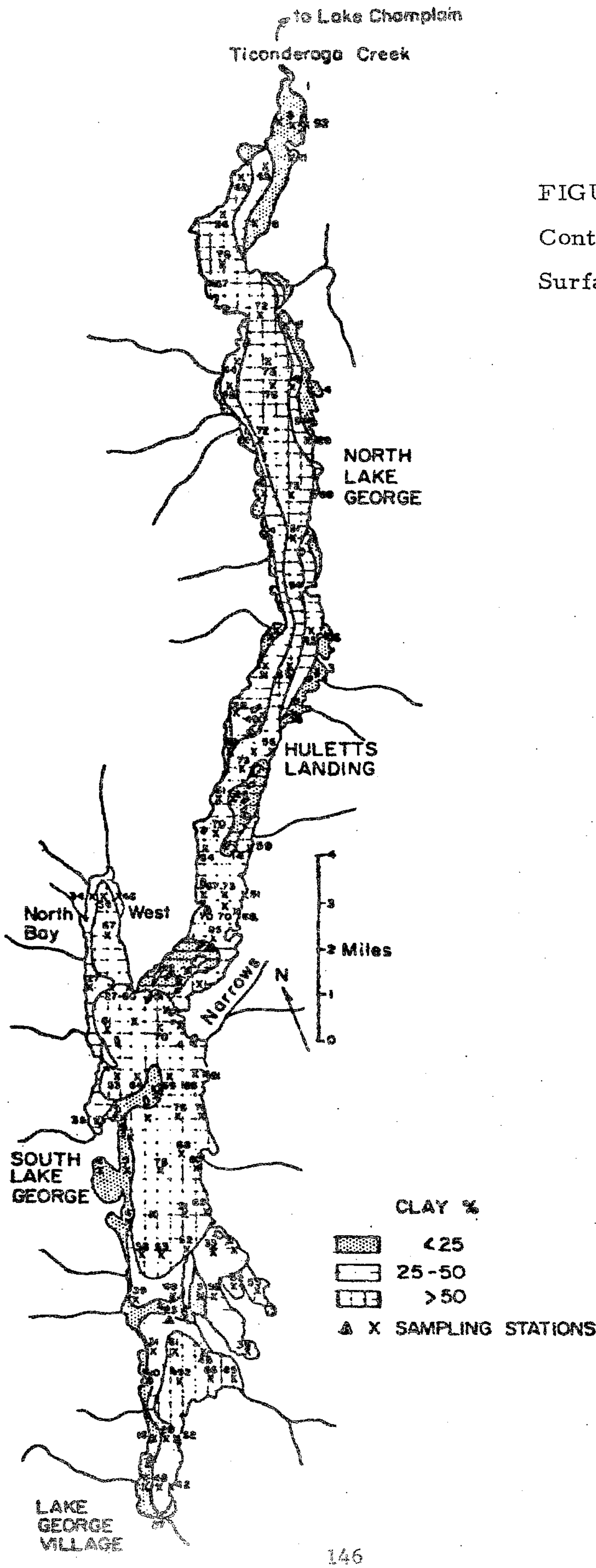


FIGURE 3. Clay Content Of Lake George Surface Sediments.

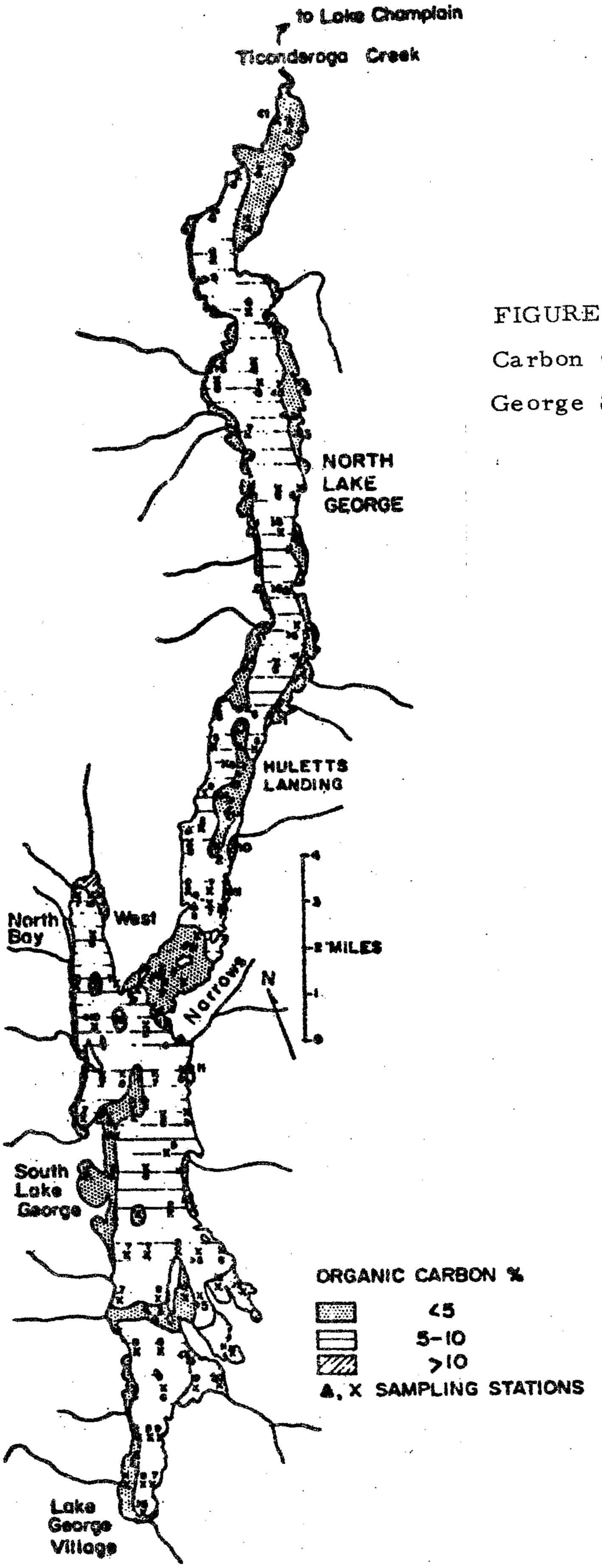


FIGURE 4. Organic Carbon Content of Lake George Surface Sediments

In the sand the light minerals are quartz and feldspars (plagioclase, orthoclase), some microcline, muscovite and biotite. The heavy mineral fraction is dominated by garnet; less abundant heavy minerals include hornblende, sillimanite, epidote, hypersthene, augite, staurolite, kyanite, zoisite, zircon, tourmaline, rutile, titanite and iron-rich biotite.

Except at water-sediment interface all sediment color is black. There the color is either black or brown; the brown color of fine-grained sediment passing downward into black. Black color at the interface dominates near the east shore in the south basin, especially near the bays, whereas brown color is present near the west shore.

The sediments in the Narrows and contiguous areas consist of varved clay in which iron-manganese nodules occur (Schoettle and Friedman, 1973).

Seasonal Variation of Monthly Precipitation Together With Maximum and Minimum Conditions on Drainage Basin - See Tables 6a and 6b (Colon, 1972).

Table 6a. AVERAGE MONTHLY PRECIPITATION FOR THE SOUTH BASIN (STATION 1), LAKE GEORGE, N.Y.+

Precipitation (Inches)

Month	1969	1970	1971
January	0.083 (0.820)	0.021* (0.210)	0.052* (0.480)
February	0.041* (0.360)	0.082 (0.830)	0.158** (1.170)
March	0.050 (0.870)	0.078 (0.650)	0.127 (1.260)
April	0.136 (1.490)	0.058 (1.090)	0.087 (1.200)
May	0.137 (1.020)	0.074 (0.950)	0.070 (0.620)
June	0.104 (1.060)	0.065 (0.810)	0.053 (0.940)
July	0.112 (0.920)	0.095 (0.800)	-1.0
August	0.078 (0.380)	0.067 (0.930)	-1.0
September	0.052 (0.600)	0.138** (1.130)	-1.0
October	0.042 (0.490)	0.092 (0.800)	-1.0
November	0.175** (1.070)	0.106 (0.940)	-1.0
December	0.081 (0.890)	0.115 (0.840)	-1.0

+ **The maximum precipitation value (inches) for each month is seen in parenthesis. Missing data are shown as -1.0. Annual minimum and maximum precipitation values are designated by an asterisk (*) and double asterisk (**) respectively.**

Table 6b. AVERAGE MONTHLY PRECIPITATION FOR THE NORTH BASIN (STATION 6), LAKE GEORGE, N.Y.+

Precipitation (Inches)

Month	1969	1970	1971
January	-1.0	0.013* (0.120)	0.050* (0.500)
February	-1.0	0.089 (0.930)	0.161** (1.180)
March	-1.0	0.070 (0.600)	0.098 (0.710)
April	-1.0	0.117 (1.770)	0.055 (0.710)
May	-1.0	0.089 (1.090)	0.065 (0.560)
June	-1.0	0.069 (0.680)	-1.0
July	-1.0	0.089 (1.090)	-1.0
August	-1.0	0.074 (1.240)	-1.0
September	-1.0	0.128** (0.980)	-1.0
October	0.030* (0.400)	0.085 (0.650)	-1.0
November	0.145 (0.860)	0.039 (0.270)	-1.0
December	0.155** (0.860)	0.076 (0.710)	-1.0

+ The maximum precipitation value (inches) for each month is seen in parenthesis. Missing data are shown as -1.0. Annual minimum and maximum precipitation values are designated by an asterisk (*) and double asterisk (**) respectively.

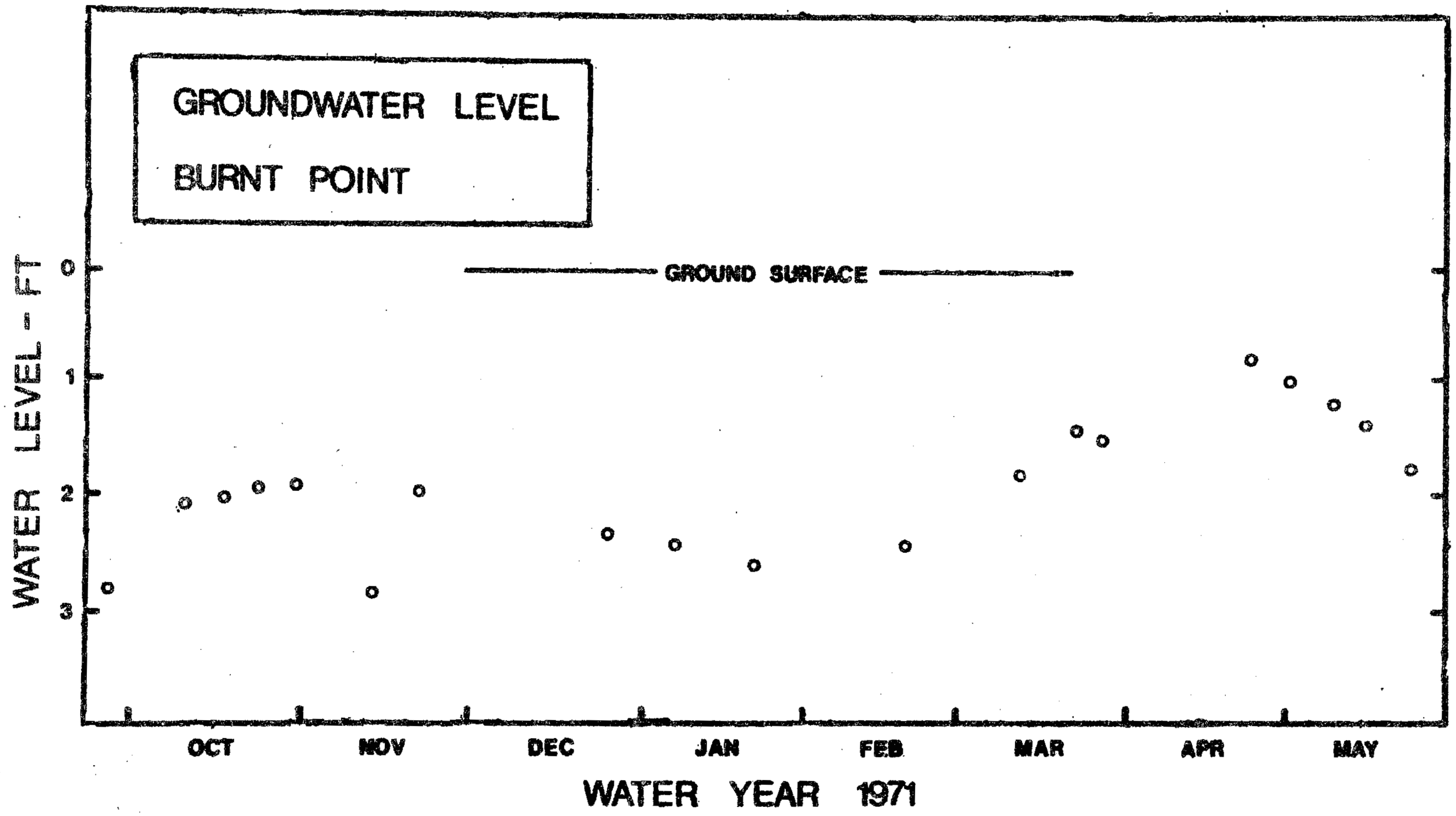
Inflow and Outflow of Water

For the period of October, 1971 through May, 1972, total water input to the lake was 94.6 in., losses were 86.5 in. and a storage of 8.1 in. Groundwater for the 1971 water year is seen in Figure 5 (Colon, 1972). Average outflow from the lake at the north (Ticonderoga) is 8.34 m^3/sec., based on 22 years of record.

Water Currents - These data have not been determined.

Water Renewal Time - Based on the volume and average outflow from the lake, the water retention time in Lake George, NY is 7.98 years.

FIGURE 5



GROUNDWATER LEVEL
BURNT POINT
GROUND SURFACE
WATER LEVEL - FT
0
1
2
3
OCT
NOV
DEC
JAN
FEB
MAR
APR
MAY
WATER YEAR 1971

IV. LIMNOLOGICAL CHARACTERIZATION (Preliminary)

Physical

1. Temperature - See Figures 6, 7, 8, and 9 (Williams and Clesceri, 1972; Colon, 1972).

2. Conductivity - These data available at this time are from April through September, 1971 and ranged from 85-95 u mohs/cm.

3. Light transmittance - Light intensity at the surface was 2,400 ft. candles during March, 1971 and 1972. During August, 1971 and 1972 the surface light intensity approached 6,000 ft. candles. Other data are shown in Tables 7 and 8 (Williams and Clesceri, 1972).

4. Color - Measurements of color of lake water have not yet been determined for Lake George, NY.

5. Solar Radiation - See Figure 10 (Colon, 1972).

Chemical

1. pH - See Table 9.

2. Dissolved oxygen - See Table 10 (Williams & Clesceri, 1972).

3. Total phosphorus including (fraction) forms - See Table 11.

4. Total nitrogen including (fraction) forms - See Tables 12 and 13.

5. Alkalinity - See Table 14.

6. Ca, Mg, Na, K, SO_4, Cl, Fe - See Table 15 for Fe; insufficient data on others. (Williams and Clesceri, 1972).

7. Silica - See Table 16.

Biological

1. Phytoplankton
   a. Chlorophyll - These data are not available.
   b. Primary production - See Figure 12. In addition, data regarding annual production of Nitella flexilis (macroalga) and other macrophytes are given in Table 17 (Stross, 1972).
   c. Identification and count - Tables 18 and 19 (Howard, 1973) and Figure 11 (Williams and Clesceri, 1972).

FIGURE 6

LAKE TEMPERATURE AND ICE COVER

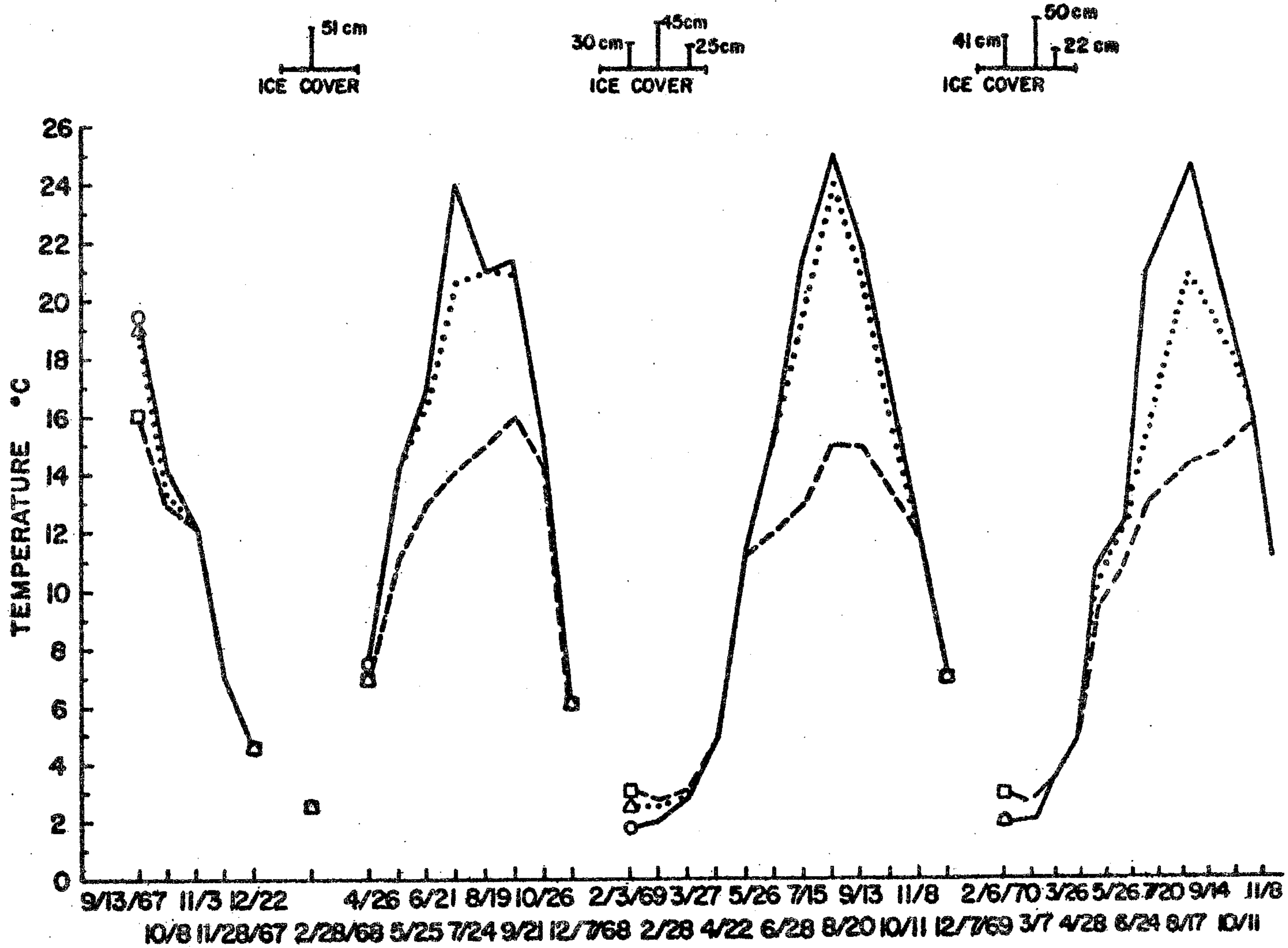


FIGURE 7. SEASONAL CHANGES IN THE LAKE GEORGE THERMOCLINE

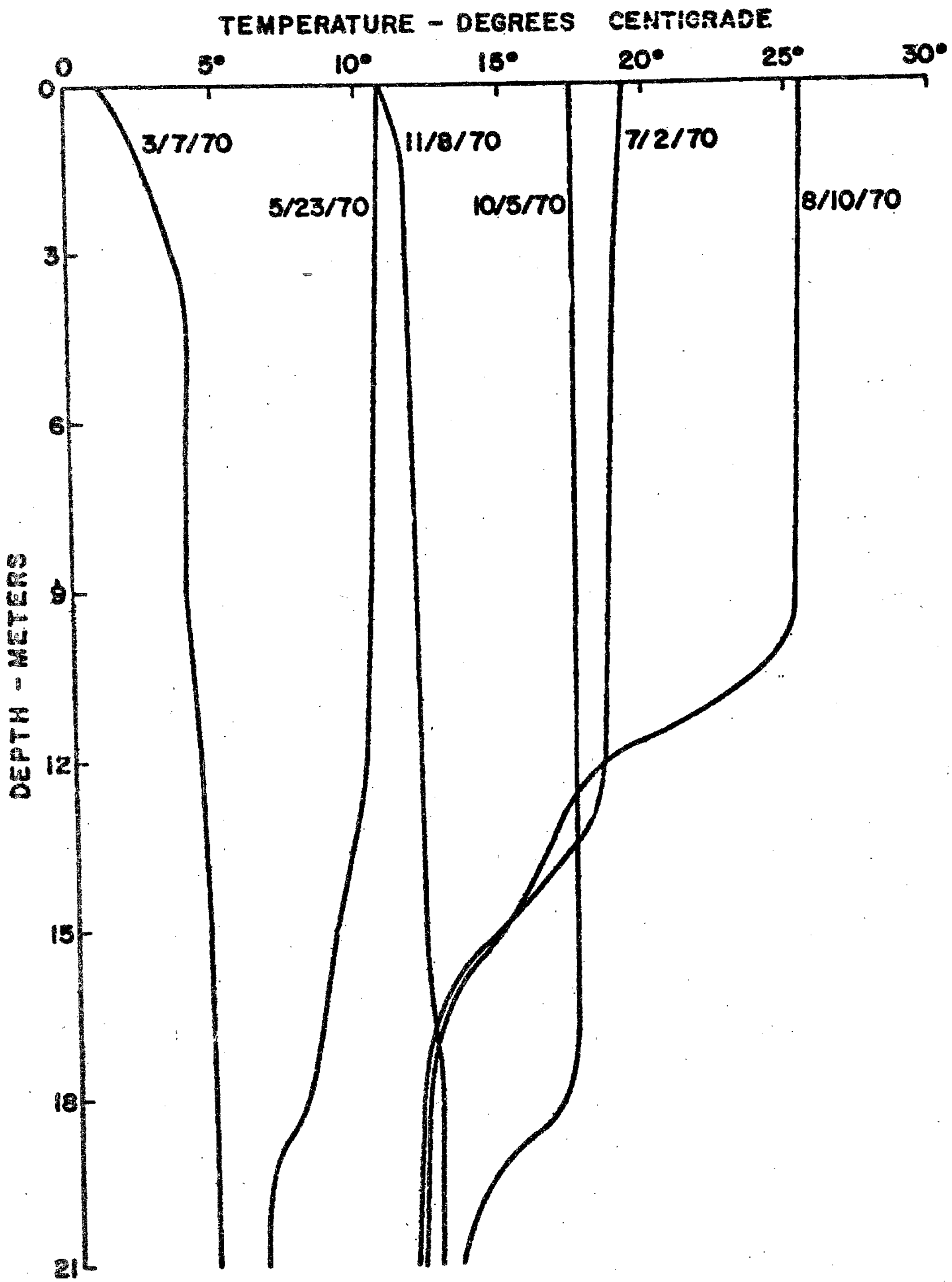


FIGURE 8. Seasonal Variation of Water Temperatures at Diamond Island, Lake George (1971)

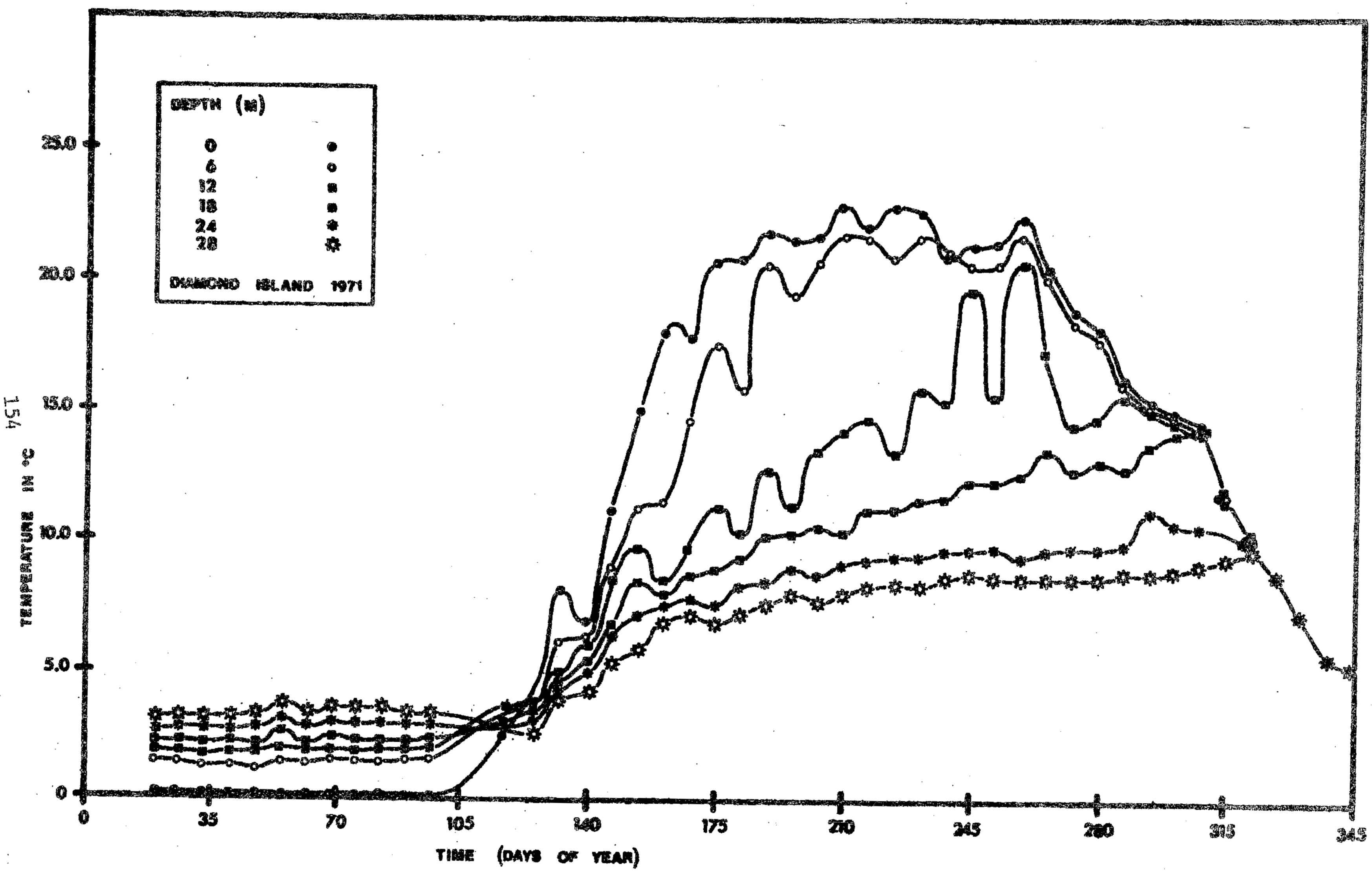


FIGURE 9. Seasonal Variation of Water Temperatures at Diamond Island, Lake George (1972)

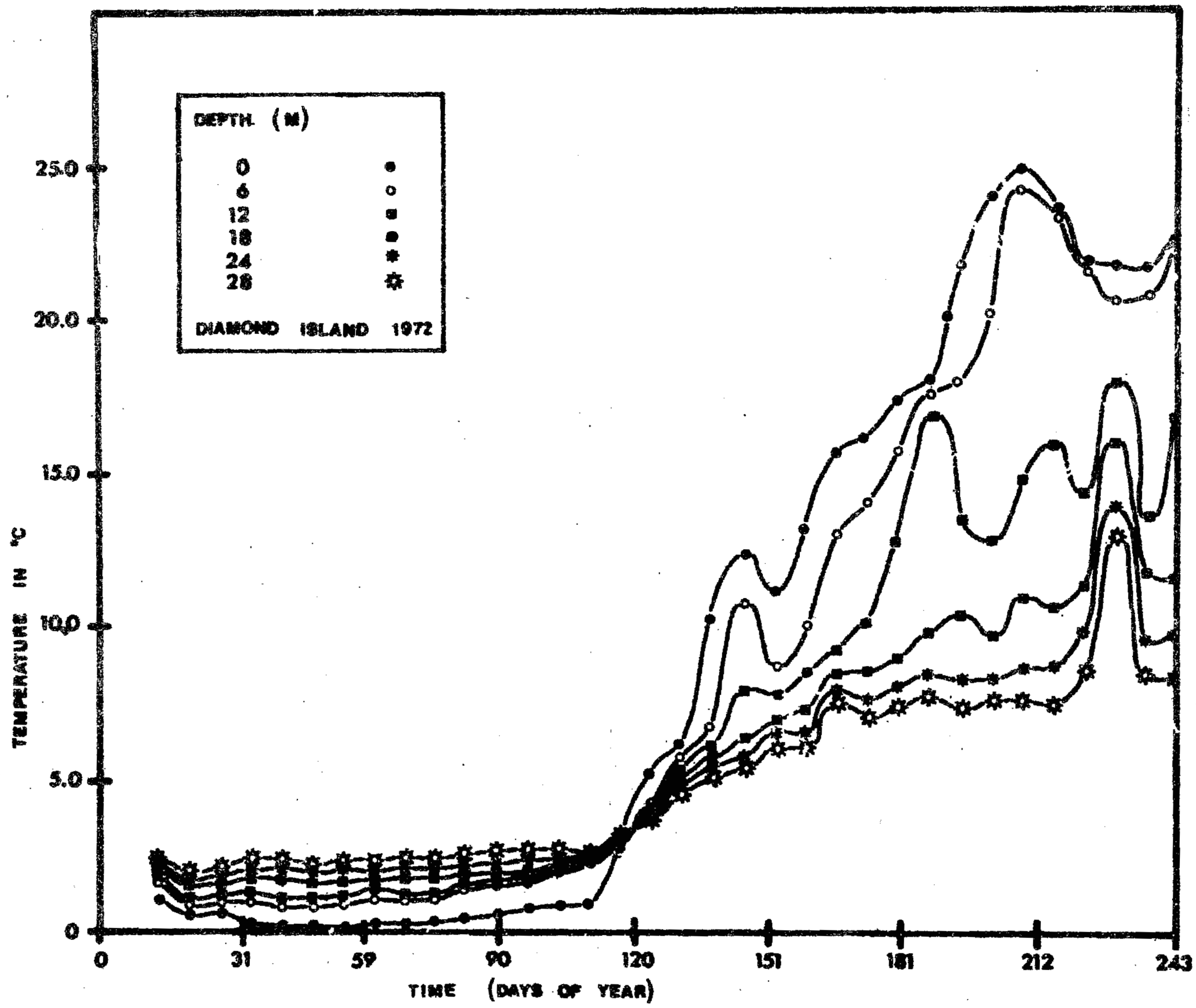


Table 7. SECCHI DISC MEASUREMENTS (METERS)

Date	Station 1	Station 6
3/26/70	7.0	8.5
6/26/70	8.5	10.0
7/17/70	7.0	13.5
8/16/70	7.0	9.0
9/28/70	7.0	9.5
10/05/70	7.0	10.0
10/11/70	6.0	9.0
11.08/70	6.5	10.0

Table 8. RELATIVE UNDERWATER LIGHT INTENSITY WITH DEPTH

(percent)

Depth	9/09/69		8/17/70		9/13/70		3/06/71	
(meters)	Station 1	Station 6	Station 1	Station 6	Station 1	Station 6	Station 1	Station 6
1	85	87	54	75.7	75.7	85.8	28.8	16.7
3	60	60	33.7	50.0	46.0	65.5	7.8	3.1
6	32	32	20.2	27.5	26.3	36.3	2.0	1.3
9	18	19	10.2	18.0	13.8	21.6	0.62	0.67
12	9.2	13	4.0	12.0	7.5	15.4	0.28	0.36
15	4.0	6.6	1.5	4.8	2.8	8.6	0.12	0.18
18	1.8	3.0	0.42	2.4	1.2	5.4		
21	0.82	1.8	0.22	1.0	0.53	2.2		
24	0.42	0.85	0.10		0.27	1.0		

FIGURE 10. Net Solar Radiation at Burnt Point, Lake George (1970)

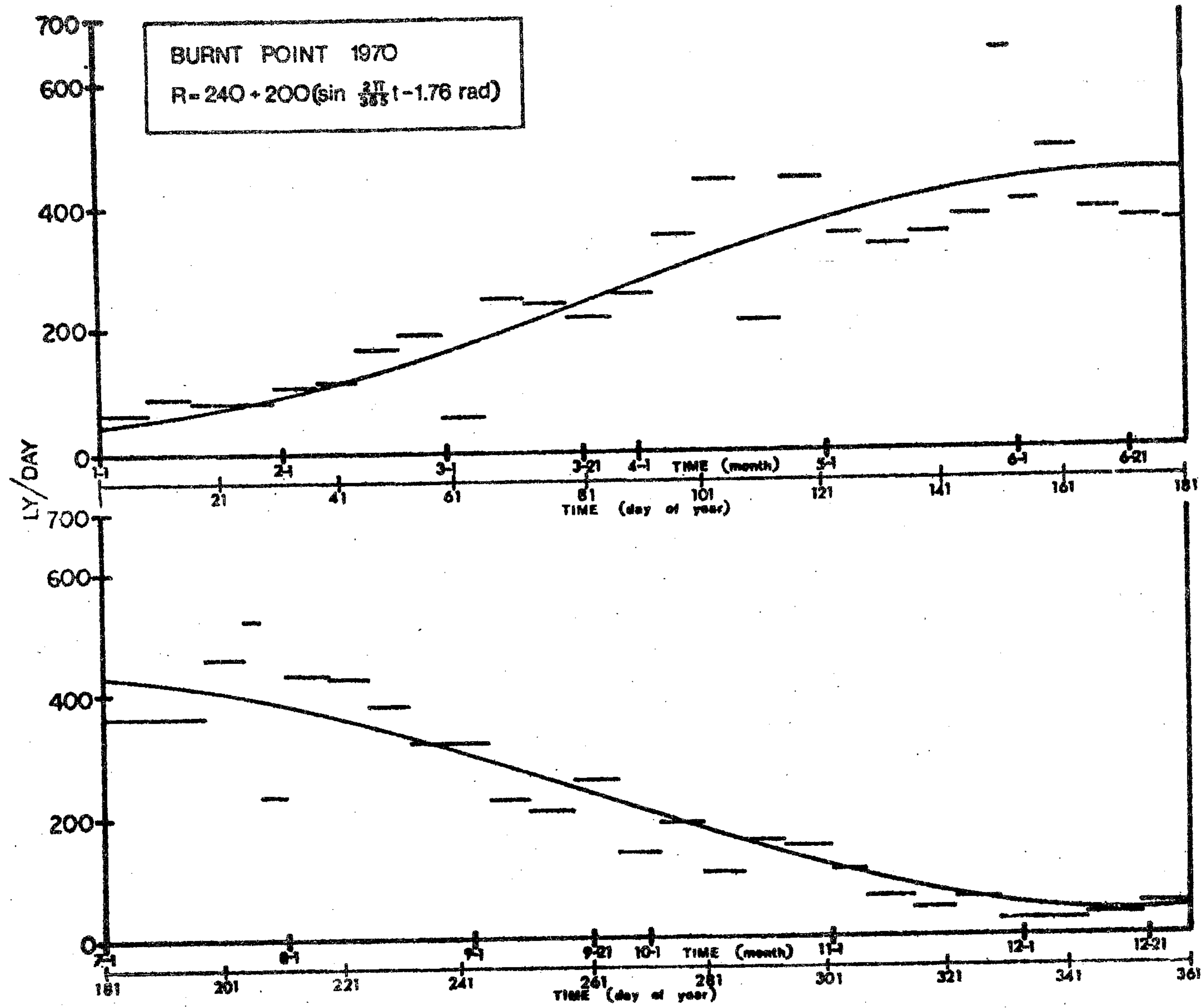


TABLE 9. REPRESENTATIVE pH VALUES FOR LAKE GEORGE

Station 1

Date:	73.065	73.107	73.171	73.199	73.214	73.241	73.255	73.304	73.324
Depth (m)									
0.5	6.97	8.17	7.54	7.76	7.72	7.77	7.50	7.30	7.43
5.0	6.96	8.07	7.60	7.71	7.61	7.78	6.99	7.14	7.41
10.0	7.03	8.13	7.72	7.84	8.09	7.70	7.00	7.50	7.50
15.0	6.95	8.14	7.59	7.38	7.58	7.37	6.95	7.42	7.40
20.0		8.19	7.56	7.15	7.55	7.22	6.82	7.16	7.46
21.0	6.92								
23.0								6.82	
23.5		8.19							
25.0			7.12						

TABLE 10. TEMPERATURE AND DISSOLVED OXYGEN (STATION 1)

Date	9/20/68		10/25/68		9/13/69		10/12/69		3/7/70		9/13/70		10/11/70		3/26/71	
Depth	Temp. °C	DO mg/l	Temp. °C	DO mg/l	Temp. °C	DO mg/l	Temp. °C	DO mg/l	Temp. °C	DO mg/l	Temp. °C	DO mg/l	Temp. °C	DO mg/l	Temp. °C	DO mg/l
3	21.1	7.8	17.4	7.2	21	10.4	16.5	9.3	3.5	12.9	20.3	8.6	16.7	8.9	1.9	11.5
6	21.0	7.8	17.4	7.3	21	10.3	16.5	9.2	3.8	12.7	20.3	8.4	16.7	8.9	2.0	11.4
9	20.1	7.7	17.3	7.2	20.8	9.2	16.2	9.0	3.8	12.7	18.1	7.7	16.6	8.5	2.0	11.4
12	19.2	7.6	17.1	7.1	19	8.4	15.5	8.6	4.2	12.3	16.1	7.1	16.6	8.4	2.0	11.2
15	15.9	7.2	16.9	7.0	16	8.0	14.0	5.6	4.5	11.8	14.7	6.6	16.3	7.7	2.3	11.0
18	14.0	6.5	14.4	3.9	14.2	8.0	13.7	4.2	4.5	11.8	14.1	6.4	11.7	4.9	2.6	10.8
21	13.1	8.0	13.9	4.0	13	7.5	13.5	2.0	4.5	11.7	13.2	5.3	11.6	4.0	3.0	10.4
24	-	-	-	-	-	-	-	-	4.7	11.5	11.8	4.5	11.5	2.2	-	-

TABLE 11

TOTAL PHOSPHORUS CONCENTRATIONS IN LAKE GEORGE

All Figures as mgP/1

Station 1

Date:	73.065	73.107	73.171	73.199	73.219	73.241	73.255	73.269
Depth (m)								
0.5	0.007	0.008	0.013	0.005	0.004	0.007	0.004	0.008
5.0	0.006	0.014	0.006	0.009	0.006	0.007	0.006	0.007
10.0	0.008		0.007		0.008	0.005	0.007	0.007
15.0	0.005	0.008	0.008	0.024			0.007	0.010
20.0		0.007	0.008	0.016	0.005	0.007	0.007	0.009
21.0	0.006							
23.5		0.008						
25.0			0.008					

SOLUBLE REACTIVE PHOSPHORUS CONCENTRATIONS IN LAKE GEORGE

0.5	0.0007	0.0014	0.0005	0.0010	0.0011	0.0	0.0027	0.0003
5.0	0.0007	0.0011	0.0006	0.0007	0.0020		0.0020	0.0002
10.0	0.0007	0.0012	0.0013	0.0013	0.0012		0.0024	0.0003
15.0	0.0009	0.0008	0.0008	0.0023	0.0002		0.0021	0.0006
20.0		0.0007	0.0014	0.0021	0.0004		0.0023	0.0003
21.0	0.0009							
23.5		0.0008						
25.0			0.0011					

TABLE 12
TOTAL KJELDAHL N (mg N/l)

Station 1

Date	Depth (m) 0.5	5.0	10.0	15.0	20.0	23.0
73.065	0.218	0.237	0.187	0.157	0.198(21m)	
73.186	0.212	0.220	0.208	0.179	0.130	
73.199	0.176	0.260	0.269	0.186	0.202	
73.213		0.302		0.236	0.167	
73.241	0.230	0.146	0.132	0.168	0.189	
73.255	0.324	0.312	0.248	0.309	0.240	
73.269	0.195	0.277	0.235	0.216	0.202	
73.304	0.267	0.230	0.266	0.245	0.221	0.203
73.324	0.178	0.205	0.202	0.247	0.185	

Station 6

Date	Depth (m) 0.5	5.0	10.0	15.0	20.0	25.0
73.186	0.101	0.062	0.086	0.138	0.127	
73.199	0.193	0.212	0.196	0.269	0.343	
73.241	0.138	0.170	0.103	0.107	0.137	0.102
73.255	0.251	0.246	0.249	0.100		
73.269	0.199	0.191	0.191	0.190	0.207	0.208
						0.200(30)
						0.206(35)
73.324	0.173	0.186	0.175	0.177	0.197	

Station 1: Range: 0.130-0.314 Mean: 0.219 ± 0.046
Station 6: Range: 0.062-0.343 Mean: 0.198 ± 0.141

TABLE 13

NITRATE CONCENTRATIONS IN LAKE GEORGE

All Figures as mgN/1

Station 1								
Date:	73.065	73.107	73.199	73.214	73.241	73.255	73.304	73.324
Depth (m)								
0.5	0.170	0.085	0.011	0.025	0.006	0.011	0.011	0.014
5.0	0.066	0.058	0.008	0.012	0.009	0.008	0.006	0.013
10.0	0.068	0.058	0.016	0.006	0.016	0.009	0.008	0.014
15.0	0.076	0.043	0.020	0.009	0.009	0.010	0.006	0.014
20.0		0.043	0.014	0.022	0.014	0.011	0.014	0.015
20.0	0.077							
23.0							0.039	
23.5		0.043						
25.0				0.016				

AMMONIA CONCENTRATIONS IN LAKE GEORGE

0.5	0.032	0.018	0.014		0.033	0.026	0.007	0.008
5.0	0.024	0.012	0.029		0.023	0.029	0.010	0.005
10.0	0.026	0.012	0.015		0.032	0.024	0.006	0.008
15.0	0.038	0.010	0.007		0.021	0.025	0.011	0.015
20.0		0.011	0.031		0.031	0.032	0.008	0.010
21.0	0.028							
23.0							0.009	
23.5		0.012						

Table 14. ALKALINITY (mg $CaCO_3$/l) FOR LAKE GEORGE, N. Y.

Date	Average Alkalinity (mg $CaCO_3$/l) North Basin	South Basin
July, 1972	23.0	22.4
August, 1972	16.95	16.7
September, 1972	17.5	16.5
October, 1972	22.6	22.1
November, 1972	23.4	23.6
March, 1973		22.8
April, 1973		22.6
May, 1973		
June, 1973	22.0	23.2
July, 1973	22.5	22.6
August, 1973	21.7	21.9
September, 1973	21.5	21.4
October, 1973	21.6	
November, 1973	20.8	21.3
December, 1973		
January, 1974	24.1	

Table 15. MEAN SEASONAL CONCENTRATIONS OF FE, MN, CU AND ZN IN THE NORTH AND SOUTH BASINS OF LAKE GEORGE, N.Y.

		South Basin (ug/l)				North Basin (ug/l)			
Season	Depth(m)	Fe	Mn	Cu	Zn	Fe	Mn	Cu	Zn
Winter (Jan. 1-Mar.31)	3	27.2	2.0	5.2	43.4	35.2	1.9	2.7	51.1
	9	42.1	2.1	3.5	49.3	34.8	1.3	2.0	79.6
	15	30.6	1.6	3.7	44.4	50.7	2.3	2.2	76.6
Spring (Apr.1-June 21)	3	25.1	3.2	3.9	32.7	41.5	2.9	2.6	33.5
	9	17.3	2.5	4.2	28.0	26.2	2.5	3.5	53.2
	15	16.9	4.0	3.8	30.4	35.4	3.2	3.2	38.6
Summer (June 21-Sept.21)	3	29.0	2.6	3.4	46.4	29.8	2.0	3.0	74.9
	9	23.5	2.2	3.1	31.8	23.8	3.3	3.2	40.4
	15	28.8	4.1	2.9	34.2	23.6	1.9	2.9	23.9
Fall (Sept.21-Dec. 7)	3	46.1	1.8	3.1	25.1	13.8	1.4	1.6	71.1
	9	39.9	1.7	2.5	23.3	20.5	1.2	1.7	88.3
	15	30.3	2.5	2.6	43.5	14.5	1.1	2.0	74.5

TABLE 16

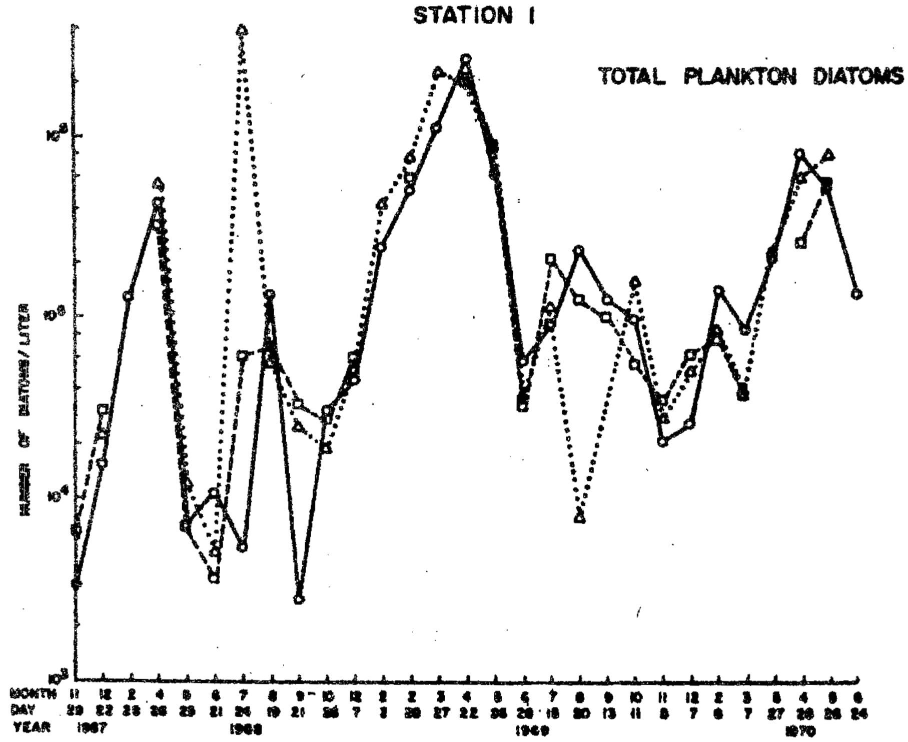
SOLUBLE REACTIVE SILICA IN LAKE GEORGE
(in mg Si/l)

Station 1

Date:	72.074	72.131	72.152	72.173	72.194	72.215	72.236	72.257	72.278	72.313
Depth (m)										
0.5	0.925	1.270	1.060	0.895	0.910	0.820	0.520	0.495	0.520	
3.0	0.990									
5.0		1.270	1.035	0.925	0.875	0.830	0.530	0.500	0.515	0.900
9.0	0.995									
10.0		1.240	1.040	1.020	0.925	0.745	0.615	0.500	0.505	
15.0		1.200	1.060	1.020	0.895	0.890	0.850	0.965	0.665	
20.0		1.135	1.125	1.040	0.950	0.980		1.050	0.845	
21.0										
26.0		1.135	1.245	1.040	1.000	0.955				

FIGURE 11

STATION I



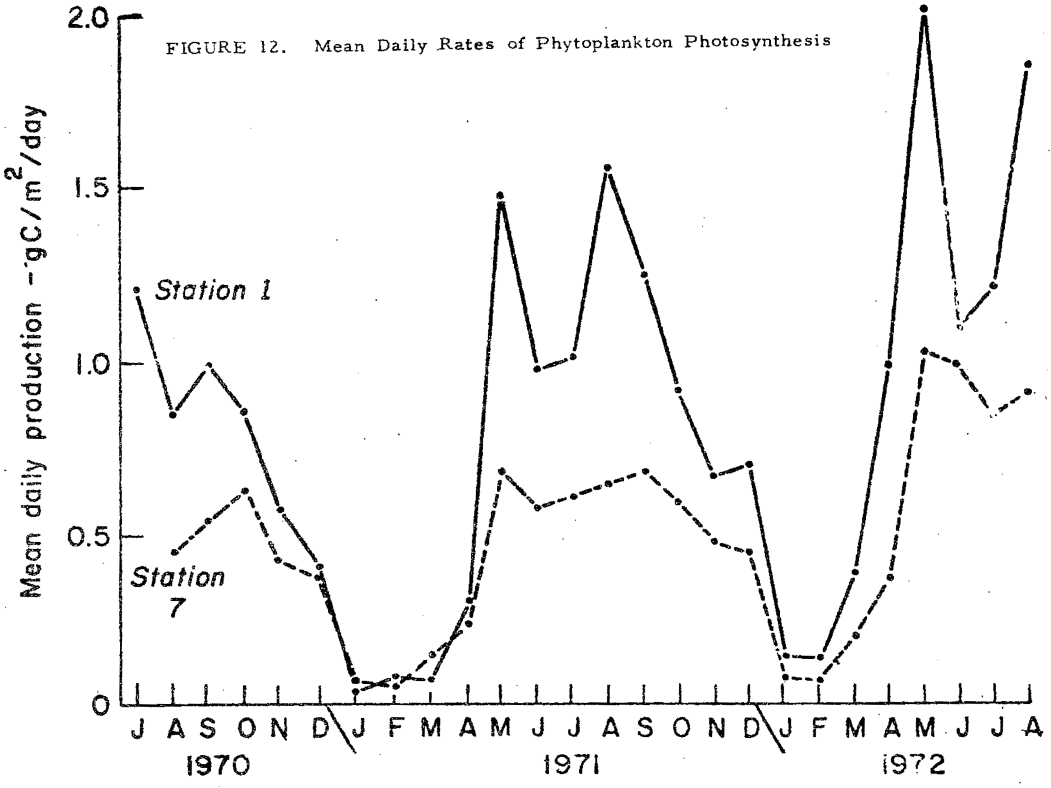


FIGURE 12. Mean Daily Rates of Phytoplankton Photosynthesis

Table 17. ANNUAL PRODUCTION OF <u>Nitella flexilis</u> AND OTHER MACROPHYTES IN THE SOUTH BASIN OF LAKE GEORGE, N. Y. FOR THE YEAR 1972. ALL MEASUREMENTS ARE IN GRAMS (dry wt.)/m$^2 \pm$ STANDARD ERROR (Stross, 1972).

Depth (meters)	Nitella flexilis June	Nitella flexilis Sept.	Other Sepcies Sept.	Annual Production
3.0			97.65 ± 28.0	97.65 ± 28.0
4.0			95.10 ± 31.8	95.10 ± 3.18
5.0			39.38 ± 31.6	39.38 ± 31.6
6.0		43.79 ± 10.1	42.42 ± 15.9	130.00 ± 26.0
7.0	16.41 ± 2.3	53.68 ± 19.8	32.95 ± 18.1	103.04 ± 40.2
8.0	30.42 ± 7.9	73.48 ± 31.5	3.67 ± 3.5	107.57 ± 42.9
9.0	44.06 ± 5.1	76.93 ± 16.55		120.99 ± 21.6
10.0	57.69 ± 5.0	133.00 ± 39.3		190.69 ± 43.3
11.0	57.83 ± 5.1	53.39 ± 34.4		111.22 ± 39.5
12.0	57.91 ± 5.2	105.96 ± 36.9		163.93 ± 43.1

Table 18. SPECIES FOUND IN LAKE GEORGE PHYTOPLANKTON

Ne: net plankton (maximum dimension greater than 50 u)

Na: nannoplankton (maximum dimension 50 u or less)

1. Eudorina elegans Ehrenberg. (Na)
2. Sphaerocystis Schroeteri (Wolle) W. & G. S. West. (Na)
3. Gloeocystis gigas (Kuetzing) Lagerheim. (Na)
4. Elakatothrix gelatinosa Wille. (Na)
5. Planktosphaeria gelatinosa G. M. Smith. (Na)
6. Oocystis crassa Wittrock. (Na)
7. Oocystis pusilla Hansgirg. (Na)
8. Oocystis submarina Lagerheim. (Na)
9. Oocystis sp. (Na)
10. Botryococcus braunii Kuetzing. (Na)
11. Dimorphococcus lunatus A. Braun. (Na)
12. Ankistrodesmus falcatus (Corda) Ralfs var. acicularis (A. Braun) G. S. West. (Na)
13. Selenastrum minutum (Naeg.) Collins. (Na)
14. Quadrigula closterioides (Bohlin) Printz. (Na)
15. Tetraedron minimum (A. Braun) Hansgirg. (Na)
16. Scenedesmus bijuga (Turp.) Lagerheim. (Na)
17. Crucigenia rectangularis (A. Braun) Gay. (Na)
18. Crucigenia tetrapaedia (Kirch.) W. & G. S. West. (Na)
19. Cosmarium sp. (Na)
20. Cosmarium sp. (Na)
21. Staurastrum furcigerum De Brebisson.
22. Spondylosium planum (Wolle) W. & G. S. West. (Ne)
23. Tribonema sp. (Ne)
24. Ochromonas sp. (Na)
25. Bitrichia chodati (Reverdin) Chodat. (Na)
26. Dinobryon bavaracum Imhof. (Na)
27. Dinobryon cylindricum Imhof. (Na)
28. Dinobryon divergens Imhof. (Na)

Table 18 (Continued). SPECIES FOUND IN LAKE GEORGE PHYTOPLANKTON

29. Epipyxis sp. (Na)
30. Mallomonas sp. (Na)
31. Mallomonas sp. (Na)
32. Melosira sp. (Ne)
33. Cyclotella comta (Ehren.) Kuetzing. (Na)
34. Cyclotella stelligera Clet & Grunow. (Na)
35. Stephanodiscus astrea (Ehren.) Grunow. (Na)
36. Tabellaria fenestrata (Lyngb.) Kuetzing. (Na)
37. Meridion circulare (Grev.) Agardh. (Na)
38. Fragilaria crotonensis Kitton. (Ne)
39. Asterionella formosa Rassall. (Ne)
40. Synedra sp. (Ne)
41. Gymnodinium sp. (Ne)
42. Glenodinium pulvisculus (Ehren.) Stein. (Na)
43. Peridinium cinctum (Muell.) Ehrenberg. (Ne)
44. Cryptomonas sp. (Na)
45. Chroococcus dispersus (Keissl.) Lemmermann. (Na)
46. Chroococcus limneticus Lemmermann. (Na)
47. Gloeocapsa punctata Naegeli. (Na)
48. Aphanocapsa elachista West and West. (Na)
49. Microcystis incerta Lemmermann. (Na)
50. Gloethece linearis Naegeli var. composita G. M. Smith. (Na)
51. Aphanothece clathrata G. S. West. (Na)
52. Aphanothece nidulans P. Richter. (Na)
53. Coelosphaerium Naegelianum Unger. (Na)
54. Gomphosphaeria aponina Kuetzing. (Na)
55. Lyngbya limnetica Lemmermann. (Ne)
56. Anabaena sp. (Na)
57. - 64. Unknown coccoid cells and flagellates. All (Na)

TABLE 19

PHYTOPLANKTON BIOMASS IN LAKE GEORGE

All data collected at Station 1 and reported as micrograms per liter*.

DATE DEPTH (m)	72.257	.307	73.065	.107	.186	.216	.241
0.5	144.	11.0	10.0	511.	325.	189.	79.0
2.0			51.0	758.	129.	461.	72.0
5.0	188.	11.0	124.	558.	261.	260.	131.
10.0	631.	5.60	80.0	531.	190.		126.
15.0	271.		103.	721.	188.	260.	106.

* Assuming a density of 1 gm/cm^3

Data from Howard (1973)

2. Zooplankton (McNaught, et al., 1972)

a. Identification and count -

Species	Number/m^3/day
Diaptomus sicilis	961
Diaptomus minutus	2554
Cyclops bicuspidetus	3737
Daphnia galeata	714
Daphnia longiremus	212
Bosmina spp.	358

3. Bottom Fauna - See Table 20 (Perrotte, 1974).

4. Fish - The data shown in Table 21 are from 1973 surveys of the littoral region (15 sites) of Lake George. There are no census figures, etc. for the fish populations of the entire lake. Of major importance to this trophic level and yet not included herein due to a lack of reliable figures at this time are the Cisco and Lake trout populations for this body of water (George, et al., 1974).

5. Bacteria - The organisms listed are the most abundant bacteria observed in Lake George, NY:

   Achromobacter spp.
   Aeromonas liquefaciens
   Aeromonas spp.
   Arthrobacter spp.
   Brevibacterium haelis
   Brevibacterium sp.
   Cellulomonas sp.
   Kurthia sp.
   Proteus sp.
   Pseudomonas cohaerens
   Pseudomonas spp.

6. Bottom flora - These data have not yet been determined.

7. Macrophytes - See Table 22 (Boylen and Sheldon, 1973).

V. NUTRIENT BUDGETS - See Table 23.

Table 20. THE DENSITY, LENGTHS, DRY WEIGHTS AND WEIGHTS/mm OF LENGTH FOR THE DOMINANT BENTHIC MACROINVERTEBRATES AT TEA ISLAND AND SMITH BAY, LAKE GEORGE, NEW YORK

TEA ISLAND

Dates: July 19, 1973, (October 27, 1973)
Depth: 7M

Taxa	Number/ft.^{2}	Total Length (mm)	Dry Wt. (g)	wt/mm of length $\times 10^{-5}$
Chironomus spp.	25 (139)*	144.48 (1056.5)	0.0054 (.0229)	3.73 (2.11)
Gammarus fasciatus	21 (33)	72.7 (74.5)	0.0170 (.0173)	
Hyalella azteca	39 (3)	125.6 (6.0)	0.0160 (.0007)	23.3
Snails (*Viviparus* sp.)**	7 (3)		0.6602 (0.2899)	12.7
Snail Shells	7 (3)	118.4 (52.0)	2.6819 (1.1778)	226.5
Asellus communis	10	39.1	0.0105	26.8
Others			0.8678 (.5160)	
Total Biomass/ft.^{2}			4.2588 (1.7347)	

SMITH BAY

Date: July 24, 1973
Depth: 7M

Taxa	Number/ft.^{2}	Total Length (mm)	Dry Wt. (g)	Wt/mm of length $\times 10^{-5}$
Chironomus spp.	67 (124)	353.9 (1011.6)	0.0050 (.0234)	1.41 (2.3)
Gammarus fasciatus	20 (15)	80.0 (32.0)	0.0226 (.0090)	28.2
Hyalella azteca	39 (4)	72.5 (7.0)	0.0084 (.0008)	11.6
Snails (*Viviparus* sp.)	66 (9)	181.86 (39.0)	0.0316 (.1460)	17.4 (37.4)
Asellus communis	3 (5)	15.2 (26.3)	0.0044 (.006)	28.9 (22.8)
Others			0.0463 (.0406)	
Total Biomass/ft.^{2}			0.1183 (.2168)	

*Number in brackets are from October 27, 1973 samples.
**With shells

Table 21. FISH POPULATIONS IN THE LITTORAL REGION OF LAKE GEORGE, N.Y.

Species	Total Number (N) 15 Sites	Approx. Total (entire shoreline*)	Approx. Biomass (kg)	% Total
Red Breast (*Lepomis auritus*)	1,149	48,600	2,576	34.9
Rock Bass (*Ambloplites rupestris*)	905	38,300	2,480	27.5
Pumpkinseed (*Lepomis gibbosus*)	707	29,900	1,582	21.5
Smallmouth Bass (*Micropterus dolomieui*)	400	16,900	1,815	12.1
Yellow Perch (*Perca flavescens*)	85	3,600	313	2.6
Largemouth Bass (*Micropterus salmoides*)	30	1,300		0.9
Northern Pike (*Esox lucius*)	11	500		0.3
Other	7	300		0.2
Total Fish	3,294	147,400		
Total Length of Runs (m)	4,952 m			
Total Area (m^2)	47,004 m^2			
N/1,000 m	9,897			
N/10,000 m^2 (ha)	10,389			

* Using 209.6 km as shoreline length and knowing number of fish species/km.

Table 22. MOST COMMON MACROPHYTE SPECIES FOUND IN THE LITTORAL ZONE OF LAKE GEORGE, N.Y.*

Species	Average Dry Weight of mature plant+	Average Maximum Height of mature plant	Depth of maximum abundance
Bidens beckii	.483 g	56.3 cm	2-7 m
Chara globularis	.075 g	12 cm	1 m
Elatine minima			1 m
Elodea canadensis	.540 g	60 cm	1-9 m
Eriocaulon septangulare	.237 g	2.8 cm	1 m
Heteranthera dubia	.947 g	84 cm	1-3 m
Isoetes echinospora			1-3 m
Isoetes macrospora			3-8 m
Juncus sp.			1 m
Lobelia dortmanna			1 m
Myriophyllum alterniflorum	.268 g	51.3 cm	1-3 m
Myriophyllum tenellum			1 m
Najas flexilis	080 g	24 cm	1-7 m
Nitella flexilis			9 m
Potamogeton amplifolius	2.677 g	75.7 cm	3 m
Potamogeton gramineus	.307 g	84 cm	1-5 m
Potamogeton perfoliatus	.284 g	74.5 cm	1-5 m
Potamogeton praelongus	.836 g	73.3 cm	5 m
Potamogeton pusillus	.081 g	29.3 cm	2-5 m
Potamogeton robbinsii	.873 g	69.7 cm	7 m
Ranunculus longirostris	.154 g	46 cm	1-3 m
Sagittaria sp.	.394 g	11 cm	1 m
Utricularia resupinata			1 m
Vallisneria americana	.536 g	77.7 cm	1-5 m
Subularia sp.	.014 g	6.2 cm	1 m

* All species collected from 1 m depth or greater. All were submergent.

+ Plants were collected on 8/30/73. Visual observation suggests that plants collected were smaller than mature plants found earlier in the summer.

TABLE 23

Estimated Phosphorus and Nitrogen Budget
Based on Normal Precipitation of Basin

	Phosphorus		Nitrogen	
Sources	kg	% of Total Sources	kg	% of Total Sources
Runoff	2890	37.1	86,700	43.1
Precipitation	2400	30.8	84,600	42.1
Sewage Treatment Plant Effluents	0	0	18,000	9.1
Septic tank Effluents	2300	29.5	9,580	4.8
Lawn Fertilizer	208	2.6	2,080	1.0
Total	7800	100	201,000	10.0
Sinks		% of Total Sinks		% of Total Sinks
Outflow at Ticonderoga	2040	26.2	62,800	31.2
Sedimentation	5760	73.8	138,000	68.8
Retention		73.8		68.8
Surface loading	0.0684 $g/m^2/yr$		1.76 $g/m^2/yr$	

(From Gibble, 1974)

VI. DISCUSSION

The geologic history of Lake George appears to be the primary element in the present trophic status of the lake. Lying essentially in a long narrow channel bordered by heights reaching in excess of 600 meters above the lake surface, the ratio of drainage basin surface area to lake surface area is only 4.3. The bedrock is precambrian metamorphic, plutonic and igneous and lies close to the surface with numerous outcroppings in the basin. Thus, only a thin soil cover overlies much of the basin. Precipitation is the only form of hydrologic import, and the basin represents a headwater for the Lake Champlain catchment area.

If one can assume that 15% of the Lake George basin is represented by cleared lands, regardless of purpose, then the export of phosphorus, calculated from runoff loadings (Gibble, 1974) would be 6.9 $mg/m^2/yr$. This figure lies within the range of estimates presented by Dillon and Kirchner (1975) for forested land overlying igneous rock. The latter category corresponds to Vollenweider's (1968) classification of "oligotrophic" soils. Apparently, phosphorus exports in the Lake George watershed are typical for this type of soil-vegetation cover.

The small basin to lake area ratio emphasizes the importance of precipitation directly upon the lake surface as a source of N and P loadings to Lake George. Combined with runoff, these two sources account for 68 and 85% of the phosphorus and nitrogen loadings, respectively. Anthropogenic phosphorus sources are already reduced through application of treated sewage effluent (from the Lake George Village area) onto sand beds, and adsorption onto soils in the numerous septic tank tile fields. There are no known sources of untreated sewage into the lake.

Having recognized the need for a relatively simple approach to the classification of the productivity or trophic state of lakes, Vollenweider (1968) and Vollenweider and Dillon (1974) have concentrated their attention upon phosphorus as the limiting element. However, recognizing that as a limiting element its concentration in the water column would simply represent a "residual," they have focused on the importation, or phosphorus loading, as the proper relationship to productivity. Internal loading or recycling must also be considered, especially in small lakes, but external loading is more important in the larger lakes (Vollenweider, 1968).

The lake volume to phosphorus loading relationship was originally taking into account through the mean depth of the lake. However, recognizing that retention time was equally significant, Vollenweider and Dillon (1974) regressed phosphorus loading against an areal water loading, expressed as mean depth divided by mean residence time. The new relationship provides a significantly better fit for lakes in which mean detention times are within very long, e.g., Lake Tahoe, or very short as is the case of some Canadian Shield lakes.

Referring to the nutrient budget for Lake George (Gibble, 1974), the estimated phosphorus loading is 0.0684 gm/m^2/yr. (See Table 23). With a mean depth of 18 meters and a mean retention time of 8 years, Lake George can be classified as "oligotrophic" on this basis.

Aulenbach and Clesceri (1973) have emphasized the fact, however, that Lake George consists of two distinct basins, south and north. The lake surface area and drainage basin area are 57.6 km^2 and 313.2 km^2 for the south basin and 56.4 km^2 and 178.8 km^2 for the north basin. The year-round population in the south basin is approximately four times that of the north, but during the summer season, this figure increases to approximately six times. Additionally, the south basin contains the two sewage treatment plants located within the total watershed.

Using proportional estimates, the phosphorus loading to the south basin would be 0.0908 gm/m^2/yr. With a mean depth of 15.5 meters and assuming the same mean retention time of 8 years, the south basin would still lie within the "oligotrophic" classification. The similarity of the phosphorus loading to the south basin and to the total lake, once again points to the importance of direct precipitation to the lake surface as a nutrient source.

The correctness of the loading approach to determine productivity, at least as it applies to Lake George, is borne out by the relative success of the process model CLEANX which describes the pelagic epilimnetic zone (Scavia, 1975; Bloomfield, et al., 1973). The compartments represented are the net- and nannophytoplankton, herbivorous and omnivorous zooplankton, non-piscivorous and piscivorous fish, particulate and dissolved organic matter, dissolved inorganic nutrients and decomposers. The driving functions are the phosphorus, nitrogen and carbon inputs from streams, as well as water temperature, incident solar radiation and the level of benthic insect biomass.

The basic processes are obvious but also factual. High spring nutrient loadings, abetted by winter thaws, in the presence of rising temperature and solar radiation levels, result in a pulse of phytoplankton biomass, principally the net plankton, Asterionella formosa. Available dissolved nutrients are further increased by decomposer activity upon organic matter in the runoff. Mean daily production rises to 1.5 gm C/m^2/dy or higher (Figure 12). Zooplankton predation follows with Cyclops bicuspidatus, as a principal species. Cropping by the non-piscivores reduces pressure upon the phytoplankton, but in the presence of lower summer concentrations of nutrients, the nannoplankton become dominant (in terms of biomass, Cyclotella compta becomes the principal species). A biomass and phytoplankton production pulse again occurs in the August-September period. This pulse precedes turnover in Lake George, and is therefore probably unrelated to nutrient increases from the hypolimnion.

In Figures 13, 14 and 15 observed levels of biomass are compared with those simulated by the model. The reasonable fit of the simulation indicates that the modeling of the ecologic processes is sound, and that nutrient inputs from streams with subsequent internal recycling are the principal non-physical driving forces in the Lake George ecosystem.

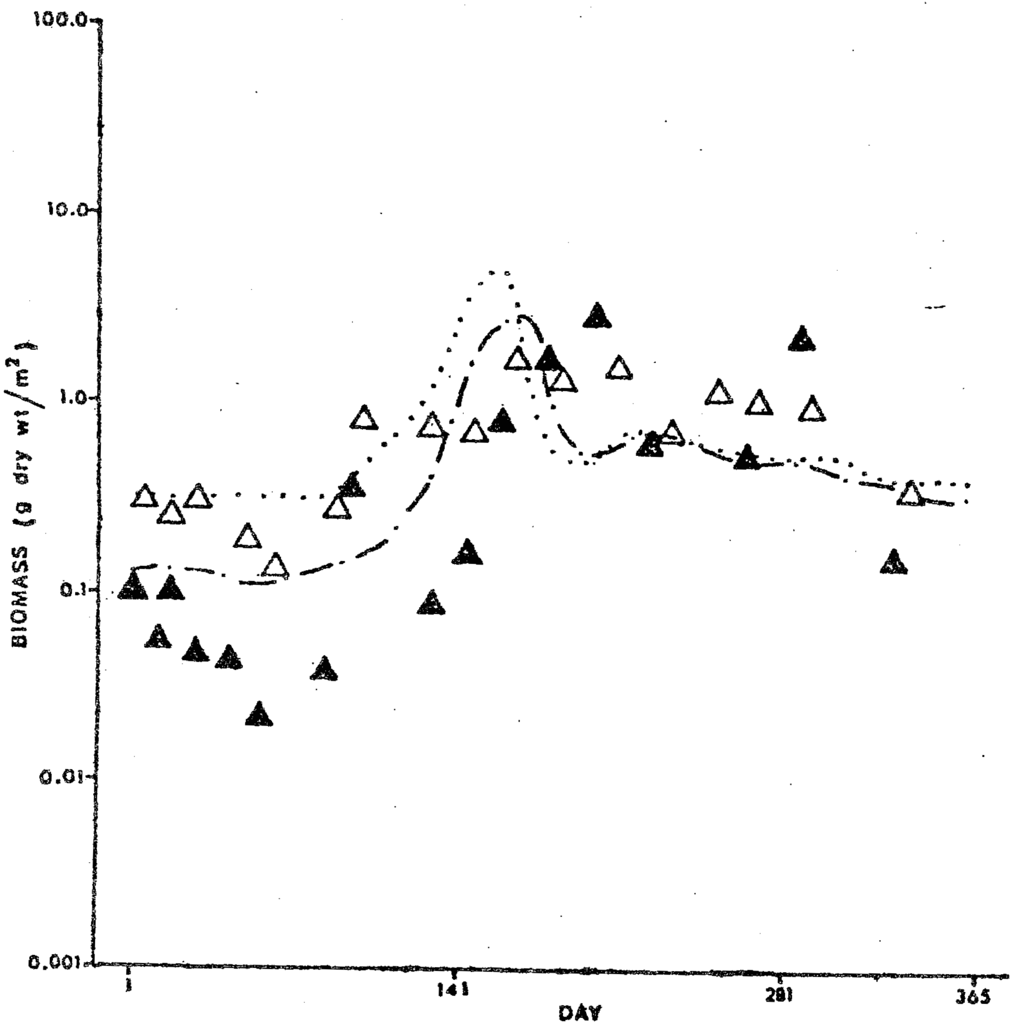


FIGURE 13. Predicted and Observed Biomass Levels of Cladocerans and Copepods in Lake George. Observed Values are from McNaught, et al. (1972) △ = copepods, ▲ = cladocerans

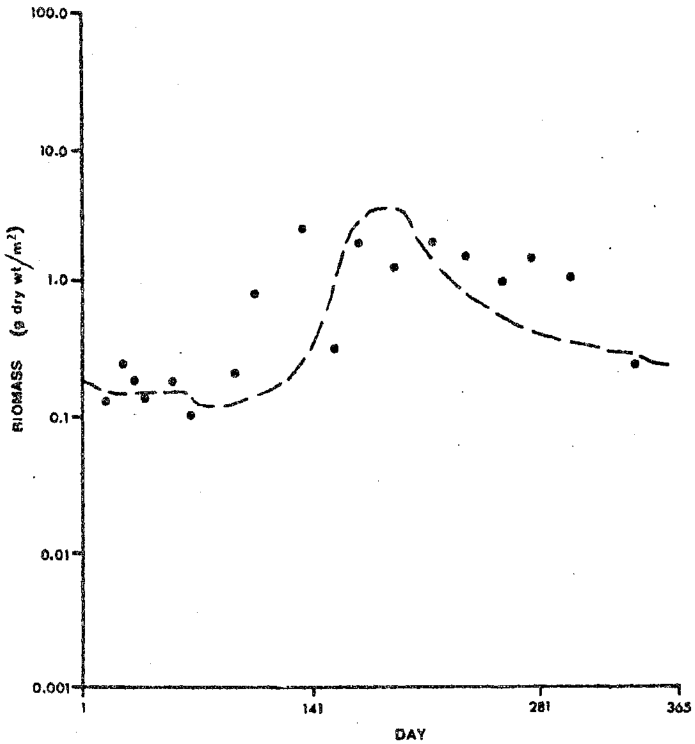


FIGURE 14. Predicted and Observed Biomass Levels of Omnivorous Zooplankton in Lake George. Observed Values from McNaught, et al. (1972)

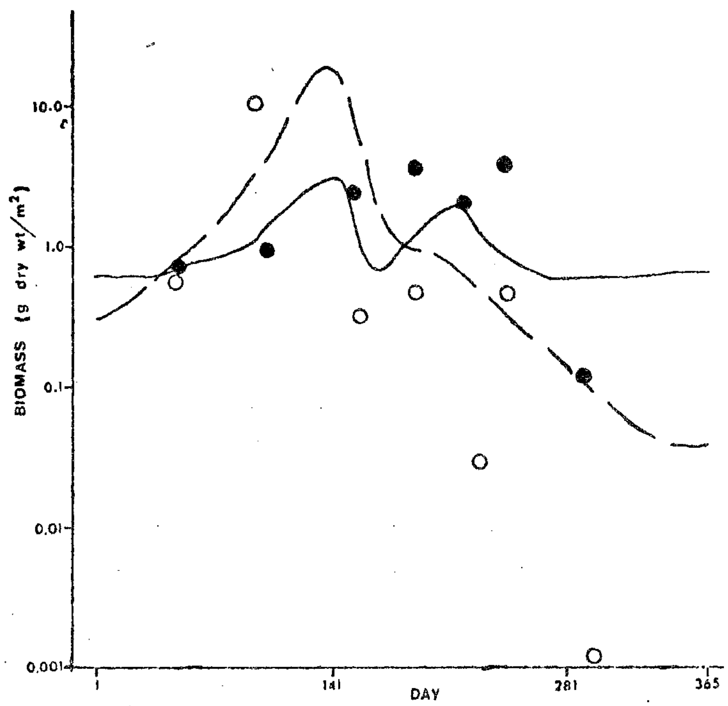


FIGURE 15. Predicted and Observed Biomass Levels of Nanno- and Net Phytoplankton in Lake George. Observed Values from Howard (1973). (Key is same as Figure 17: O=net plantkton, O=nannoplankton.)

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THE LIMNOLOGY OF CAYUGA LAKE, NEW YORK

- A SUMMARY -

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The Indians who lived in villages around the lake called it "Tiohero," the lake of flags or rushes or lake of the marsh. The first white man known to have visited its shores was a Jesuit priest whose journal (Raffieux, 1671-72) described the members of the Cayuga tribe as accomplished agriculturalists, fishermen, and hunters. They had probably modified the land extensively by annual burnings (Dudley, 1886; Thompson, 1972) as evidenced by the "almost continuous plains bordered by beautiful forests" observed by Raffieux at the northern end of the lake.

The orchards and fields of the Indians were laid waste by the punitive expedition under Sullivan during the War of the American Revolution. Ten years later the first white settlement, Ithaca, was started at the south end of the lake which by then was called Cayuga after its earlier inhabitants. A large influx of settlers followed the connection of Cayuga Lake to the Erie Barge Canal in 1821 and the completion of a lock in 1829 (Whitford, 1906).

The 1800's witnessed the growth of numerous small industries in the Ithaca area as the ready sources of power and water represented by the larger steep gradient streams such as Fall Creek were exploited (Anonymous, 1879). The basin was also heavily agriculturalized and as much as 80% of the land area may have been under cultivation by the turn of the century. Soil erosion must have reached massive proportions during periods of heavy runoff.

The development of the Appalachian coal fields, railroads and the exploitation of fertile prairie soils in the Midwest dictated a rapid decline in both industry and farming with a major abandonment of land taking place under Federal programs to combat rural poverty in the late 1920's and early 1930's.

GEOGRAPHY

Cayuga Lake is located (intersection of longitudinal and cross axes) at 40°41'30" N and 76°41'20" W at an altitude of 116.4 m (382 ft) above sea level (Greeson and Williams, 1970). Its catchment area (including lake surface) is 2,033 km^2 (785 mi^2) according to the U. S. Department of the Interior (1971).

The climate is of the humid continental type with warm summers and long, cold winters. The area lies on the main west to east track of cyclonic storms and hence its weather is highly variable and is characterized by considerable cloudiness. Annual precipitation ranges from 71-117 cm (28-46 in) with half of this normally falling from May through September. Seasonal changes (from Dethier and Pack, 1963) in several climatological properties are shown in Figure 1. Cayuga Lake has been frozen over its entire length during at least ten winters since 1796. Typically, however, sheets of ice extend out from the north and south ends only to about the 5-10 m depth contours with maximum coverage in February.

The bedrock of the basin consists mostly of acid shale and sandstone but is intersected by two major limestone formations (Rickard and Fisher, 1970). Outcroppings of the southernmost formation extend to nearly the head end of the lake. The soils of the northern two-thirds of the Cayuga basin are dominated by moderately coarse textured types with calcareous substrata. Those of the major tributaries and highlands surrounding the southern part of the basin are composed of a diverse and complex assemblage and, in general, are less well drained and more acid (Cline and Arnold, 1970).

Cayuga Lake is located in an elongated, glaciated basin that opens into rather flat terrain at its north end but becomes progressively steeper towards the south. On the east side of the lake this rise becomes an obvious feature

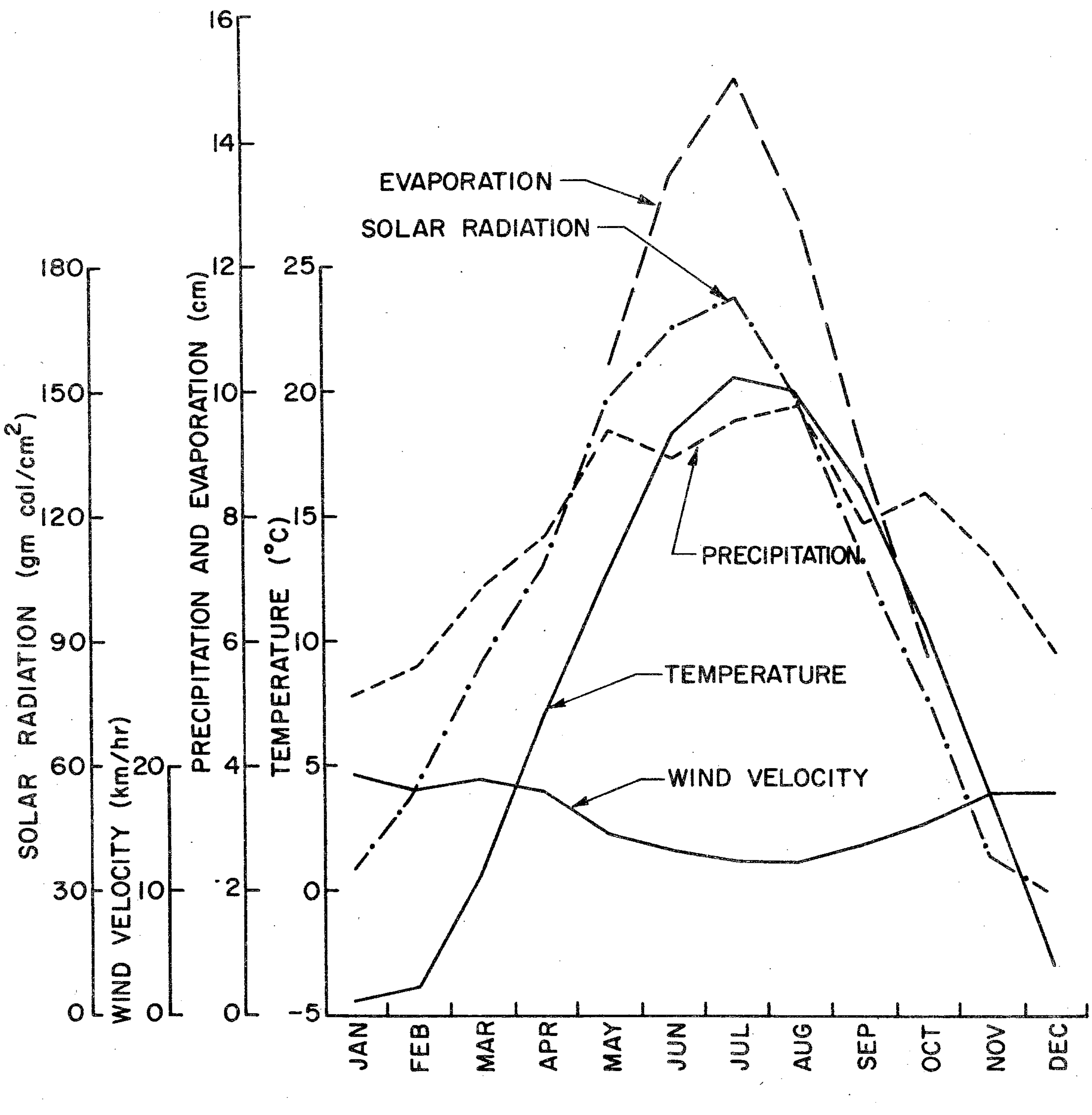

Figure 1. Monthly mean values for solar radiation, wind velocity, precipitation, evaporation and temperature at Ithaca, New York.

about one-third of the lake's length from its northern terminus and a similar increase in elevation occurs on the west side slightly further to the south. The upland plateau at the southern end is at an elevation of 250-300 m (800-1,000 ft) with the hills beyond occasionally extending to about 600 m (2,000 ft).

The dominant tree association in the northern one-fourth of the basin is elm-ash-red maple. The remainder of the lake border and the valleys are dominated by oak and hickory while the upland association is mainly beech-hard maple. Forage crops constitute the principal vegetation associated with agriculture. Child, Oglesby and Raymond (1971) determined the 1968 land usage to be: 904 km^2 (48.3%) active agriculture, 292 km^2 (15.6%) inactive agriculture, 582 km^2 (3%) of the usage falls into other categories such as transportation and mining. Using data from the 1970 census (U. S. Bureau of the Census 1970a and 1970b) the population of the basin is calculated to be 90,221.

Cayuga Lake serves as a supply of potable water for five towns or villages in the basin and an additional, combined supply for three other towns is under development. Water supplied to the major population center (City of Ithaca) in the basin comes from an impounded upland source. A 435 MW_e fossil fueled power plant currently takes its cooling water from the lake and a second such facility, almost double in size, is under consideration. Cayuga Lake is extensively used for fishing, boating and swimming. Three state, one city and one town park are located on its shoreline.

Industrial wastes discharged to Cayuga Lake tributaries are in excess of 5,109 m^3 (1,350,000 gal) day^{-1}. All are treated at the industrial sites prior to discharge and/or are put into the sanitary sewer system of Ithaca. In the past, large quantities of NaCl entered the lake as runoff from the site of a rock salt mine. Municipal waste is discharged to the lake at the rate of 23,881 m^3 (6.3 mg) day^{-1} with all but about 1% receiving at least secondary treatment. Additional treatment for phosphorus removal is, or will shortly be, given to 3,369 m^3 day^{-1} (14%) of the sewage being discharged.

MORPHOMETRIC AND HYDROLOGIC CHARACTERISTICS

Cayuga Lake has a simple eavestrough shaped basin with the steepest dropoffs and greatest depths occurring in the southern two-thirds of its length. Morphometric properties according to Birge and Juday (1914) are: 172.1 km^2 (66.4 mi^2) surface area, 61.4 km (38.1 mi) length, 5.60 km (3.50 mi) maximum width, 2.80 km (1.74 mi) average width, 153.4 km (95.3 mi) shoreline length, 3.35 shoreline development, 9,379.4 X 10^{-6} m^3 (331,080 X 10^{-6} ft^3) volume, 132.6 m (435 ft) maximum depth, 54.5 m (172.3 ft) mean depth, and 1.23 volume development. Their calculations indicate that a plane at the 40.3 m depth would divide the lake into two equal volumes.

The water level in Cayuga Lake is regulated by Mud Lock at the north end. The lake level is generally lowered by drawdown about mid-December and is again raised in the spring by input from snowmelt and rain. A maximum recorded lake level of 117.7 m (386 ft) occurred following Tropical Storm Agnes in 1972.

Using the data of Singley (1973) it can be seen that for 1950-52 and 1968-72 the relative volumes of the various thermal strata underwent a complex pattern of change over the stratified season (Figure 2). Three periods of downmixing and two when the relative volumes of the thermal strata remained constant are apparent. Duration of stratification is discussed below.

Sediments in the profundal zone of Cayuga Lake are fine textured mixes of silts and clays. From Ludlam's (1967) work it appears that 1.2-1.4 m (4 ft) of sediments have been deposited during the past 100 years. Littoral

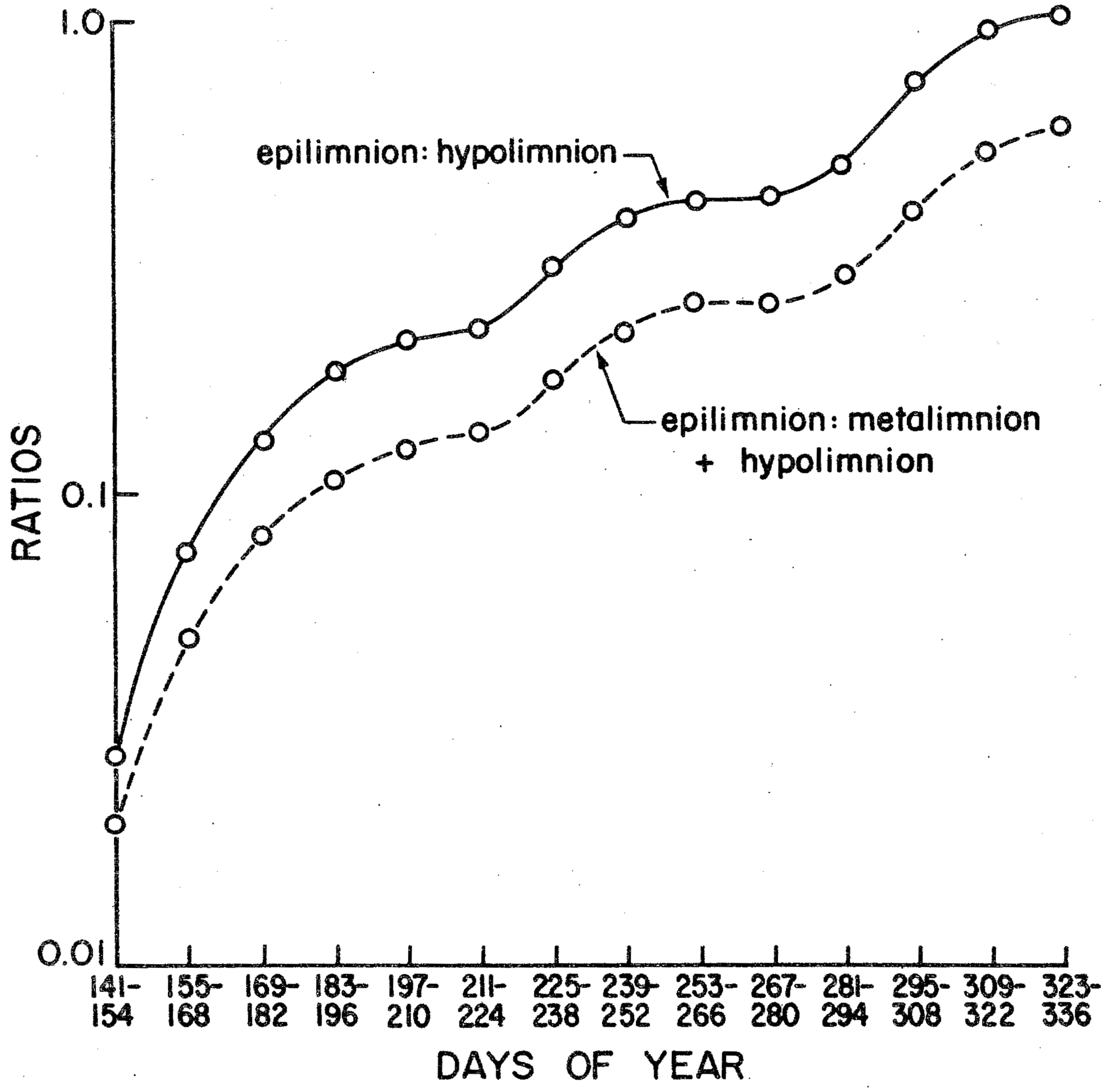

Figure 2. Ratios (by volume) of epilimnion to hypolimnion and of epilimnion to metalimnion plus hypolimnion as a function of season.

sediments at the south end of the lake, studied by Vogel (1973), were found to contain from less than 1% to 5% organic matter. Up to 3 m (10 ft) of sediment appear to have accumulated in portions of this area during the past three-quarters of a century based on comparisons of bathymetric charts (Maffa, personal communication).

Seasonal patterns of precipitation are shown in Figure 1. The heaviest single storm rainfall of record occurred on July 7-8, 1935 when 20.89 cm (8.22 in) fell and the second heaviest (17.91 cm or 7.05 in) was associated with a tropical storm June 21-25, 1972.

Cayuga Lake is perpendicularly intercepted at its north end by the Seneca River which is, at the same time, the major tributary and the sole surface outlet. In calculating material budgets the most logical course would seem to dictate the exclusion of Seneca River inputs since its entrance to and exit from the lake are so contiguous. Most of the other larger tributaries are located at or near the south end (Figure 3). Fall Creek has the highest annual flow followed respectively by Cayuga Inlet, Salmon Creek and Taughannock Creek (U. S. Department of the Interior, 1971). There are no measurements of subsurface flows, but the close agreement of most inflow and outflow estimates for surface waters indicates that groundwater is not likely to be a large component of the hydrologic budget.

Currents have been little studied. Sundaram et al. (1969) estimated that in mid to late September, 1968 typical wind induced surface currents were less than 3 cm (0.1 ft) sec^{-1}. At other times during the stratified season, major currents with velocities as great as 50 cm (1.6 ft) sec^{-1} were associated with seiche motions. They found significant hypolimnetic currents, shown in an example to be as high as 10 cm (0.3 ft) sec^{-1}, were found only in associations with seiches. There are several indications (Sundaram et al., 1969; Henson et al., 1961; and Wright, 1969a) of geostrophic effects but these have not

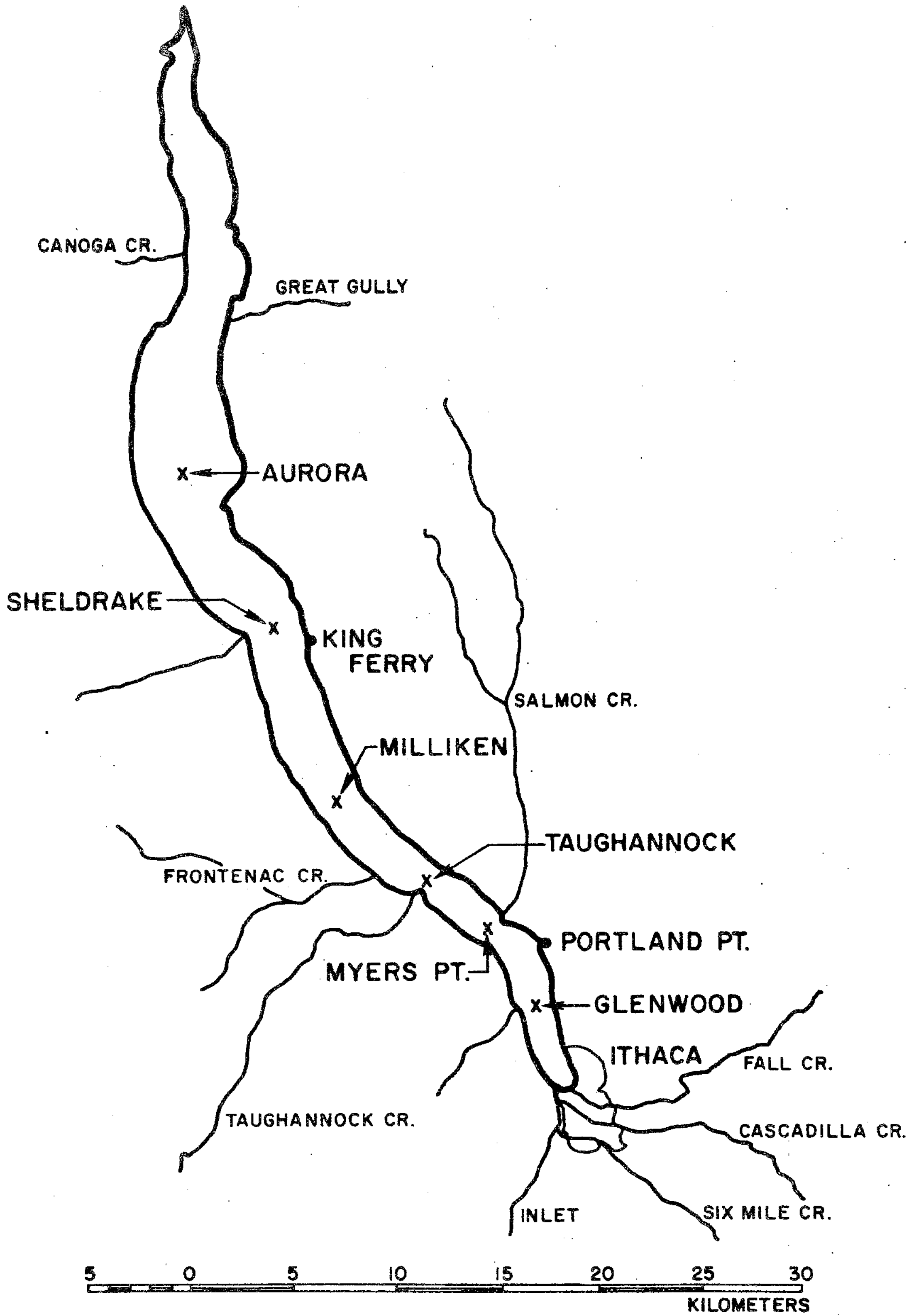

Figure 3. Location map of Cayuga Lake showing principal tributaries and sites of sampling stations used in the various limnological studies.

been systematically studied.

Henson et al. (1961), Wright (1969b), Singley (1973), Likens (1974b) and Oglesby (unpublished) have used differing methodologies and rationales to compute water renewal times in Cayuga Lake. Excluding the Seneca River from the calculations, Oglesby (unpublished) estimates this to have been 8.6 yr for the 1970-71 hydrologic year. Wright (1969b) has computed that, depending upon climatic conditions, the water renewal time may vary between 8.1 and 24.1 yr. Singley (1973) calculated that for 1965, an exceptionally dry year with higher than normal evaporation rates, the water renewal time was about 18 yr.

LIMNOLOGICAL CHARACTERISTICS - PHYSICAL AND CHEMICAL

Beginning with winter isothermy, a generalized temperature regime would show minimum homothermy at a temperature of 1.5-3.3°C sometime between late February and early April. Gradual warming but continued homothermy occur until about mid-May at which time surface temperatures begin to gradually increase. Stratification exists by mid-June or early July. Maximum summer bottom temperatures are largely a function of mixing in May and early June and vary from year to year between 4.1 and 5.5°C. Maximum recorded surface temperature for a given year ranges from about 20 to 27°C. Annual maximum bottom temperatures of 6.6 - 9.6°C are associated with fall homothermy which occurs between early November and December. The water column generally mixes freely until minimum homothermy is reached. Since 1910, one or more temperature profiles have been taken during seventeen years.

Specific conductance during the winter is about 600 μmhos cm^{-1} (Wright, 1969c). As a result of ion dilution by heavy spring runoffs of snowmelt and rain, values decrease to below 500 μmhos cm^{-1}. A gradual increase takes place over the summer, especially in the hypolimnion, with temporary decreases occurring in association with periods of heavy precipitation (Dahlberg, 1973). The higher specific conductances characteristic of the hypolimnion during the stratified period are thought to be due to deep groundwater inflow or solubilization of bedrock within the basin proper.

Data on solar radiation and light extinction coefficients have been summarized for 1968-70 (Table 1) by Peterson (1971).

Table 1. Monthly means of solar radiation and extinction coefficients during 1968-70. Number of values averaged are shown in parentheses.

Month	Solar radiation (gm cal cm^{-2} day^{-1})	Extinction coefficient (k)
Jan	112 (3)	0.250 (1)
Feb	221 (3)	0.292 (1)
Mar	293 (3)	0.250 (1)
Apr	421 (3)	0.463 (2)
May	450 (3)	0.301 (4)
Jun	511 (3)	0.370 (3)
Jul	529 (3)	0.854 (4)
Aug	472 (3)	0.598 (10)
Sep	358 (3)	0.403 (4)
Oct	234 (3)	0.321 (2)
Nov	103 (3)	0.286 (3)
Dec	93 (3)	---

Color was reported (Dahlberg, 1973) to average 6 mg l^{-1} (presumably he meant color units) in samples taken from various strata at a series of stations during the summer and early fall of 1972.

In reviewing the published (Wagner, 1927; Burkholder, 1931; Henson et al., 1961; Wright, 1969d; and Dahlberg, 1973) and unpublished data on hydrogen ion concentration in Cayuga Lake, a general pattern emerges of minimum water column averages (pH 7.7-8.0) during the winter months. An increase to a pH of about 8 occurs prior to stratification. During the summer, hypolimnetic pH decreases fairly rapidly to a low of 7.7-7.8, with occasional values to

7.5 prior to autumnal mixing. At the same time pH in the epilimnion reaches a maximum (as high as 9.0) averaging 8.2-8.4. The hydrogen ion concentration then drops during the mixing period and reaches its winter minimum in January or February.

Dissolved oxygen follows a pattern to be expected for a cold, deep, moderately productive lake. During the summer, daytime supersaturation is fairly common in the epilimnion, and hypolimnetic concentrations decrease seasonally, reaching a minimum of about 6 mg l^{-1} in the deepest portion of the lake just before fall overturn. The water column is only 80-90% saturated at the time of complete autumnal mixing. Dissolved oxygen increases gradually during the winter and reaches 90-95% of saturation by the time thermal stratification is reestablished in the summer. Hypolimnetic minima do not appear to have changed since the early part of this century. Data on the spatial and temporal distribution of dissolved oxygen have been reported by Birge and Juday (1914), Wagner (1927), Burkholder (1931), Henson et al. (1961), Wright (1969e), and additional, unpublished (Godfrey and Oglesby) records are available for 1972-74.

Total phosphorus (TP) typically ranges from about 15 to 22 mg m^{-3} throughout the water column during all seasons of the year (Peterson, 1971; Oglesby and Schaffner, MS 1975). During the stratified season, this becomes partitioned in the epilimnion so that soluble reactive phosphorus (SRP) is only 5-15% of the total (Barlow, 1969) with resultant SRP concentrations being almost always less than 5 mg m^{-3} and often only 1-2 mg m^{-3} and with concomitant increases in soluble unreactive and particulate phosphorus (Peterson, 1971). In the hypolimnion SRP was nearly always 50% and sometimes as much as 90% of the total during 1968 (Barlow, 1969). Over a three year period (1968-70, n = 133) of sampling Peterson (1971) found that TP ranged from 9.1 to 56.7 mg m^{-3}. Seasonal variations in the forms and concentrations of phosphorus

(Wright, 1969f; Barlow, unpublished; and Godfrey, unpublished), an elegant series of continuous culture bioassays (Peterson et al., 1973) and alkaline phosphatase activity (Griffin, 1974) all indicate that phosphorus is the critical element in controlling the level of summer phytoplankton production.

Nitrate nitrogen varies with depth during the summer but the most marked fluctuations occur between seasons and, on occasion, between years. Concentrations are almost always high enough to be in excess of the minimum needed for unrestricted phytoplankton growth (Barlow, 1969) with the mid-summer period of 1973 being an exception (Godfrey, unpublished). Maximum input is via tributary inflow during the spring and concentrations are still typically 800-900 mg m^{-3} in mid-May. Following stratification, nitrate decreases erratically in the epilimnion and a slight vertical cline of increasing concentration becomes apparent (Federal Water Quality Administration, 1965; Wright, 1969f; Dahlberg, 1973; Godfrey, unpublished). Data on ammonium nitrogen are scarce but concentrations appear to be generally low, ca. 0-40 mg m^{-3} (Dahlberg, 1973). In the summer of 1972, Kjeldahl nitrogen ranged from about 200-500 mg m^{-3} (Dahlberg, 1973) with hypolimnetic concentrations being lower than those of the surface water but with maxima sometimes occurring in the metalimnion.

Total alkalinities of Cayuga Lake water are on the order of 100 mg l^{-1} as $CaCO_3$ (Wagner, 1927; Burkholder, 1931; Henson et al., 1961; Wright, 1969d; Dahlberg, 1973; and Godfrey, unpublished). Winter values are generally higher than this and an annual minimum occurs in July - September. The variation within a year is 10-15 mg l^{-1} as $CaCO_3$. During the stratified season there is a slight increase in alkalinity with depth. The only published values of acidity are those of Dahlberg (1973) for 1972. Mean, minimum and maximum concentrations were, respectively, 2.6, 0 and 9.3 mg l^{-1}. Increases during the stratified season were noted for metalimnetic and hypolimnetic samples.

Cayuga Lake has a well developed calcium carbonate buffer system, and concentrations of sodium and chloride are unusually high for an inland lake in the northeastern United States (Federal Water Quality Administration, 1965; Berg, 1966; Dahlberg, 1973; and Oglesby, unpublished) as shown in Table 2.

Table 2. Major anions and cations in Cayuga Lake during April, 1973 as determined from samples composited for depth.

Cations			Anions		
	MEQ	$mg\ l^{-1}$		MEQ	$mg\ l^{-1}$
Ca^{++}	2.20	44.0	$CO_3^{=}$	0.07	2.0
Mg^{++}	0.86	10.5	HCO_3^{-}	2.00	122
Na^{+}	1.85	42.5	$SO_4^{=}$	0.76	37.0
K^{+}	0.07	2.8	Cl^{-}	2.34	83.2
TOTAL	4.98		TOTAL	5.17	

Data on inorganic trace elements are summarized in Table 3.

Table 3. Inorganic trace elements in Cayuga Lake based on data obtained in the summers of 1971 and 1973. With the exception of those for boron, "observations" represent averages for samples taken from two or three depths corresponding to the major thermal strata at from one to five sampling stations.

Element	Range ($mg\ m^{-3}$)	Typical concentration ($mg\ m^{-3}$)	Number of observations	References
Fe	3-220	-	11	Dahlberg (1973), Oglesby (unpublished)
Mn	1-30	-	4	Oglesby (unpublished)
Bo	22-55	34	21	" "
Zn[1]	0.51-9.41	2.7	4	Mills and Oglesby (1971)
Cu[1]	0.10-0.93	0.6	4	" " " "
Pb[1]	0.10-0.93	0.12	4	" " " "
Cd[1]	0.015-1.98	0.54	4	" " " "
Co[1]	0.003-0.093	0.005	4	" " " "
Al	0-20	-	2	Dahlberg (1973)
Mg	0.6-14	-	5	" "

1] Insoluble form, euphotic zone

Data on pigment concentrations in Cayuga Lake have been reported by Hamilton (1969), Wright (1969g), Barlow (1969), Peterson (1971), Dahlberg (1973) and Oglesby and Schaffner (1975) and detailed information for 1972-73 has been collected by Godfrey (unpublished). An annual maximum is generally found in late June or July and a secondary peak often occurs in the autumn. Peterson's (1971) chlorophyll a concentrations for June, July and August of 1968-70 averaged 4.8 mg m^{-3} in the euphotic zone. For the same period in 1972-74, epilimnetic mean chlorophyll a plus phaeophytin ranged from 7.8 to 9.7 mg m^{-3} (Oglesby and Schaffner, MS 1975).

Primary productivity, as determined by ^{14}C uptake, values have been reported for 1957-58 (Howard, 1963) and 1968-70 (Barlow, 1969 and Peterson, 1971). The variation between years is considerable, but for the latter period production is about 160 mg C m^{-2} day^{-1} averaged on an annual basis.

Bioassays to determine nutrients critical to the growth of Cayuga Lake phytoplankton are among the best designed and most comprehensive of any so far done for freshwater systems. Using continuous cultures of naturally occurring phytoplankton communities in lake water, the role of phosphorus in limiting growth during mid and late summer of 1971 and 1972 has been convincingly demonstrated (Barlow et al., 1973 and Peterson et al., 1973).

The phytoplankton of Cayuga Lake is comprised of a mixture of associations some of which have been described in the literature as being indicative of oligotrophy and others as typifying eutrophic conditions. Myxophycean "blooms" occur at times during the summer but are not persistent. Seasonal patterns of succession and peaks of abundance as indicated by cell counts, species biomass, and pigment concentration are highly variable from year to year. A general pattern of maximum standing crop from late June into early October exhibits large week to week fluctuations with surface chlorophyll a ranging from a low of near 1 mg m^{-3} to over 20 mg m^{-3} on one occasion in 1972. Both cell counts and species composition indicate a probable trend to more eutrophic conditions when data from 1910-1930 are compared with those for 1950-74. Barlow (1969), Dahlberg (1973), and Godfrey (unpublished) have compiled comprehensive descriptions of the phytoplankton communities species composition in recent years.

The zooplankton (Birge and Juday, 1914; Birge and Juday, 1921; Muenscher, 1927; Bradshaw, 1964; Hennick, 1973; and Behrman, unpublished) and benthic fauna (Birge and Juday, 1921; Henson, 1954; Green, 1965; and Dahlberg, 1973) are typical of deep, moderately productive north temperate latitude lakes. There is no evidence of qualitative changes in either over the last sixty years, but limited data indicate that summer standing crops of zooplankton may have increased. Cyclops bicuspidatus is the dominant copepod and Bosmina longirostris the most abundant of the Cladocera in the summer zooplankton. Abundant Mysis relicta are an important food resource for some species of fish.

The fish population is managed to maximize salmonid production (Youngs and Oglesby, 1972), Significant sport fisheries also exist for smelt, smallmouth bass and other species. The principal food chain associated with the limnetic zone is: phytoplankton and detritis $\longrightarrow$ zooplankton $\longrightarrow$ alewives $\longrightarrow$ salmonids.

The bacterial flora is virtually undefined. A limited amount of data are provided by Dahlberg (1973). The benthic flora, exclusive of rooted plants, is unstudied. Cladophora sp. is abundant is the littoral zone in some locations and fishermen report that growths of this attached alga have increased in recent years.

Dense growths of rooted macrophytes occur in a limited area of shallow water at the southern end of the lake and over a much larger area at the northern end. Historical data on plant growths at the head end of the lake indicate a possible increase in plant density and a shift to species, especially millfoil, that constitute more of a nuisance (Vogel, 1973; Oglesby, Vogel, Peverly and Johnson, MS 1974).

NUTRIENT BUDGETS

Budgets and loadings for three different kinds of phosphorus are given in Table 4. Inputs of total phosphorus and molybdate reactive phosphorus

Table 4. Phosphorus inputs and loadings (excluding the Seneca River) for total and molybdate reactive phosphorus (1970-71) and for "biologically available" phosphorus (1972).

Source	Total P ($kg \times 10^{-3}\ yr^{-1}$)	MRP ($kg \times 10^{-3}\ yr^{-1}$)	"Biologically available" P ($kg \times 10^{-3}\ yr^{-1}$)
Waste discharges	88.6	88.6	88.6
Land runoff	47.4	2.0	26.1
Precipitation	3.4	3.4	3.4
Ground water	?	?	?
Other	?	?	?
Total	139.4	94.0	118.1
Volumetric loading ($mg\ m^{-3}\ yr^{-1}$)	14.9	10.0	12.6
Areal loading $g\ m^{-2}\ yr^{-1}$	0.81	0.54	0.69

(unfiltered) are based on a one year study by Likens (1972, 1974a and b) in which the contributions of P in precipitation and in 25 tributaries (draining almost 78% of the lake basin watershed) were monitored. "Waste discharge" and "Land runoff" categories were subsequently determined by calculating the former based on estimates of per capita discharge of phosphorus to the tributaries and adding to this the P in wastes discharged directly to the lake (Oglesby and Schaffner, 1975 and Oglesby and Schaffner, MS 1975). Phosphorus from land runoff was then determined by difference. The budget for "biologically available" P contains a "Land runoff" estimate based on the use of export coefficients, determined during an intensive 18 month study of the Fall Creek watershed (Bouldin, unpublished), for the sum of SRP, dissolved unreactive P and the fraction of P associated with suspended particulates that desorbs in aqueous solution. The runoff (export) coefficients (mg m^{-2} yr^{-1}) for land in various use categories were: 13.2 for agriculture, 8.3 for forest and 100 for residential usage.

A budget for soluble nitrogen exclusive of organic N, is given in Table 5. The essential components are derived from Likens' (1972, 1974a and b)

Table 5. Soluble nitrogen inputs and loadings (excluding the Seneca River) for 1970-71.

Source	Soluble N (kg x 10^{-3} yr^{-1})
Waste discharges	200.3
Land runoff	1,694.1
Precipitation	565.6
Ground water	?
Other	?
Total	2,460.0
Volumetric loading (mg m^{-3} yr^{-1})	262
Areal loading (g m^{-2} yr^{-1})	14.2

1970-71 study of tributary and precipitation inputs. As was the case for phosphorus, the "Waste discharge" category was calculated and "Land runoff" obtained by difference for the tributary input. In the calculation of the former a per capita discharge of 4.44 kg yr^{-1} (Olsson,Karlgren and Tullander, 1968) and a treatment efficiency (all types of disposal systems) for N removal of 50% were assumed.

Other macronutrient budget information calculated by Likens is summarized in Table 6.

Table 6. Sulfur, silicon, calcium, magnesium and bicarbonate inputs and loadings from precipitation and tributary inflow (excluding the Seneca River) during 1970-71.

Nutrient	Input ($kg \times 10^{-3}\ yr^{-1}$)	Volumetric loading ($mg\ m^{-3}\ yr^{-1}$)	Areal loading ($gm\ m^{-2}\ yr^{-1}$)
$SO_4^{=}$ - S	13,253	1,410	76.6
SiO_2 - Si	2,147	229	12.4
Ca^{++}	63,802	6,800	369
Mg^{++}	15,263	1,630	88.2
HCO_3^- - C	31,523	3,360	182

DISCUSSION

As a moderately large and deep, cold water lake affected by a variety of human influences, Cayuga is representative of many important bodies of water formed in north temperate latitudes by the action of glaciers. Its relatively uncomplicated morphology (low shoreline development, restricted littoral zone except at the tail end, and single basin) make Cayuga an excellent site for elucidating limnological principles. The existence of a substantial body of knowledge, accumulated over the past century, places it among the limnologically better defined lakes in the world.

For the present emphasis on examining primary production in an ecological context, adequate data are available on the more static properties, e.g., geology and morphometry, and, for one or more years, on many of the more changeable parameters such as the distribution and loading of primary nutrients, algal standing crops and transparency. Data on primary production rates, grazing, and benthic productivity are more limited.

Based on most biotic and associated abiotic descriptive properties, Cayuga Lake falls in the mesotrophic category; yet, for given parameters and at specific times, it could be termed either eutrophic or oligotrophic.

The composition of the phytoplankton is especially illustrative of an intermediate trophic status since dominant groupings commonly cited as being typical of both oligotrophic (e.g., *Cyclotella*, *Tabellaria*, chrysomonads and *Sphaerocystis*) and eutrophic (e.g., Myxophyceae and *Melosira*) conditions occur. Mean summer euphotic zone chlorophyll concentrations (ca. 5-10 mg m^{-3}), primary production rates (annually on the order of 160 mg C m^{-2} day^{-1}), hypolimnetic dissolved oxygen (minimum concentration of about 6 mg l^{-1}), the composition of the fauna (the fishes include both salmonids and carp, *Mysis relicta*), and the standing crop of profundal benthos (0.5-1.0 gm organic matter m^{-2}) reinforce the picture of mesotrophy. There is evidence that productivity has increased when data from the 1910-1930 period are compared with those from 1950-1974.

When data on Cayuga Lake are fitted to the graphics of trophic state as a function of total P loading vs. mean depth (Vollenweider, 1968) or vs. the ratio of mean depth to water residence time (Vollenweider as given by Dillon, MS 1974), a eutrophic condition (about the same as Mälaren) is indicated with total P loadings above the so-called "dangerous" level. The reasons for this lack of fit to the Vollenweider plots can only be speculated on at present. Several possible factors are: (1) 15% of the total phosphorus loading, namely that adsorbed to soil particles, is estimated to remain unavailable for biological uptake, (2) Cayuga's simplified morphology and aerobic hypolimnion probably minimize the internal recycling of phosphorus compared with that which occurs in some lakes, (3) there could be significant errors in the calculation of specific phosphorus loading, and (4) the parameters used in the Vollenweider plots are invalid, or at least inaccurate, in defining trophic state.

Ignoring mean depth and water retention time, Oglesby and Schaffner (MS 1975) have obtained the following relation between summer chlorophyll (mean for the epilimnion) and the specific loading of "biologically available" phosphorus for New York's Finger Lakes.

$$Y = 21.8X - 1.57 \qquad (r = 0.62, n = 21)$$

They postulate that depth becomes an important factor only when it is necessary to separate lakes that essentially mix to the bottom throughout most of all of the year from those that exhibit summer stratification.

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TROPHIC STATUS AND NUTRIENT BALANCE FOR CANADARAGO LAKE

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INTRODUCTION

Canadarago Lake, located in Otsego County, east-central New York State has been the scene of an intensive investigation by the New York State Departments of Environmental Conservation and Health.

Canadarago Lake is a stratified lake of moderate size (759 ha). From its morphometry (7m mean depth), it can be expected to be moderately eutrophic but at the beginning of the New York State study, appeared strongly eutrophic with dense blue-green algae blooms, a condition which appeared to be caused by the input of sewage from the village of Richfield Springs and from summer camps. When the study began in 1968, Richfield Springs was under State Health Department orders to stop discharging raw sewage into the lake.

The advanced state of eutrophication in this lake called for the construction of a modern sewage treatment plant which included some form of nutrient removal. The concern of the local residents, the proximity of the lake to Albany, and the fact that Canadarago Lake typically represents the condition of a number of lakes within the state, made it a logical candidate for a pilot demonstration study.

The only prior published data concerning an investigation of Canadarago Lake occurred as the result of the biological survey of the Delaware and Susquehanna watersheds performed by the New York State Conservation Department during the summer of 1935 (1). Even at this early data Canadarago Lake showed evidence of eutrophication. Throughout the text comparisons of the present study with that of 1935 are presented.

BRIEF GEOGRAPHIC DESCRIPTION OF WATER BODY

A-D. GEOGRAPHICAL DATA

Canadarago Lake is situated in east-central New York State, Figure 1, at an elevation of 390m (1280 ft) above mean sea level. Canadarago Lake together with its 175 km^2 of drainage area, Figure 2, forms the northeastern headwaters of the Susquehanna River watershed, originating in Herkimer and Otsego counties. The drainage basin for this lake is bounded between 74° 53' 33" West Longitude and 42° 46' 18" North Latitude with the centroid of the lake located at 75° 00' 25" West Longitude and 42° 49' 00" North Latitude. The surrounding terrain is hilly, with ground elevations ranging from 396m (1300 ft) to 579m (1900 ft) above mean sea level.

Four major tributaries, Figure 3, drain 78.3 percent of the watershed: Ocquionis Creek which discharges at the north end of the lake, and Mink Creek, Hyder Creek and Herkimer Creek which discharge along its western shore. The eastern portion of the watershed is too narrow and steep to support permanent streams. The lake is drained at its southern end by Oaks Creek, which flows south to join the Susquehanna River at Index, New York.

FIGURE I.
LOCATION OF CANADARAGO LAKE
IN NEW YORK STATE

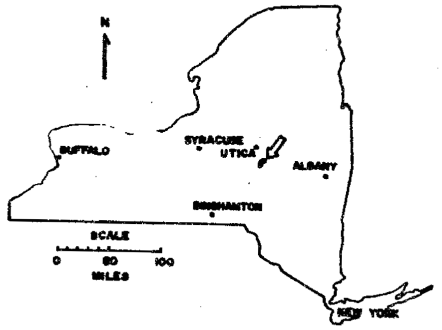
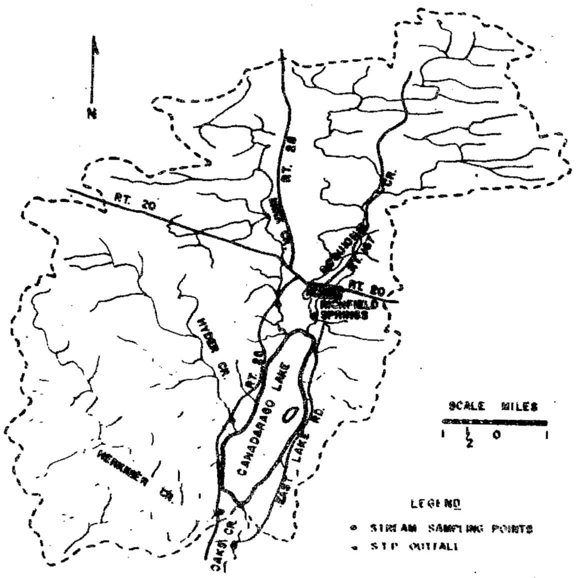


FIGURE 2.
CANADARAGO LAKE WATERSHED



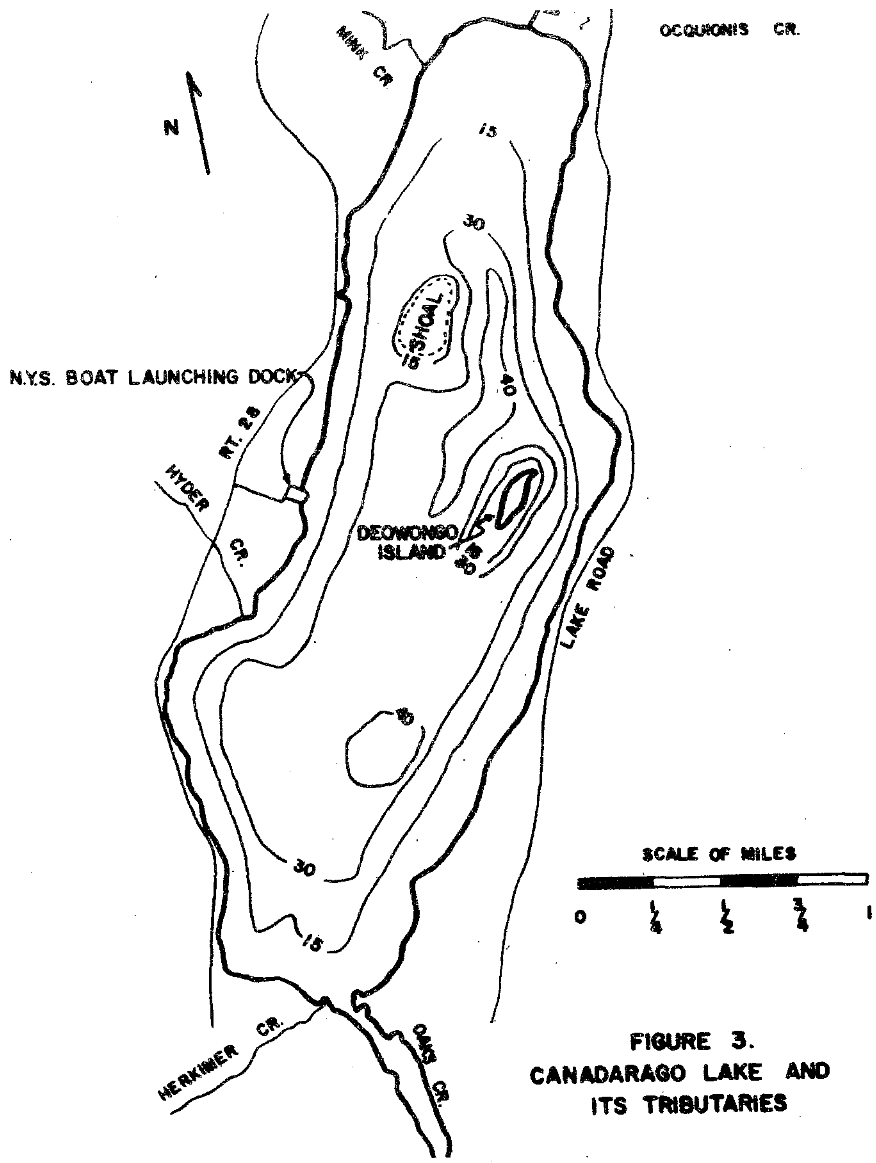


FIGURE 3.
CANADARAGO LAKE AND ITS TRIBUTARIES

E. GENERAL GEOLOGICAL CHARACTERISTICS

The bedrock geology and Pleistocene glacial modifications are strongly reflected by the present physiography of the Canadarago Lake drainage basin. The bedrock of the drainage basin is predominantly Onondaga and Helderberg limestone in the north and the Hamilton shales and siltstones in the south. The contact between these two formations is the boundary between two distinct physiographic units.

This area of New York State was glaciated several times during the Pleistocene epoch, but evidence is preserved only for the latter stages of the Wisconsin Glacial period. Two major glacial lobes thrust over the drainage area during this Glacial period, approximately 11,000-12,000 years ago. One advance was in the north-south direction and was probably responsible for the outwash deposits found to the south of the drainage basin. This advance may have been responsible for forming the oversteepened north faces of the shale siltstone ridges which predominate in the southern half of the drainage basin. The second advance was in the west-southwesterly direction. This advance is marked by several endmorainic deposits in the northern section of the drainage basin. Also, the lack of high-lime glacial drift on the southern portions of the drainage basin indicates little movement of limestone in the north to south direction. Flint (2) estimated the thickness of the ice sheet at the time of its maximum advance to have been 1000 to 1200m (3281 to 3937 ft). The glacial and subsequent periglacial periods strongly influenced the character of soils found in the drainage basin. Ocquionis and Mink Creeks originate in Herkimer County and drain from the gently undulating east-west oriented limestone unit. The stream forms a typical trellis drainage pattern. The valley floors contain many swamps and muck deposits. Local relief between valley floors and ridge tops is generally less than 30m (98.5 ft). Hyder and Herkimer Creeks, in contrast, originating in Otsego County, are in the shale upland unit. Physiographically, this area is characterized by a strong local relief and dendritic drainage patterns. The local relief is as much as 200m (656 ft) from valley floors to surrounding ridges in some places. The streambeds are on gravel or bedrock. There are no muck deposits within the major stream system.

F. VEGETATION

On the slopes and hills of the Canadarago Lake watershed are woods of mixed deciduous trees, primarily maple and oak. A narrow band of trees surround the lake and an agricultural belt is located between them. A swampy woodland is located at the southern end of the lake.

G. POPULATION

The village of Richfield Springs is the only significant, permanent population concentration in the watershed. In 1970, the population of the village was 1527 (3) and records indicate that the population has been nearly constant for 20 years (4, 5). During the summer months, approximately 1300 additional people occupy summer cottages around the lake shore. The total permanent resident population of the entire watershed is not known but it is estimated to be on the order of 3500 people.

H. LAND USAGE

About 49 percent of the watershed is devoted to agriculture, mostly dairy farms, and approximately 34 percent is in forest or brushland (6,7,8). Table 1 is a summary of the land use within the watershed. Using the

Table 1. LAND USE IN THE CANADARAGO LAKE WATERSHED

Use	Area (km^2)	Watershed Area (Percentage)
Agriculture and Agriculture Facilities	86.53	49.00
Forest	60.10	34.02
Water Resources	12.38	7.01
Wetlands	11.49	6.51
Residential	3.68	2.26
Commercial	0.68	.38
Industrial	0.07	.04
Mining	0.35	.20
Public, Semipublic and Transportation	0.46	.26
Outdoor Recreation	0.56	.32
Total	176.61	100.00

1964 agricultural census, (9, 10), it has been estimated that there are approximately 6000 cattle, 40 hogs, 50 sheep and 5000 chickens in the watershed. A more extensive study of soil type and land use has been conducted by the Department of Agronomy at Cornell University, Ithaca, New York which adds support to these data (11).

I. WATER USAGE

The lake is used primarily for recreational purposes, offering recreational opportunities for the urban residents of Albany, 100 km (60 miles) to the east, and Utica, 40 km (25 miles) to the northwest. The recreational potential of Canadarago Lake has long been recognized and utilized. Around the turn of the century, this lake and its larger sister lake, Otsego Lake, were sites of summer homes and health spas for the wealthy. Although the economic strata of the users have changed, the recreationists today are the source of a substantial portion of the area's economy.

A study of the economic contribution of the recreational aspects of Canadarago Lake by the Soil Conservation Service of the United States Department of Agriculture (12) revealed that more than \$663,000 annual sales were directly related to the lake and its existing recreational facilities. In addition, the lake oriented properties contribute about \$90,000 annually in local real estate and school taxes. This study further concluded that the lake and its recreational assets are a significant contributor to the local community and that, if the Richfield Springs area was deprived of the lake, the area could undergo the economic decline being experienced by many other rural communities in New York State.

J. SEWAGE AND EFFLUENT DISCHARGE

The village of Richfield Springs has been served by a combined sewer system which discharged through a primary wastewater treatment plant, to Occquionis Creek, at a point approximately 0.8 km (0.5 mile) upstream of its mouth. The plant had not been operational for several years since it needed significant repairs. The cottages and residences located around the perimeter of the lake are served by septic tanks. In 1969, a New York State Health Department survey revealed that 24.4 percent of the septic tanks had some type of direct discharge into the lake, by-passing the leaching fields (13).

During 1972 the village of Richfield Springs constructed a modern wastewater treatment facility to replace the former sewage treatment plant. The effluent from the new facility is discharged to Ocquionis Creek at the same point as from the previous facility. Construction of the facility was completed in the summer of 1973. In November 1972, the plant began operation as a secondary treatment plant. In January 1973, the tertiary system for removal of phosphorus was completed.

This facility, operating as a tertiary treatment plant, is capable of treating 0.37×10^6 gal $\cdot$ day^{-1}($1.4 \times 10^3 m^3 \cdot$ day^{-1}). The lagoon system, which provides secondary treatment, can handle up to 2.5×10^6 gal $\cdot$ day^{-1} ($9.5 \times 10^3 m^3 \cdot$ day^{-1}). Flow in excess of 0.37×10^6 gal $\cdot$ day^{-1}(1.4×10^3 $m^3 \cdot$ day^{-1}) is given only secondary treatment and disinfection at the plant.

Wastewater processed through tertiary treatment will provide 93-94% BOD removal and up to 90% phosphorus removal, or to a maximum effluent concentration of 0.5 mg P $\cdot$ liter^{-1}.

In addition to the wastewater treatment plant discharge, natural mineral springs effluents enter the lake through the Richfield Springs wastewater treatment plant and Ocquionis Creek, introducing quantities of sulfate, sulfide, magnesium and calcium as pollutants. Another source of pollution is a stockpile of road salt located near Mink Creek which introduces additional chlorides to the lake.

BRIEF DESCRIPTION OF MORPHOMETRIC AND HYDROLOGIC CHARACTERISTICS

A. LAKE LOCATION AND DESCRIPTION

Canadarago Lake is nearly 6.4 km (4 mi) long, running north-south but is only 1.9 km (1.2 mi) wide at its maximum width. The shore length of the lake is 14.4 km (9 mi) and 80 percent of the area around the lake is densely populated with summer homes, trailer parks and year-round residences. During the summer months, seasonal transient occupancy of these facilities increases the population of the area by approximately 1300 people.

An island, Deowongo Island, is located nearly midway between the northern and southern extremes of the lake and approximately 400 m (1300 ft) west of the eastern shoreline. The island possesses an area of approximately 3 ha (7.5 acres) and has a shoreline of approximately 0.8 km (0.5 mi). In addition, a shoal, submerged in 1 to 2 m (3.3 to 6.6 ft) of water, is located approximately 0.4 km (0.25 mi) from the western shore and 1.5 km (0.93 mi) south of the northern extreme of the lake. Nearly 10 ha (24 acres) of this shoal is submerged in 3 m (10 ft) or less, of water. Normally, in the summer, it is heavily covered with weed beds.

B. VOLUME

The volume of the lake has been calculated as 57.5×10^6 m^3 (2.03×10^9 ft^3) (14).

Until 1956 the lake was allowed to seek its own natural level. This frequently resulted in the lake becoming so low that large areas of the lake bottom around the perimeter of the lake became exposed. This condition reduced the lake's attractiveness for recreational and sporting use, decreased the asthetic quality of the area and frequently caused obnoxious disagreeable odors. To correct this situation, the Canadarago Lake Property Owners Association, in 1955, obtained permission from New York State to erect a regulatory barrier in Oaks Creek, the outlet from the lake. This barrier could be raised or lowered, as required, in order to maintain the lake at a convenient level. The barrier was constructed and put into operation in 1956 and still is controlled by the Property Owners Association.

C-D. DEPTH

The maximum water depth in Canadarago Lake is 12.8 m (42 ft) and the average depth is 7.7 m (25.3 ft). The lake area is 759 ha (2050 acres). The lake can be divided into two shallow areas, less than 5 m (16.4 ft) deep in its northern and southernmost parts. The remainder of the lake is between 5 and 10 m (16.4 and 32.8 ft) deep. The deepest points occur in a trench just north of the center of the lake and in a spot south of the center of the lake.

E-F. STRATIFICATION

Extensive mixing by the wind of this well-exposed lake produces a thermocline that is not very sharp and, during most of the season, is found at a depth of 6 to 8 m (19.7 to 26.3 ft) while secondary thermoclines move in from the surface. The mean depth of the epilimnion during the summer stratification is 6.7 m (21 ft). The epilimnion encloses approximately 72 percent of the lake volume and is maintained from June through September.

G. NATURE OF LAKE SEDIMENTS

The appearance of the sediments of Canadarago Lake is that of a silty black mud. Some sand which contains snail shells can be found along shoreline areas.

In August 1973, a sediment core was taken from Canadarago Lake by driving through 30 cm (12 in.) of sediment with a Kojak Brinkhurst (KB) corer. The core was extruded, fractionated in the field into 7.5 cm (3 in.) intervals, placed in polyethylene bags, frozen, and stored at -20°C (-4°F) until analysis. Chemical analysis of the core showed the macrocomponents of the sediment to be in a range typical of hardwater lakes: silica 270 mg $\cdot$ g^{-1}, calcium 100 mg $\cdot$ g^{-1}, aluminum 50 mg $\cdot$ g^{-1}, magnesium 0.6 mg $\cdot$ g^{-1} (expressed on a dry weight basis). Organic carbon and total phosphorus contents of the sediments, commonly regarded as indicators of the lake's trophic level, were found to be 50 mg $\cdot$ g^{-1} and 5 mg $\cdot$ g^{-1} respectively, suggesting moderate eutrophication. From previous surveys it was known that the sediments in the northern, unstratified portion of the lake, off Ocquionis Creek which has carried raw sewage, were softer and higher in organic matter than the sediments in the main basin and particularly in its southern part.

H. PRECIPITATION

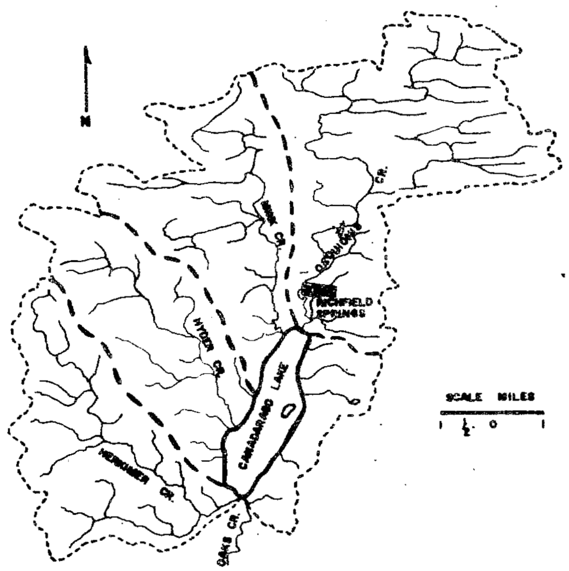
The area has a humid climate with cold winters and mild summers. The watershed is subject to occasional local cloudburst type of storms. A deficiency of rainfall frequently occurs extending through the upper few inches of soil during part of the summer.

The total annual precipitation for the Canadarago Drainage Basin over the 1951-1960 period was 1005 mm (39.3 in.) which includes an average annual snowfall of 2.5 m (8.2 ft). The mean temperature for this location, over the same period of time was 6.9°C (44.4°F) with varying extremes of 37°C (98°F) down to -36°C (-32.8°F) (15).

I. INFLOW AND OUTFLOW OF WATER

The watershed for the Canadarago Lake area can be divided into four natural areas, Figure 4, each drained by a creek which flows throughout the year, and the remaining area, which contains no year-round flowing drainage system, located primarily on the steep slope east of the lake. The former areas constitute 78.2 percent of the entire watershed.

FIGURE 4.
CANADARAGO LAKE SUBWATERSHEDS



From April 15, 1969 through April 14, 1970, data were collected for determining a water balance for Canadarago Lake (16). Flows of the four major influent streams, Figure 3, Ocquionis, Mink, Hyder and Herkimer Creeks, and the effluent stream, Oaks Creek, were measured using staff gauges, set by the U. S. Geological Survey, located near the mouth of each of the influent streams and at the head of the effluent stream.

The effluent from the wastewater treatment plant for the village of Richfield Springs, which flows into Ocquionis Creek below the gauging station and before its entrance to Canadarago Lake, was measured employing a 90 degree V-notch weir.

Using the flow data obtained from these sources, calculations were performed to synthesize a daily hydrograph for each source. A summary of these hydrographs is given in Table 2.

Table 2. WATER BALANCE FOR CANADARAGO LAKE

April 15, 1969 through April 14, 1970

Source	Average Flow		Drainage Area	
	m^3/s	Percent of Oaks Creek	km^2	Percent of Oaks Creek
Oaks Creek	2.865	100.0	175.0	100.0
Gauged Tributaries				
Ocquionis Creek	0.684	23.9	51.3	29.3
Mink Creek	0.414	14.5	27.2	15.5
Hyder Creek	0.375	13.0	27.5	15.7
Herkimer Creek	0.741	25.8	30.8	17.6
Sub Total	2.214	77.2	136.7	78.1
Wastewater Treatment Facility	0.021	0.7		
Total Gauged Inputs	2.235	78.0		

Evaporation losses from the lake were estimated by using surface water temperature over the same time period and climatological data from Albany, Binghamton and Syracuse, New York (17). These data yielded a total evaporation for the study period of 37,000 m^3 (1.3×10^6 ft^3), only 0.04 percent of the total annual effluent flow from Oaks Creek. During the summer, the evaporation rate was as high as two percent of the Oaks Creek flow. As shown in Table 2, the gauged influent tributary streams accounted for 77.2 percent of the effluent Oaks Creek flow and 78.2 percent of its watershed. A calculated value of influent, equal to the average gauged area runoff, was assumed for the ungauged areas. Examination of the morphology of the lake led to the conclusion that ground water outflows are negligible. Oaks Creek flow is, therefore, approximately equal to the total water input.

J. WATER CURRENTS

Very little study has been made of the water currents in Canadarago Lake. One dye study did, however, indicate that Herkimer Creek, which enters the lake at its southern end and near the mouth of the Oaks Creek outlet, may, at times, be shunted into the outlet without having much influence on the lake.

K. WATER RENEWAL TIME

In both 1969 and 1970, the spring melt occurred in late March and early April. The peak flows recorded during the 1970 melt, from the influent streams are listed in Table 3. The peak gauged input from these four streams was 33.1 $m^3 \cdot sec^{-1}$ (1169 $ft^3 \cdot sec^{-1}$). The peak lake discharge through Oaks Creek lagged behind the input by four days and reached only 18.8 $m^3 \cdot sec^{-1}$ The minimum recorded summer flows in the four tributaries, also listed in Table 3, occurred in late August and early September 1969. The minimum flow in Oaks Creek was 0.048 $m^3 \cdot sec^{-1}$ (1.7 $ft^3 \cdot sec^{-1}$) and occurred in late September, nearly one month after the minimum flows were recorded in the influent tributaries.

Table 3. MAXIMUM AND MINIMUM FLOWS OF THE [illegible] TRIBUTARIES TO CANADARAGO LAKE

April 15, 1969 - April 14, 1970

Source	Maximum Flow (cfs)	Maximum Flow (m^3/s)	Minimum Flow (cfs)	Minimum Flow (liter/s)
Ocquionis Creek	284	7.95	0.69	20
Mink Creek	240	6.72	0.16	4.5
Hyder Creek	272	7.70	0.10	2.5
Herkimer Creek	439	12.4	0.10	2.5

Because of the great variation in stream flow during the year, the spring melt constitutes one of the major annual events. During the period April 15, 1969 through April 14, 1970, about 43 percent of the lake's total gauged input and 32 percent of the gauged output occurred during the month of April, when the lake was not stratified (16). As a result, the average lake turnover time during April was about 60 days, whereas the annual average was about 231 days during the time period indicated. Based on 31 years of outflow data from Canadarago Lake through Oaks Creek (17), the average lake turnover time was calculated to be 217 days.

A summary of the Morphometric Characteristics of Canadarago Lake is shown in Table 4 (14).

Table 4. MORPHOMETRIC CHARACTERISTICS OF CANADARAGO LAKE

A.	Area	759 ha
B.	Mean Depth	7.7 m
C.	Maximum Depth	12.8 m
D.	Mean Depth : Maximum Depth	0.60
E.	Relative Depth	0.41%
F.	Volume	$57.51 \times 10^6 m^3$
G.	Development of Volume	1.8
H.	Mean Slope	0°36'8" exclusive of island and shoal
I.	Altitude	390 m
J.	Latitude	42°49'00" North
K.	Longitude	75°00'25" West
L.	Shore Length	14.4 km
M.	Development of Shoreline	1.47
N.	Littoral Development	Town of Richfield Springs; 80% of area around lake is densely populated with summer homes
O.	Number of Islands	One - Deowongo Island. One shoal.
P.	Area of Island	3.0 ha
Q.	Shore Length of Island	0.8 km
R.	Drainage Area	175 km^2
S.	Average Outflow	2.95 m^3/s calculated from average discharge for a period of 31 years
T.	Time of "Flushing"	217 days
U.	Average Precipitation	1005 mm

LIMNOLOGICAL CHARACTERIZATION SUMMARY

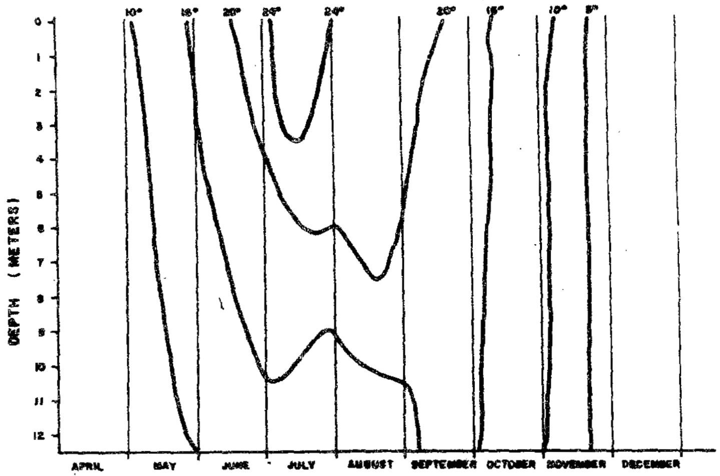
A. PHYSICAL

1. Temperature

The temperature profile of Canadarago Lake for 1968, Figure 5, is typical for this lake. It is observed that the maximum temperature in the deepest sections of the lake occurred in September (15°C, 59°F). It will be noted that the deepest 2½ meters (8.2 feet) of the lake seldom exceed a temperature of 15°C (59°F). During the winter season, usually from December through April, the lake is covered with a layer of ice that reaches a thickness of ½ meter (20 inches).

FIGURE 5.

CANADARAGO LAKE 1968 TEMPERATURES (°C)



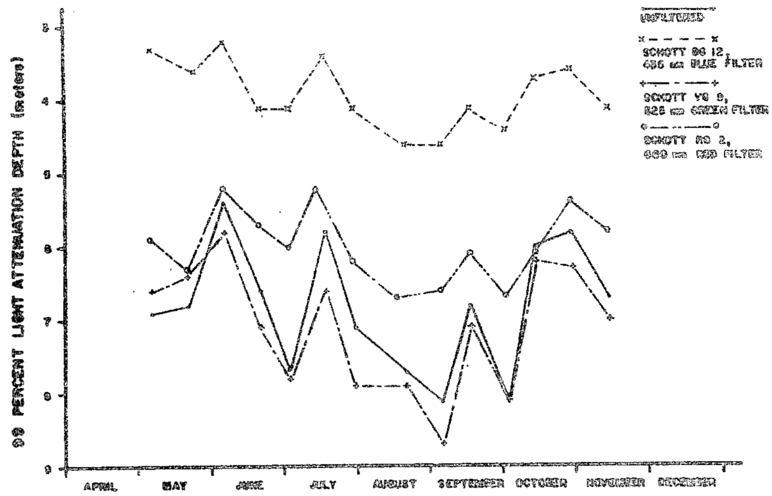
2. Conductivity

Conductivity measurements of Canadarago Lake were recorded as part of the New York State Water Quality Surveillance Network. Data for a five year period (1968-1972 inclusive) indicated a maximum conductivity of 374.0 μ mhos · cm^{-1} and a minimum value of 174.0 μ mhos· cm^{-1}.

3. Light Attenuation

Vertical light extinction in Canadarago Lake was determined from simultaneous measurements of surface and subsurface irradiation with a submarine photometer. The readings were converted to vertical extinction coefficients (per meter, in base-10 logarithms) using Table 10 by Sauberer (19) with estimated values of cloud cover and a calculated value for the zenith distance of the sun at true local time, and the geographical coordinates (43°N, 75°W), i.e. the elements of the "nautical triangle", and the procedures of spherical trigonometry (18). Figure 6 displays the 99 percent light attenuation depth for white light and the blue, green, and red regions of the spectrum as measured in 1969 at Station 5, located in the deepest section of the southeastern quadrant.

FIGURE 8.
1969 CANADARAGO LAKE LIGHT ATTENUATION
STATION 5



4. Secchi Disc Measurements

Measurements were made from May through November. During this period of time, when measurements were made at 10 different locations on the lake, a depth of greater than 2 meters (6.5 ft) was infrequently recorded. The highest reading recorded was 3.2 meters (10.5 ft) and the lowest reading recorded was 0.9 meter (2.9 ft) Averaging the readings recorded from the ten stations, the highest average reading occurred in the middle of September, 2.62 meters (8.6 ft), and the lowest average reading occurred in the middle of July, 1.08 meters (3.5 ft). All readings were made between 10:00 AM and 4:00 PM.

5. Solar Radiation

Solar radiation was measured at Canadarago Lake as part of the data recorded by the New York State Automatic Water Quality Acquisition System station on an hourly basis. During 1969 the maximum solar radiation recorded was 1.66 gram calories $\cdot$ $cm^{-2} \cdot hr^{-1}$.

B. CHEMICAL

1. pH

The pH of Canadarago Lake has been measured as part of the New York State Water Quality Surveillance Network, the New York State Automatic Water Quality Acquisition System and during the New York State Canadarago Lake Eutrophication Study. Five years data (1968-1972 inclusive) from the Water Quality Surveillance Network indicates a surface pH of 8.76 as a high value and 6.9 as a low value. During the Canadarago Lake Eutrophication Study, pH measurements were taken at approximately 14 day intervals during 1968 and 1969 at 10 different stations and, where possible, at three different depths. From this investigation values of pH from a low of 6.92 to a high of 9.16 were obtained. A summary of the average pH of the lake at three different depths is shown in Table 5.

Table 5. AVERAGE pH OF CANADARAGO LAKE, 1968-1969

Depth (meters)	pH 1968	1969
0-4.5	8.12	8.11
4.5-8.0	8.03	7.93
8.0-12.5	7.26	7.66

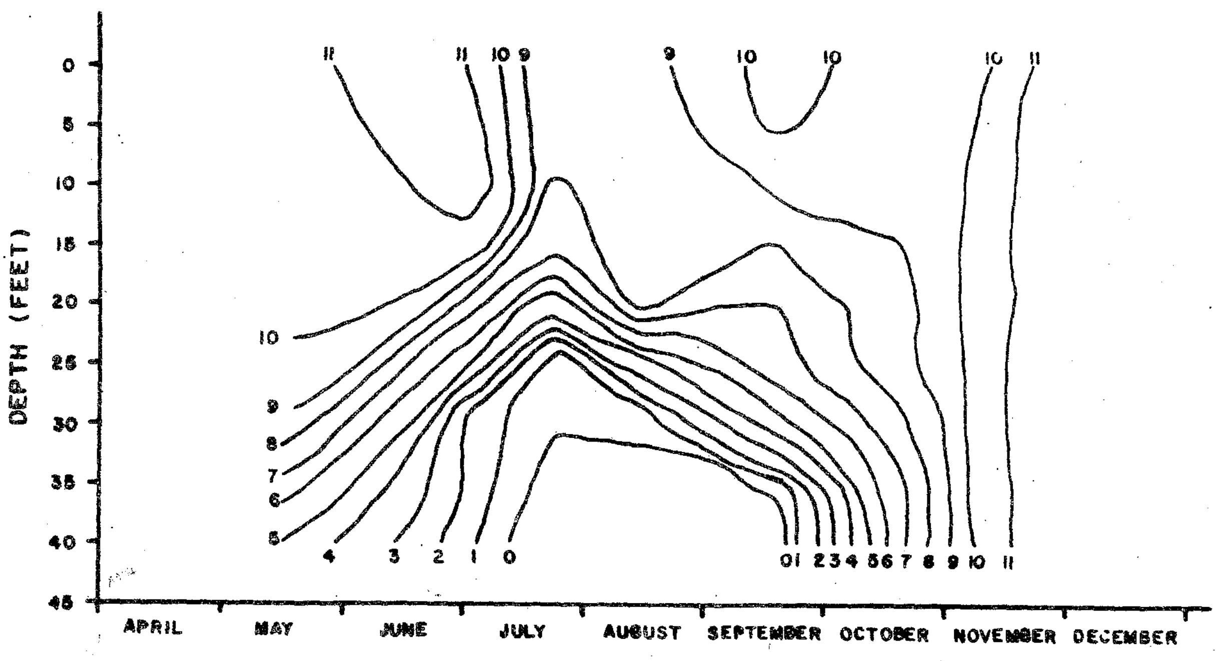
2. Dissolved Oxygen

Dissolved oxygen profile measurements were made at Canadarago Lake during 1968 and 1969. The measurements were made during the hours of 10:00 AM and 4:00 PM. A dissolved oxygen profile for 1968 is shown in Figure 7 (18). This profile is typical for the lake and indicates that the bottom 3 m (10 foot) depth of the lake becomes void of oxygen from early in July until late September. The 1935 survey (1) also reported the absence of oxygen in the deepest portions of the lake at the end of July.

FIGURE 7.

1968 CANADARAGO LAKE AVERAGE DISSOLVED OXYGEN CONCENTRATION

(parts per million = mg / l)



3. Phosphorus

Phosphorus determinations were made on Canadarago Lake samples collected at approximately 2 weeks intervals during 1968 and 1969. A variety of different forms of phosphorus were determined, including: orthophosphate, soluble and particulate phosphorus. The soluble and particulate phosphorus components were separated by means of vacuum filtration through a 47 mm diameter, 0.8 μ membrane filter. A summary of the average values of total phosphorus and the various phosphorus forms at three different depths of the lake are shown in Table 6 (18).

Table 6. AVERAGE CONCENTRATION OF CHEMICAL CHARACTERISTICS OF CANADARAGO LAKE, 1968-1969

Depth (meters)	1968 (micrograms per liter)	1969
	Soluble Phosphorus	
0-4.5	16.4	13.7
4.5-9.0	16.2	11.2
9.0-12.6	25.2	25.2
Total Lake	17.2	13.9
	Particulate Phosphorus	
0-4.5	31.4	29.1
4.5-9.0	29.3	30.0
9.0-12.6	50.0	37.7
Total Lake	32.4	30.5
	Total Phosphorus	
0-4.5	47.8	42.9
4.5-9.0	45.5	41.2
9.0-12.6	75.2	62.9
Total Lake	49.6	44.4
	Organic Nitrogen	
0-4.5	1308.4	764.9
4.5-9.0	1122.4	585
9.0-12.6	1297	573.3
Total Lake	1246.7	709.2
	Ammonia Nitrogen	
0-4.5	70.9	120
4.5-9.0	135.3	148.3
9.0-12.6	585.5	439.1
Total Lake	127.5	151.3
	Nitrite and Nitrate Nitrogen	
0-4.5	160.5	125.6
4.5-9.0	163.4	157.8
9.0-12.6	192.5	108.2
Total Lake	163	134.4

Depth (meters)	1968 (milligrams per liter)	1969
	Soluble Organic Carbon	
0-4.5	2.96	4.66
4.5-9.0	2.47	3.59
9.0-12.6	1.40	4.27
Total Lake	2.64	4.41
	Particulate Organic Carbon	
0-4.5	1.55	1.96
4.5-9.0	1.36	1.56
9.0-12.6	1.28	1.37
Total Lake	1.49	1.82
	Total Organic Carbon	
0-4.5	4.51	6.62
4.5-9.0	3.83	5.15
9.0-12.6	2.68	5.64
Total Lake	4.13	6.23

Depth (meters)	1968 (micrograms per liter)	1969
	Chlorophyll a	
0-4.5	13.3	8.5
4.5-9.0	12.7	6.1
9.5-12.6	7.5	4.9
Total Lake	12.5	7.5

Depth (meters)	1968 (Milliequivalents per liter)	1969
	Sodium	
0-4.5	0.225	0.159
4.5-9.0	0.247	0.151
9.0-12.6	0.226	0.189
Total Lake	0.229	0.153
	Potassium	
0-4.5	0.045	0.075
4.5-9.0	0.038	0.070
9.0-12.6	0.040	0.086
Total Lake	0.044	0.076
	Total Calcium	
0-4.5	1.76	2.78
4.5-9.0	1.85	2.78
9.0-12.6	1.99	3.72
Total Lake	1.80	2.84
	Magnesium	
0-4.5	0.527	0.483
4.5-9.0	0.545	0.496
9.0-12.6	0.461	0.614
Total Lake	0.540	0.489
	Chloride	
0-4.5	0.185	0.182
4.5-9.0	0.190	0.175
9.0-12.6	0.184	0.214
Total Lake	0.186	0.179
	Sulfate	
0-4.5	0.295	0.311
4.5-9.0	0.304	0.318
9.0-12.6	0.327	0.428
Total Lake	0.299	0.342

4. Nitrogen

Nitrogen determinations were made on Canadarago Lake samples collected at approximately 2 week intervals during 1968 and 1969. A variety of different forms of nitrogen were determined, including: ammonia, nitrate and nitrite, soluble and particulate-organic nitrogen and total organic nitrogen. The soluble and particulate organic nitrogen components were separated by means of vacuum filtration through a 47 mm diameter, 0.8 μ membrane filter. A summary of the average values of various forms of nitrogen at three different depths of the lake are shown in Table 6 (18).

5. Alkalinity

Total alkalinity determinations were made on Canadarago Lake samples collected at approximately 2 week intervals during 1968 and 1969. Alkalinity was determined by titration with mineral acid with time allowed for any suspended calcium carbonate to dissolve and for a stable end point to be attained (18). A summary of the average values of alkalinity at three different depths of the lake are shown in Table 6 (18).

6. Cations

Analyses for concentration of the cations of Ca, Mg, Na, K and Fe were made on Canadarago Lake samples collected at approximately 2 week intervals during 1968 and 1969. A summary of the average values of these cations at three different depths of the lake are shown in Table 6 (18).

7. Anions

Analyses for concentrations of the anions of chloride and sulfate were made on Canadarago Lake samples collected at approximately 2 week intervals during 1968 and 1969. A summary of the average values of these anions at three different depths of the lake are shown in Table 6 (18).

8. Trace Metals

Copper, zinc, cadmium and lead concentrations have been measured in Canadarago Lake water and sediments to characterize the heavy metal distribution at the sediment-water interface. Composite epilimnion and hypolimnion lake water samples were taken in August 1973 with a Van Dorn type sampler. Sediment cores and sediment supernatant water were obtained using a Kajak-Brinkhurst (KB) corer. Results showed very low concentrations of heavy metals in the lake water, with cadmium below 2 $\mu g \cdot l^{-1}$ and copper, zinc, and lead in the range of 5 to 20 $\mu g \cdot l^{-1}$. Sediment cadmium content was less than 10 $\mu g \cdot g^{-1}$ and lead less than 20 $\mu g \cdot g^{-1}$, while copper was in the range of 40-80 $\mu g \cdot g^{-1}$. Zinc content increased with sediment depth, ranging from 100 $\mu g \cdot g^{-1}$ in the 0-7.5 cm section of the core to 275 $\mu g \cdot g^{-1}$ in the 19.0-26.5 cm section. (All analyses of sediments are expressed on a dry weight basis.)

C. BIOLOGICAL

1. Phytoplankton

a. Chlorophyll

Chlorophyll, a concentration of Canadarago Lake, was determined from samples collected at approximately two week intervals during 1968 and 1969. The average concentration of chlorophyll a that was determined at three different depths, are shown in Table 6.

b. Primary Production

Primary production in Canadarago Lake has been calculated in terms of phosphorus and carbon for the period between May and December 1969, using an improved sedimentation trap designed by G. W. Fuhs (20). The contents of the traps were analyzed for total phosphorus and total carbon, as well as other parameters. Tables 7 and 8 present data for

Table 7 . CARBON PRODUCTION 1969 CANADARAGO LAKE

Sedimentation Period	C_T mg/l	C_T kg/ha	Trophogenic Zone* ha	S metric tons	Δ B mg/l	Trophogenic Zone $x10^6m^3$	Δ B metric tons	Δ B+S metric tons
5/9 - 23	57**	0.08**	611	49**	+1.55	40	+62	+111**
5/23 - 6/6	105	0.15	633	95	-0.72	34.2	-25	+70
6/6 - 6/20	101	0.14	633	89	[illegible]	34.8	+7	[illegible]
6/20 - 7/1	63	0.088	613	54	[illegible]	40	+20	+74
7/1 - 7/18	38	0.053	620	[illegible]	0	38	0	[illegible]
7/18 - 8/1	0	0	527	0	-0.10	35	-3.5	-3.5
8/1 - 8/20	?	?	600	?	+0.4	42	+17	est. -5
8/20 - 9/3	25	0.035	588	21	-0.8	45	-36	-15
9/3 - 9/15	54	0.076	600	46	-0.7	43	-30	+16
9/15 - 10/1	89	0.1246	634	79	+1.0	33	+33	+112
10/1 - 10/15	63	0.088	671	59	+1.1	27	+30	+89
10/15 - 10/31	389	0.5446	641	349	+3.4	33	+112	+461
10/31 - 11/12	37	0.052	631	33	-4.2	35	-147	-114
11/12 - 12/4	191	0.267	533	142	+2.7	37	+100	+242
			ave. 609					total 1166.5

$$\text{C production} = \frac{1166.5 \times 10^6 g}{609 \times 10^4 m^2 \cdot 195 \text{ days}} = 1.0 \text{ gC} \cdot m^{-2} \cdot \text{day}^{-1}$$

C_T total carbon, top minus bottom compartments of sedimentation trap, average value of replicates (see text)

* mid-depth area of trophogenic zone

S sedimentation, represented by C_T in sediment traps (see text)

Δ B change in biomass of C_T in lake at 1-3 meter depth

Δ B+S sum of biomass change and sedimentation

** correction obtained from C/P atomic ratios using P values in epilimnion and sediment

Table 8 . PHOSPHORUS PRODUCTION 1969 CANADARAGO LAKE

Sedimentation Period	P_T µg/l	P_T g/ha	Trophogenic Zone* ha	S kg	Δ B µg/l	Trophogenic Zone $x10^6m^3$	Δ B kg	Δ B+S kg
5/9 - 23	552	773	611	472	-6.2	40	-248	+224
5/23 - 6/6	1077	1508	633	954	+8.4	34.2	+287	+1241
6/6 - 6/20	525	735	633	[illegible]	-9.7	34.8	-338	+127
6/20 - 7/1	473	662	613	406	+2.2	40	+88	+494
7/1 - 7/18	81	113	620	70	+4.4	38	+167	+237
7/18 - 8/1	628	879	527	[illegible]	+2.5	35	+88	+551
8/1 - 8/20	460	644	600	387	-13.3	42	-559	-172
8/20 - 9/3	149	209	588	[illegible]	+5.8	45	+261	+384
9/3 - 9/15	314	440	600	[illegible]	+1.5	43	+65	+330
9/15 - 10/1	111	155	634	[illegible]	+24.3	33	+802	+990
10/1 - 10/15	0	18	671	[illegible]	-19.7	27	-532	-520
10/15 - 10/21	3990	5586	641	3580	+14.4	33	+475	+4055
10/21 - 11/12	2042	2858	631	[illegible]	-1.7	35	-60	+1744
11/12 - 12/4	[illegible]	4553	533	24.7	-8.2	37	-303	+2124
			ave. 609					total 11719

$$\text{P production} = \frac{11719 \text{ kg P}}{609 \text{ha} \cdot 195 \text{ days}} = \frac{11719 \times 10^6 \text{ mg}}{609 \times 10^4 m^2 \cdot 195 \text{ days}} = 10 \text{ mgP} \cdot m^{-2} \cdot \text{day}^{-1}$$

$$\text{C production on C/P atomic ratio basis} = \frac{1229.9 \times 10^6 g}{609 \times 10^4 m^2 \cdot 195 \text{ days}} = 1.0 g^C \cdot m^{-2} \cdot \text{day}^{-1}$$

P_T total phosphorus, top minus bottom compartments of sedimentation trap, average value of replicates (see text)

* mid-depth area of trophogenic zone

S sedimentation, represented by P_T in sediment traps (see text)

ΔB change in biomass of P_T in lake at 1-3 meter depth

ΔB+S sum of biomass change and sedimentation the calculation of primary production via total phosphorus (P_T) and particulate organic carbon (C_T) determinations. The biomass (B) of phosphorus or carbon in the lake is that amount retained by a 0.8 μ membrane filter. Total phosphorus production yielded 10 mg P $\cdot$ m^{-1}. Total carbon production determined from the C/P atomic ratios and experimentally via the sedimentation traps yielded 1.0 g C $\cdot$ m^{-2} $\cdot$ day^{-1}.

c. Algal Assays

A long term bioassay with lake water collected from Canadarago Lake in May 1969, and phosphate additions (but no additional inoculum) showed a very clear response to phosphates. The increase in biomass with 1 mg $\cdot$ l^{-1} P added as compared with the effect of 100 μg $\cdot$ l^{-1} P indicates a plentiful supply of nitrogen (including organically bound N) and minor elements.

Short-term bioassay studies with ^{14}C were run on two days in 1968. Additions of nutrients were made to produce the identical final concentrations in both short-term and long-term experiments and included K_2HPO_4, $NaNO_3$, Fe as Fe^{++} - EDTA chelate, chelator alone, unchelated trace metal mix and vitamin mix. All additions showed stimulation or inhibition except nitrate additions which were always without effect.

d. Identification and Count

Phytoplankton from Canadarago Lake were sampled and quantified from 1968 to 1973, with the exception that during 1970 only qualitative analyses were performed. Major plankton organisms from the standpoint of number and size were chosen and identified to genus or species.

Prior to 1968 there had been massive blooms of Oscillatoria prolifica(Grev.) Gomont in the lake. Such blooms recurred in 1972 and 1973. The most commonly occurring predominant algae quantified between 1968 and 1973 are shown in Table 9. During the summer of 1935 slight shore blooms were noted in Canadarago Lake but there was never a bloom over the entire lake. The shore blooms consisted of the blue-green algae Anabaena and Coelosphaerium (1).

Table 9. MOST COMMONLY OCCURRING PREDOMINANT ALGAE IN CANADARAGO LAKE, 1968-1973

CYANOPHYTA

Anacystis incerta Drouet and Daily
Coelosphaerium naegelianum Unger

CHLOROPHYTA

Sphaerocystis schroeteri Chodat

CHRYSOPHYTA

Cyclotella comta var. radiosa Grun.
Dinobryon divergens Imhof
Stephanodiscus niagarae Ehrenberg

PYRRHOPHYTA

Ceratium hirundinella (O.S. Muell.) Dujardin
Cryptomonas ovata Ehrenberg

2. Zooplankton

Zooplankton sampling was initiated in August 1972 as part of the Canadarago Lake Eutrophication Project with samples being collected every two weeks during the ice free season (21,22,23).

a. Identification and Count

Peak abundance for zooplankton during the fall of 1972 was on October 14, with 179,309 organisms $\cdot$ m^{-3}. The peak in zooplankton was due mainly to *Eubosmina coregoni* (74,944 organisms $\cdot$ m^{-3}) which comprised 42 percent of the total. Along with the *E. coregoni*, the cladocerans made up 70 percent of the population. From July 6 to July 19, 1973 there was a drastic change in the zooplankton population from 76 percent composition of rotifers on July 6 to 16 percent composition of rotifers on July 19; while at the same time, cladocerans made up 13 percent of the total composition on July 6 and changed to 75 percent composition of the total number of zooplankton on July 19. There have been 20 species of cladocerans, six species of copepods and six species of rotifers identified from Canadarago Lake. The main pulse of zooplankton occurred in early July with 311,677 organisms $\cdot$ m^{-3}. The spring pulse, typical of many lakes in April and May was not present or was delayed possibly due to the heavy bloom of *Oscillatoria prolifica* present in the lake until mid July. Changes in the zooplankton population from the 1935 survey (1) appear to be negligible.

3. Bottom Fauna

Monthly benthos samples are collected from seven stations representing different water column depths and substrates. Samples during a one year period (Sept. 72-Sept. 73) of benthic organisms from the combined substrates were comprised mostly of chironomic larvae. The percent composition of chironomids ranged from a low of 21.5 percent in September 1973 to a high of 66.2 percent in November 1972. The only other group of invertebrates that were numerically important in combined substrates were the oligochaetes. The percent composition of these species ranged from a low of 26.4 percent in November 1972 to a high of 67 percent in July 1963. The peak in the abundance of bethnic invertebrates occurred in March 1973 and was due almost exclusively to chironomide larvae which, after peak abundance in March, gradually decreased until mid July.

As depth increases, difference in abundance of benthic invertebrates becomes apparent. With a depth of 3 m (10 ft) or greater, the numbers of oligochaetes decrease as depths increase; however, the percent composition of oligochaetes appears to remain consistent. The chironomid numbers and percent composition decreased with increasing depth, while at depths of 4.6 m (15 ft) or greater, the numbers and percent composition of *Chaoborus* increased as depth increases. In general, the total number of benthic fauna decreased as depth increased.

In 1935 the major organisms below 6 m (20 ft) were *Chaoborus* and large chironomid larvae with no mention of the presence of oligochaetes (1). The 1972 survey samples contained large quantities of empty mollusc shells indicating that large numbers of clams and snails were at one time present in this lake. Harman (24) reports that Canadarago Lake once supported dense populations of mollusks that are now severely depleted.

4. Fish

In 1972 a detailed study of the fisheries of Canadarago Lake was initiated by Cornell University's Department of Natural Resources (23,25). This effort is being conducted to measure changes in the structure and dynamics of fish populations in a highly eutrophic lake following a reduction in cultural eutrophication with the objective of developing fish management techniques applicable to lakes undergoing nutrient control and examine nutrient control as a fish management tool.

Yellow perch are the most abundant fish in Canadarago Lake. Other abundant species are golden shiner, spottail shiner, white sucker, Johnny darter, black crappie and brown bullhead. Principal game species are smallmouth bass, chain pickerel and largemouth bass. Smelt and black crappie, recently introduced in the late 1960's, have rapidly expanded their populations. Walleye, American eel, banded killifish, bridle shiner, satinfin shiner, blackchin shiner and blunt-nose shiner have either decreased greatly in numbers present or are no longer present. New species reported for the lake are bluegill sunfish, brook trout, burbot, shortnosed redhorse, fathead minnow and stoneroller.

Surveys of the Canadarago Lake fish populations during a period of increasing eutrophication from 1935 to 1972 indicate three species maintained their dominance throughout the period. During the 1935 Biological Survey (1) the golden shiner and yellow perch were the most abundant forage fish and the chain pickerel the most predominant predator. Subsequent surveys in 1958, 1964 and 1969 (26) found the same species were predominant.

Table 10 lists the species of fish that have been found in Canadarago Lake. Historical records are not adequate to evaluate possible changes in abundance of all of the species reported from this lake.

5. Bacteria

No bacteriological studies of major significance have been undertaken in Canadarago Lake as yet. Gassing (release of marsh gas) from sediments, indicative of methane fermentation, can be observed in the northern half of the lake, increasing in intensity from mid-lake to the northern shore. The phenomenon is observed when an anchor is dropped from a boat during surveys. Spontaneous gas release has not been observed, e. g. as gas accumulation in the inverted reference compartment of sediment traps. A preliminary study of sulfate reduction (27) showed organic matter, not sulfate, to be the limiting factor in bacterial sulfate reduction in the sediments. L. W. Wood (28) found indication of oxidation of Rhodamine B dye in the sediments, presumably by microbes.

Table 10. CANADARAGO LAKE FISH SPECIES

Brown trout--*Salmo trutta*

Chain pickerel--*Esox niger*

Largemouth bass--*Micropterus salmoides*

Smallmouth bass--*M. dolomieui*

Yellow perch--*Perca flavescens*

Common sunfish--*Lepomis gibbosus* (pumpkinseed)

Redbreast sunfish--*L. auritus*

Rock bass--*Ambloplites rupestris*

Brown bullhead--*Ictalurus nebulosus*

American eel--*Anguilla rostrata*

Johnny darter--*Etheostoma nigrum olmstedi*

Banded killifish--*Fundulus diaphanus*

Bluntnose minnow--*Pimephales notatus*

Golden shiner--*Notemigonus crysoleucas*

Cutlips minnow--*Exoglossum maxillingua*

Common shiner--*Notropis cornutus*

Spottail shiner--*N. hudsonius*

Bridle shiner--*N. bifrenatus*

Satinfin shiner--*N. analostanus*

Blackchin shiner--*N. heterodon*

Creek chub--*Semotilus atromaculatus*

Pearl dace--*S. margarita*

Blacknose dace--*Rhinichthys atratulus*

Longnose dace--*R. cataractae*

Carp--*Cyprinus carpio*

Common sucker--*Catostomus commersoni* white sucker

Creek chubsucker--*Erimyzon oblongus*

Rainbow smelt--*Osmerus mordax*

Walleye--*Stizostedion vitreum vitreum*

Northern pike--*Esox lucius*

Muskellunge--*Esox masquinongy*

Burbot--*Lota lota*

Brook trout--*Salvelinus fontinalis*

Black crappie--*Pomoxis nigromaculatus*

Bluegill sunfish--*Lepomis machrochirus*

Fallfish--*Semotilus corporalis*

Shorthead redhorse--*Moxostoma macrolepidotum*

Stoneroller--*Campostoma anomalum*

River chub--*Nacomis micropogon* (Cope)

6. Bottom Flora

No bottom flora studies of major significance have been undertaken in Canadarago Lake as yet. Algae attached to rocks located near shore, October 1973, included:

CHRYSOPHYTA	CHLOROPHYTA
Navicula sp.	*Spirogyra* sp.
Cymbella sp.	*Oedogonium* sp.

7. Macrophytes

Canadarago Lake supports emergent, floating, and submersed aquatic macrophytes around its periphery. The main species observed between 1968 and 1973 are shown in Table 11.

Table 11. CANADARAGO LAKE MACROPHYTE SPECIES, 1968-1970

Emergent

1. Softstem bulrush - Scirpus validus Vahl.
2. Harstem bulrush - Scirpus acutus Muhl.
3. Pickerelweed - Pontederia cordata L.
4. Narrow-leaved cattail - Typha angustifolia L.
5. Bur reed - Sparganium eurycarpum Engelm.

Floating

6. Yellow water lily - Nuphar variegatum Engelm.
7. (White) water lily - Nymphaea odorata Ait.

Submersed

8. Narrow-leaved pondweed - Potamogeton spp.
9. Water milfoil - Myriophyllum sp.
10. Waterweed - Anacharis canadensis (Michx.) Planchon
11. Coontail - Ceratophyllum demersum L.
12. Curly-leaf pondweed - Potamogeton crispus L.

The plants ranking highest in lake surface area coverage in 1968 and 1969 were the two bulrush species and yellow water lily. The location of greatest abundance of emergent plants is at the southwestern end of the lake where hardstem bulrush, yellow water lily and pickerel weed predominate. A great increase in the amount of submersed vegetation occurred between 1969 and 1973. Water milfoil, curly-leaf pondweed, narrow-leaved pondweed, and water weed predominated. The submersed plants existed around almost the entire periphery in water 3 m (10 feet) or less in depth. Areas of greatest density were primarily in the northeastern end, where milfoil was extremely abundant, and secondarily the southwestern end.

NUTRIENT BUDGET SUMMARY

A. ESTIMATION OF INPUTS

1. Waste Discharges

There are no significant industrial waste discharges in the Canadarago Lake Basin. There are two significant sources of sanitary waste, the village of Richfield Springs sewage system and the unsewered homes, mostly summer cottages, along the lake shore.

As noted elsewhere, the village of Richfield Springs is served by a combined sewer system which discharges into Ocquionis Creek about 840 m (2750 ft) upstream from Canadarago Lake. Until 1973, the village had a primary wastewater treatment plant. Detailed estimates of the major nutrients discharged from this source were made by direct measurement of the plant effluent, the difference between upstream and downstream samples and calculation from per capita contributions. The details of these estimates are presented elsewhere (16). The results for those major nutrient studies are summarized in Table 12.

Estimation of chemical contributions to the lake from the unsewered homes on the lake shore is especially difficult. During the summer months, about 1300 people occupy summer cottages around the lake shore and are served by septic tanks and leaching fields (12). In 1969 a sanitary survey of these systems revealed that 24 percent of the septic tank systems had some sort of direct discharge to the lake, bypassing the leaching fields (13).

Table 12. CANADARAGO LAKE CHEMICAL INPUT FROM RICHFIELD SPRINGS SEWAGE TREATMENT PLANT

	$Kg \cdot yr^{-1}$	$gm \cdot m^{-2}$ of lake surface $\cdot yr^{-1}$
K^{+}	4150	0.55
Mg^{++}	8710	1.15
Cl^{-}	20,030	2.64
$NO_3^{-}+NO_2^{-}-N$	33	4.39
NH_4-N	3563	0.47
N_{os}	1771	0.23
N_{op}	200	0.03
N_t	5567	0.73
P_{st}	2310	0.30
P_{pt}	343	0.05

Table 13. CANADARAGO LAKE CHEMICAL INPUT FROM SEPTIC TANK AND LEACHING FIELD SYSTEMS

	$Kg \cdot yr^{-1}$	$gm \cdot m^{-2}$ of lake surface $\cdot yr^{-1}$
N_t	2220	0.29
P_t	121	0.02

Assuming that any phosphorus entering a septic tank leaching field was retained in the field, and that none of the nitrogen was retained, Table 13 was constructed. No estimate of inputs of other chemicals were made.

2. Land Runoff

The Candarago Lake Eutrophication Study has shown that stream loadings with soluble mineral species derived mainly from bedrock may be estimated by the regression method. Errors become larger in the case of constituents that are subject to or products of biological processes or that are in particulate form and therefore subject to sedimentation and sudden dislocation during periods of high flow.

Utilizing measured stream flows and concentrations, regression analysis was employed to estimate the chemical runoff and lake loading from land. Details of the regression models are described elsewhere (29). The results of these estimates are shown in Tables 14, 15, and 16.

3. Precipitation

Because of the large ratio of the watershed area to that of the lake surface (23:1), the contribution of chemicals from precipitation is very small in the lake, in most cases less than two percent, and can be neglected.

Table 14. HYDROLOGICAL DATA FOR COMPUTATIONS OF CHEMICAL LOADINGS FROM CANADARAGO LAKE TRIBUTARIES

after Hetling and Sykes (16)

Name	Drainage Area		Measured Annual Flow			Measured Annual Flow Over Drainage Area	Contributed to Lake
	mi^2	ha	cfs	m^3/s	$10^6 m^3/yr$	m/yr	m/yr
Herkimer Creek	11.9	3077	26.1	1.020	23.3	0.759	3.03
Hyder Creek	10.6	2741	13.2	0.423	11.8	0.431	1.53
Mink Creek	10.5	2715	14.6	0.581	13.1	0.481	1.70
Ocquionis Creek	19.8	5120	24.1	0.829	21.6	0.421	2.80
gauged total	52.8	13653	78.0	2.853	69.8	0.511	9.06
Oaks Creek	67.4	17427	101	5.144	90.4	0.519	11.74

Lake area: 759 ha.

Table 15. CHEMICAL RUNOFF PER HECTARE OF WATERSHED PER YEAR[1]

	Unit	Herkimer	Hyder	Mink	Ocquionis (except STP)	Gauged Watershed
Na^+	kg	23.4	16.6	28.3	14.3	19.5
K^+	kg	11.4	11.6	11.6	7.8	10.1
Mg^{++}	kg	21.1	14.5	24.1	23.1	21.1
Ca_t	kg	649	617	701	581	629
Fe_t	gm	1.27	0.92	0.92	0.76	0.94
Cl^-	kg	25.5	22.5	42.3	20.7	26.5
$SO_4^=$	kg	222	126	196	197	188
$NO_3^-+NO_2^--N$	gm	4350	5590	5970	4170	4860
$NO_2^- -N$	gm	6.11	5.70	7.90	26.0	13.8
NH_4^+-N	gm	477	309	444	425	417
N_{os}	gm	1130	861	2260	1300	1360
N_{op}	gm	636	344	424	282	403
N_t	gm	6590	7100	9100	6180	7040
P_o	gm	13.9	8.15	23.4	24.7	18.7
P_{st}	gm	66.7	23.6	47.7	57.2	50.7
P_{pt}	gm	105	134	125	188	133
C_{os}	kg	33.6	13.5	20.1	28.0	24.8
C_{op}	kg	6.18	4.22	4.80	3.29	4.43
ΣCO_2	kg	695	572	655	538	603

1 K^+, Mg^{++}, Cl^-, $NO_3^-+NO_2^--N$, NH_4^+-N, N_{os}, N_{op}, N_t, P_{st}, and P_{pt} inputs were estimated by summation of the product of the measured daily flow and the concentration from the regression of flow and concentration for that particular tributary. The remaining elements were estimated by multiplying the log mean concentration by the average flow.

Table 16. CANADARAGO LAKE CHEMICAL INPUT FROM LAND RUNOFF PER SQUARE METER OF LAKE SURFACE PER YEAR

	Units	Herkimer	Hyder	Mink	Ocquionis	Total Gauged Watershed	Estimated Ungauged Watershed	Total Land Runoff
Na^+	gm	9.51	5.98	10.1	9.61	35.2	9.8	44.9
K^+	gm	4.62	4.18	4.14	5.24	18.2	5.1	23.2
Mg^{++}	gm	3.56	5.24	8.61	15.7	38.1	10.6	48.7
Ca_t	gm	263	223	251	392	1129	314	1440
Fe_t	mg	0.51	0.33	0.33	0.51	1.69	0.47	2.16
Cl^-	gm	10.3	8.12	15.2	14.0	47.6	13.2	60.8
$SO_4^=$	gm	90.0	45.4	70.1	132	338	94	432
$NO_2^-+NO_3^--N$	mg	1760	2020	2140	2820	8730	2430	11200
NO_2^--N	mg	2.48	2.06	2.83	17.5	24.9	6.9	31.8
NH_4^+-N	mg	193	117	159	287	751	209	959
N_{os}	mg	458	311	809	875	2453	682	3135
N_{op}	mg	257	124	151	190	724	201	925
N_t	mg	2670	2560	3260	4170	12700	3520	16200
P_o	mg	5.64	2.94	8.37	16.7	33.6	9.3	43.0
P_{st}	mg	27.0	8.52	17.1	38.6	91.2	25.3	116
P_{pt}	mg	42.6	48.4	44.7	126.8	262.5	73.0	336
C_{os}	gm	13.6	4.86	7.19	18.8	44.5	12.3	56.9
C_{op}	gm	2.51	1.52	1.72	2.22	7.97	2.21	10.2
ΣCO_2	gm	281	207	234	363	1080	301	1390

4. Groundwater

During our study 78.2 percent of the watershed, including all significant tributaries, was gauged. An estimate of nutrient contribution from the ungauged areas was achieved by assuming that the runoff for these areas would be equal to the average of the area drained by the tributaries, not counting the wastewater treatment plant effluent (16). The total nutrient input from ungauged sources was thus calculated by dividing the gauged land runoff by 0.782. This is groundwater and surface runoff, in part routed through small and ephemeral streams.

B. PHOSPHORUS

Utilizing the monthly average loadings from the gauged sources and flows from the hydrographs that had been generated, nutrient budgets for phosphorus and nitrogen were calculated (16). Phosphorus data have been given the greatest attention because it was determined that the algal-limiting nutrient in the lake was phosphorus (30).

On an annual basis, the principal source of phosphorus in the watershed was the village of Richfield Springs which contributed 44.1 percent of the total annual input (16). If computed for the growing season, June through September, the village's share of the phosphorus input rises to 66.4 percent. These figures are equal to about 4.8 g (0.17 oz) P $\cdot$ day^{-1} $\cdot$ $capita^{-1}$ and include commercial as well as domestic sources. In determining the contribution of phosphorus from lake shore cottages and trailers, it was assumed that only failing septic tank systems with direct discharge into the lake contributed phosphorus. In 1969, 24.4 percent of the septic tanks, servicing 317 people on the lake, had some type of direct discharge into the lake (13). Using 2.9 g (0.1 oz) P $\cdot$ day^{-1} $\cdot$ $capita^{-1}$ (31, 32) for phosphorus production and an average residence time of 151 days (12), the annual phosphorus input from the cottages were estimated at 140 kg (309 lbs) P $\cdot$ $year^{-1}$, or 2.3 percent of the annual total.

The gauged tributaries carried 42.4 percent of the total phosphorus input to the lake for an average areal rate of 0.187 kg $\cdot$ yr^{-1} $\cdot$ ha^{-1} (0.167 lbs $\cdot$ yr^{-1} $\cdot$ $acre^{-1}$) (16). Applying the same rate for the area that did not have gauged tributaries yielded another 570 kg (1257 lbs) P $\cdot$ yr^{-1} for a total of 3120 kg (6880 lbs) P $\cdot$ yr^{-1}, 51.8 percent of the total. During the growing season, when the stream flows became very small, the streamborne phosphorus was only 23.5 percent of the total summer input.

Phosphorus inputs caused by rainfall and dustfall were estimated from literature values. The reported range was about 0.206 to 0.612 kg PO_4 $\cdot$ yr^{-1} $\cdot$ ha^{-1} (0.184 to 0.546 lbs PO_4 $\cdot$ yr^{-1} $\cdot$ $acre^{-1}$) (33), which suggests an atmospheric contribution of about 100 kg (220.5 lb) P $\cdot$ yr^{-1} onto the lake surface itself, less than 2 percent of the total. The results of the phosphorus input data are shown in Table 17.

Similar estimates were made for soluble phosphorus alone. These calculations are summarized in Table 17. Because the wastewater phosphorus

Table 18. ESTIMATED TOTAL NITROGEN INPUTS TO CANADARAGO LAKE

April 15, 1969 - April 14, 1970

Source	Total Value			Growing Season Values*	
	kg/yr	Percent of Total Value	kg	Percent of Annual Value of Source	Percent of Growing Season Value
Village of Richfield Springs	5730	4.2	1920	33.5	13.3
Lake Shore Dwellings	2020	1.5	1630	80.7	11.3
Sub Total	7750	5.7	3550	45.8	24.6
Gauged Tributaries	97350	71.3	7660	7.9	53.0
Ungauged Tributaries	27100	19.9	2130	7.9	14.8
Sub Total	124450	91.2	9790	7.9	67.8
Rainfall	4200	3.1	1100	26.2	7.6
Total Input	136400	100.0	14440	10.6	100.0
Oaks Creek Output	82500	60.5	10400	12.6	71.7
Net Accumulation and Dissipation	53900	39.5	4040	7.6	28.3

*June 1, 1969 through September 30, 1969

Table 17. ESTIMATED SOLUBLE, PARTICULATE AND TOTAL PHOSPHORUS INPUTS TO CANADARAGO LAKE, APRIL 15, 1969 - APRIL 14, 1970

Source	Total P (kg/yr)	Percent of Total P	Soluble P kg/yr	Percent of Total P	Percent of Soluble P	Particulate P (kg/yr)	Percent of Total P	Percent of Particulate P
Village of Richfield Springs	2660	44.1	2310	38.4	68.9	350	5.8	13.6
Lake Shore Dwellings	140	2.3	121	2.0	3.6	19	.3	.8
Sub Total	2800	46.4	2431	40.4	72.5	369	6.1	14.4
Gauged Tributaries	2550	42.4	723	12.0	21.5	1827	30.3	71.2
Ungauged Tributaries	570	9.4	200	3.3	6.0	370	6.1	14.4
Sub Total	3120	51.8	923	15.3	27.5	2197	36.5	85.6
Rainfall	100	1.7						
Total Input	6020	100.0	3354	55.7	100.0	2566	42.6	100.0
Oaks Creek Output	4660	77.5	1740	28.9	52.0	2920	48.5	113.8
Net Accumulation	1360	22.5	1614	26.8	48.0	- 354	- 5.9	- 13.8

is about 87 percent soluble, whereas the streamborne phosphorus is only 28 percent soluble, the wastewater contribution to the soluble phosphorus inputs is very large, amounting to 72.5 percent on an annual basis and 88.6 percent during the growing season. In addition, Table 17 includes a value for particulate phosphorus. Here the soluble phosphorus has been subtracted from the total phosphorus to yield the value for particulate phosphorus. The output of particulate phosphorus is larger than the input. This may be misleading, however it is assumed that algae within the lake converted some of the soluble phosphorus to an insoluble form which accounts for a larger output than input of particulate phosphorus.

C. NITROGEN

The gauged nitrogen contribution from various sources was calculated in a manner similar to that for the phosphorus contributions. The wastewater treatment plant loadings for soluble organic nitrogen were deduced from Ocquionis Creek data. The remaining wastewater data were based on raw wastewater analyses (16).

About 62 percent of the wastewater nitrogen was in the form of ammonia, and another 31 percent was present as soluble organic nitrogen (16). In contrast, about two-thirds of the nitrogen in the tributaries was either in the form of nitrite or nitrate, therefore, there are qualitative as well as quantitative differences among the nitrogen sources.

Estimates of the different nitrogen sources are given in Table 18. The village contribution is equivalent to about 10.3 g (0.363 oz) $N \cdot day^{-1} \cdot capita^{-1}$ and seems to be a result of domestic activities only. The same per capita rate was taken for the lake shore residences. This time it was assumed that the nitrogen was not retained in the septic tank leaching fields, so the contributing population was taken as the entire lake shore dwelling population of 1300 people. The data indicated that human wastes were a minor source of nitrogen input to the lake and were the same order of magnitude as rainfall and dustfall. The atmospheric rate was taken to be 1.50 kg $N \cdot yr^{-1} \cdot ha^{-1}$ (1.34 lbs $N \cdot yr^{-1} \cdot acre^{-1}$) (33, 34).

The principal sources of nitrogen were the tributary streams, which accounted for approximately 91.2 percent of the annual input. The predominance of the tributary streams is marked, even during the summer months when over two-thirds of the nitrogen input is transported by streams. The average annual nitrogen loading carried by these streams was 7.10 kg $N \cdot yr^{-1} \cdot ha^{-1}$ (6.34 lb $N \cdot yr^{-1} \cdot acre^{-1}$).

D. MISCELLANEOUS ELEMENTS

Summary data for chlorides, magnesium and potassium that were determined during the study period, April 15, 1969 through April 14, 1970 are given in Table 19 (16). In each case, the contribution of these materials from wastewater were relatively minor.

Table 19. GAUGED MISCELLANEOUS INPUTS AND OUTPUTS TO CANADARAGO LAKE

April 15, 1969 through April 14, 1970

a. Potassium

Time Period	Quantity of Potassium ($kg \cdot day^{-1}$)				
	Input				Output
	Ocquionis Creek	Mink Creek	Hyder Creek	Herkimer Creek	Oaks Creek
April 15-30, 1969	356	215	213	184	2544
May 1969	106	72.9	62.9	72.3	1084
June 1969	104	81.0	76.1	84.5	689
July 1969	37.9	25.2	16.6	30.0	227
August 1969	13.3	9.4	5.8	11.1	74.3
September 1969	6.0	4.7	2.6	6.3	18.6
October 1969	14.1	14.5	6.5	21.9	142
November 1969	131	120	113	176	845
December 1969	125	116	102	125	865
January 1970	82.7	58.5	49.7	64.4	536
February 1970	128	89.7	98.0	99.3	871
March 1970	85.4	67.9	57.7	99.6	497
April 1-14, 1970	636	574	788	632	3018
Average	109	86.1	86.9	96.1	713

b. Magnesium

Time Period	Quantity of Magnesium ($kg \cdot day^{-1}$)				
	Input				Output
	Ocquionis Creek	Mink Creek	Hyder Creek	Herkimer Creek	Oaks Creek
April 15-30, 1969	909	431	265	343	3390
May 1969	343	160	84.5	132	1550
June 1969	335	175	100	156	1020
July 1969	150	60.0	24.1	55.4	370
August 1969	65.0	23.9	8.89	204	133
September 1969	34.9	12.9	4.31	11.6	38.1
October 1969	68.4	36.0	10.1	40.3	240
November 1969	404	251	145	326	1230
December 1969	393	244	133	232	1270
January 1970	294	131	68.2	119	817
February 1970	402	194	104	184	1270
March 1970	289	149	77.9	147	763
April 1-14, 1970	1430	1050	900	1180	3940
Average	324	179	109	178	1020

c. Chloride

Time Period	Quantity of Chloride ($kg \cdot day^{-1}$)				
	Input				Output
	Ocquionis Creek	Mink Creek	Hyder Creek	Herkimer Creek	Oaks Creek
April 15-30, 1969	985	729	413	412	5690
May 1969	273	297	114	157	2430
June 1969	271	320	140	187	1540
July 1969	926	121	27.7	65.5	508
August 1969	30.6	52.1	9.06	24.0	166
September 1969	13.1	30.1	5.65	13.5	41.6
October 1969	32.5	76.0	10.3	47.7	317
November 1969	345	443	215	394	1890
December 1969	327	435	190	279	1940
January 1970	210	249	87.9	142	1200
February 1970	334	355	142	220	1990
March 1970	216	278	104	176	1110
April 1-14, 1970	1820	1630	1670	1440	6750
Average	291	315	169	215	1600

A. LIMNOLOGICAL CHARACTERISTICS

Canadarago Lake shares many of its features with its western neighbors, New York's Finger Lakes, and with many other lakes located between 40 and 60 degrees latitude.

The climate at 43°N and 75°W is neither humid nor arid and in this respect resembles many areas in northern to southeastern Europe. The region is somewhat sheltered from the Atlantic Coast but is readily exposed to rain and snowstorms originating in the Gulf of Mexico and certainly exposed to those from the St. Lawrence Great Lakes. At 43°N on the North American Continent, winters are relatively severe and comparable to Europe at 60°N or, in eastern Europe, in the fifties. Ice cover on Canadarago lasts from December through April and reaches a thickenss of $\frac{1}{2}$ m (20 in.). Summers are as warm as in comparable latitudes of Europe, causing considerable warming to the bottom of lake of Canadarago's depth, 12.8 m (42 ft). Correspondingly, the annual cycle of Canadarago is characterized by a very short period of spring overturn which may be preceded by an algal bloom developing under the ice. Stratification proceeds in a typical manner, and since warming at the bottom is substantial, 16°C (28.8°F), breaks down early (in September) when the water is still warm enough to support considerable primary production. After a prolonged cooling period in autumn, winds, to which Canadarago is well exposed, may not permit the formation of an ice cover until the entire lake is cooled down to somewhere between 4° and 0°C (39 and 32°F), and the stability of winter stratification varies accordingly.

Located in hilly terrain, the morphometry of the lake is not unusual (Table 4). Hydrologic conditions and the size of the watershed provide for a mean retention time of 217 days.

The basic chemistry of Canadarago Lake is summarized in Figure 8. Calcium and bicarbonate ions predominate, followed by magnesium and sulfate ions. Sodium and chloride ions are nearly matched. Sulfur springs in the vicinity of the lake account for part of the sulfate and may account for the fact that in spite of the eutrophy of the lake, bacterial sulfate reduction is not limited by sulfate but by the organic carbon source (27). The calcium balance of the lake is such that extensive precipitation of this element must occur, particularly during productive periods in summer. Epilimnion calcium concentrations are about 0.5 meq $\cdot$ l^{-1} lower than those found in the tributaries. Similarly, iron is precipitated and presumably plays an important role in the ultimate deposition of phosphate.

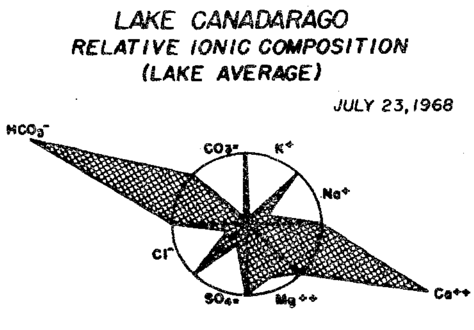


Figure 8. Canadarago Lake, relative ionic composition. Diagram after Maucha shows shaded areas proportional to concentrations. Reprinted from 30.

Conditions for primary productivity are favorable and until recently were enhanced by substantial inputs of nutrients from untreated sewage. Although wind exposure and basin shape would suggest excellent mixing, the lake has exhibited, from time to time, a slight but significant gradient in characteristics such as chlorophyl and particulate phosphorus and other parameters expressing biomass, indicating greater productivity in the northern part which is not only more shallow but also received the discharge of untreated sewage. In agreement with this observation, the sediments in the northern part have greater organic content and, upon incubation under aerobic conditions, release soluble phosphorus in greater amounts.

The principal limiting nutrient in the lake is phosphorus as indicated by:

1. The atomic ratios of the major nutrients in the tributaries (Table 20).

2. The disappearance of reactive phosphates from the epolimnion during most of the growing season (Figure 9).

3. The relative chemical composition of the plankton (C:N:P ratios, Figure 10).

4. Long and short-term bioassay (see Section IV).

ATOMIC RATIOS	C : N : P (P=1)
Herkimer Creek	2543 : 85 : 1
Hyder Creek	[illegible] : 92 : 1
Mink Creek	[illegible] : 61 : 1
Ocquionis Creek	2122 : 52 : 1
Ocquionis + STP effluent	531 : 19 : 1 [illegible] [illegible] [illegible]
(growth requirement:	106 : 16 : 1)

Table 20. Tributaries to Canadarago Lake atomic ratios C:N:P (P=1). Reprinted from 30.

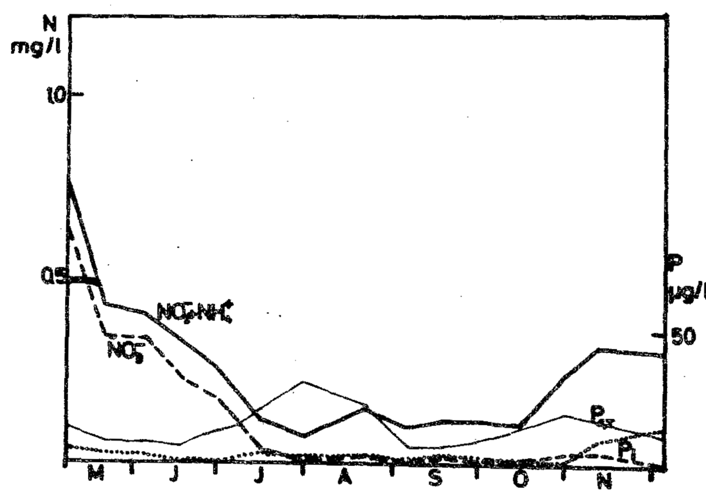


Figure 9. Soluble phosphates and forms of nitrogen in Canadarago Lake, 1969. Reprinted from 30.

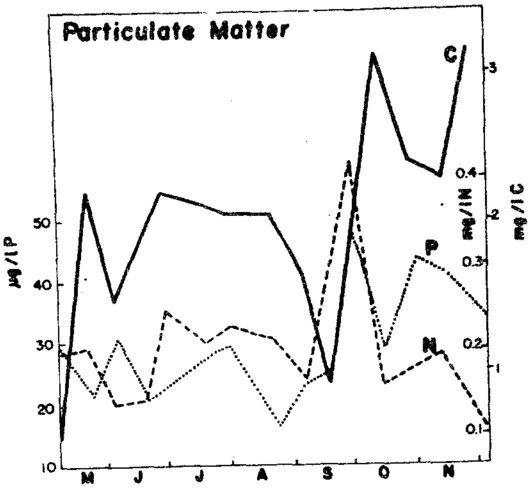


Figure 10. Elementary composition of particulate matter in Canadarago Lake, summer 1969. Ordinates scaled according to the atomic ratios C:N:P = 106:16:1. Reprinted from 30.

Nitrogen is present as nitrate except in late summer when ammonia is the only available form (except organically bound nitrogen) but both are found in concentrations that can be considered higher than limiting. Among the Cyanophyceae, the Chroococcales and Oscillatoriales are predominant, and one possible nitrogen fixing form, Aphanizomenon, occurred during short periods which were definitely not caused by nitrogen depletion. Carbon dioxide depletion can occur in a spotty fashion during summer afternoons. The thesis that such a condition favors the development of blue-green algae is not generally supported by observations in Canadarago Lake. Blue-green algal blooms do occur in summer, but the same species were found to produce blooms in winter, in early spring, and immediately after fall overturn. Simulation of growth by the availability of phsophorus in the presence of high concentrations of CO_2 is a more likely explanation of these blooms.

Silicon depletion may affect species composition in Canadarago Lake. Silicon has been a neglected element in the earlier studies on which this report is largely based. Data on this element are now being gathered. Iron appears to become limiting at times when the solubility of the element is affected by high pH which in turn is caused by phosphate eutrophication. Other forms of nutrient limitation were looked for but were not discovered.

A strange and thus far unexplained phenomenon is the reoccurrence of blooms of Oscillatoria prolifica, a red-colored member of the blue-green algae, in summer and in winter from 1972 until 1973-74. This alga was predominant also until 1967-68, and was the cause for many citizen's complaints. In the intervening years, the algae was scarce and never developed a bloom. The effect of this bloom on the food chain deserves study because much grazed-upon populations of green algae are virtually absent when O. prolifica blooms, and the collapse of O. prolifica blooms is followed by periods of great clarity of the water, suggesting the presence of substances inhibitory to the growth of other algae.

The antagonism of rooted aquatics and plankton algae is another object for study. Macrophytes, more predominant 40 years ago then they are now, may gain as algal growths are controlled by phosphate removal from sewage, and indications to this effect are seen.

A more complete assessment of secondary production and fisheries will emerge as the Canadarago study progresses.

B. DELINEATION OF TROPHIC STATES

Canadarago Lake is eutrophic by all criteria employed. The hypolimnion becomes depleted of oxygen during the summer. The lake carries algal blooms with great regularity although species composition of the blooms, duration, and time of year can vary from year to year.

Productivity during the 1969 season (May-November) was 1.0 g C $\cdot$ $m^{-2} \cdot day^{-1}$, a value also observed in eutrophic Lake Erken, Uppland, Sweden (35).

C. TROPHIC STATUS vs NUTRIENT BUDGETS

Phosphorus loading on Canadarago Lake is 0.8 g $\cdot$ $m^{-2} \cdot yr^{-1}$. If Vollenweider's (35) representation of phosphorus loading and mean depth is expressed numerically as follows:

$$E = 40 \cdot L \cdot \bar{Z}^{-0.6}$$

Where: L = P loading ($g \cdot m^{-2} \cdot yr^{-1}$) $\bar{Z}$ = mean depth (meters)

lakes with $E < 1$ would tend to be oligotrophic, and those with $E > 2$ eutrophic. Canadarago with $\bar{Z} = 7.7$ m gives $E = 9.4$, in agreement with its eutrophic conditions. This statement requires that the mean residence time in the lake is sufficient to allow complete conversion of phosphate inputs to biomass. With a theoretical retention time of 217 days, this condition is met. It is also seen that even after reduction of phosphate inputs to 3800 kg (6020 kg less 90 percent of 2660 kg, Table 17), by improved sewage treatment, Canadarago Lake is likely to remain eutrophic (loading of 0.51 g P $\cdot$ $m^{-2} \cdot l^{-1}$, $E = 6.0$). Canadarago Lake, therefore, appears to be a naturally eutrophic lake, a condition regularly found in lakes in a reasonably average setting with regard to nutrient runoff, which are characterized by a similar mean depth and, related to this, similar or larger ratios of littoral and deep water area and of epilimnic and hypolimnic volume.

Another representation proposed by Vollenweider (36) involves utilization of flushing time. In this representation a plot of phosphorus loading (g P · m^{-2} · $year^{-1}$) vs mean depth (m) divided by detention time (years) is constructed. Applying this to Canadarago Lake, with phosphorus loading equal to 0.79 g P · m^{-2} · yr^{-1}, and flushing time equal to 0.595 years, the plot of loading vs mean depth divided by retention time results in a point that lies above the dangerous line, indicating that Canadarago Lake is eutrophic by this evaluation.

The retention of phosphorus in Canadarago is rather low, even for a eutrophic lake, 22.5 percent over the year April 15, 1969-April 14, 1970, or 59.1 percent over the growing season June 1-September 30, 1969 (see e.g. Ref. 37). Figure 11 shows that inputs account for only 10 percent of the seasonal production in terms of phosphorus as determined by the sedimentation technique. This means that incoming phosphorus was utilized approximately 10 times before it was lost by flushing or, to a greater extent, by deposition. Much of this recirculation of phosphate occurred during fall overturn. Erosion of the thermocline during summer may be a contributing factor as in Lake Mendota (38) but increasing exposure of the lake bottom accompanies this and its effects may exceed those of thermocline erosion, Figure 12.

By fall of 1974 Canadarago Lake showed clear signs of recovery from phosphate eutrophication after phosphate removal was instituted at the Richfield Springs Wastewater Treatment Facility approximately two years earlier.

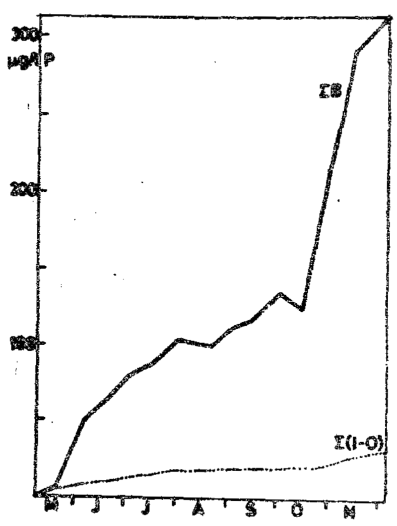


Figure 11. Excess of input over output (lower curve) and total production in Canadarago Lake, measured as phosphorus, 1969 data.

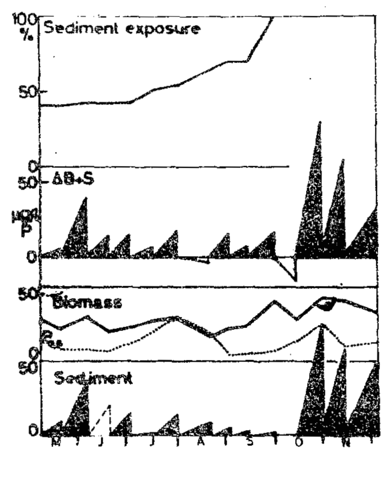


Figure 12. Sedimentation and biomass in Canadarago Lake, Otsego County, New York, 1969. Sediment is total phosphorus in the traps as described in the text. Biomass is phosphorus retained by a 0.45μ membrane filter, P is organic phosphorus passing the filter. Reactive phosphorus is negligible. Δ B + S is the sum of biomass change and sedimentation; sediment exposure represents percent of lake bottom above the thermocline. Reprinted from 20.

SUMMARY

For the past seven years, New York State's Departments of Environmental Conservation and Health have been conducting a technical investigation on Canadarago Lake, and its tributaries, at Richfield Springs, New York as part of the State's program on lake eutrophication. Portions of these data have been included in more than 30 different publications.

Canadarago Lake is situated in East Central New York in the Susquehanna River watershed. The surrounding terrain is hilly with ground elevations from 396 m (1300 ft) to 579 m (1900 ft). The lake's drainage area encompasses 175 km^2 (67.5 sq mi) with four major tributaries draining 78.3 percent of the watershed. The bedrock of the basin is predominantly limestone in the north and shales and siltstones in the south. The soils of the area consist of glacial deposited materials with some isolated recent alluvial deposits.

The permanent population of the lake basin is about 3500 people. Additionally, approximately 1300 people occupy lakeside cottages during the summer. About 49 percent of the watershed is devoted to agriculture, primarily dairy farms, and 34 percent is in forest or brushland. The lake is used primarily for recreational purposes.

Canadarago Lake is nearly 6.4 km (4.0 mi) long and is 1.9 km (1.2 mi) wide at its widest point. The mean depth of the lake is 7.7 m (22 ft) with the maximum depth being 12.8 m (42 ft). The lake has 759 ha (2050 acres) of surface area and 14.4 km (9 mi) of shoreline. The lake has a poorly defined thermocline which is seasonally found at 6 to 8 m (20 to 26 ft) depth. The epilimnion accounts for about 72 percent of the lake volume from June to September. The average hydraulic retention time of the lake has been calculated at 217 days. The lake is ice-covered from December through April.

The depth of 99 percent attenuation of white light averages about 7 meters (23 ft) with the Secchi disc depth ranging seasonally from 1 to 3 m (3.3 to 10 ft). The pH is commonly above 8, with pH's above 8.5 occasionally observed in May and September. Dissolved oxygen in the top 6 m (20 ft) averages about 10 $mg \cdot l^{-1}$ from May to November, but the region below 11 m (36 ft) becomes anoxic from the middle of July to the end of September.

Total phosphorus averages about 50 $\mu g \cdot l^{-1}$ with about 50 percent of this being soluble. Summer orthophosphate phosphorus is below 5 $\mu g \cdot l^{-1}$ in the surface water, but commonly exceeds 50 $\mu g \cdot l^{-1}$ in the anoxic deep region during August and September. Ammonia nitrogen averages 150 $\mu g \cdot l^{-1}$, and nitrate plus nitrite nitrogen drops from over 500 $\mu g \cdot l^{-1}$ in spring to less than 50 $\mu g \cdot l^{-1}$ from July to November.

Total organic carbon is about 5 $mg \cdot l^{-1}$ of which about two-thirds is soluble. The highest levels of dissolved organic carbon occur in the

euphotic zone during May and June. The lake water can be considered a moderately hard water lake. Calcium carbonate precipitation occurs to a measurable extent.

The highest chlorophyll a concentrations exist in the top 5 m (16.4 ft) with the average concentration during 1968 and 1969 being about 10 $\mu g \cdot l^{-1}$. Mean primary production is about 1 g carbon $\cdot$ $m^{-2} \cdot day^{-1}$. Algal assays have indicated that phosphate and iron - EDTA innocula significantly increased CO_2 fixation while nitrate additions were always without effect. *Cyanophyta* dominate summer plankton samples while *Chrysophyta* are most common in spring and fall. Common phytoplankton include *Aphanizomenon flos-aquae*, *Anacystis incerta*, *Stephanodiscus niagarae*, *Cyclotella comta*, *Sphaerocystis schroeteri*, *Ceratium hirundinella*, and *Trachelomonas spp*. The common zooplankton include *Eubosmina*, *Daphnia*, and *Diaptomus* with the assemblages evenly divided between Cladocerans and Copepods. *Chaoborus* and six genera of rotifers have also been identified.

The benthic fauna consists primarily of *Chironomidae* with the remainder primarily Oligochaetes. Ongoing fish studies indicate yellow perch and golden shiner to be the most common pelagic fish and chain pickerel the most predominant predator although a total of 40 species have been identified. No microbiological work has been attempted, but benthic algae and aquatic macrophyte communities have been characterized.

The prime emphasis of this project has been to develop nutrient budgets for the biologically important chemical elements and to relate the budget to the trophic status of the lake. From April 1969 to April 1970, 44.1 percent of the phosphorus input entered the lake from the Richfield Springs Sewage Treatment Plant, 42.4 percent from the lake's four major tributaries, 9.4 percent from the ungauged portion of the watershed, 2.3 percent from lakeside dwellings and 1.7 percent from direct precipitation on the lake surface. The net accumulation of phosphorus in the lake during this period (inputs minus outflow) was 790 kg/yr (2742 lbs/yr). A major portion, 68.9 percent, of the soluble phosphorus entered the lake from the Richfield Springs Sewage Treatment Plant. In contrast, 91.2 percent of the total nitrogen input during the same period resulted from stream discharge.

The phosphorus loading has been calculated to be 0.8 g $\cdot$ $m^{-2} \cdot yr^{-1}$. Following Vollenweider's work, the lake should be considered eutrophic and indeed it is.

During 1972, a modern wastewater treatment facility was constructed to replace the existing sewage treatment plant at Richfield Springs. The new plant provides phosphorus removal, and preliminary results indicate that the problem of cultural eutrophication seems to be lessening in Canadarago Lake.

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SECTION IV - OHIO

LIMNOLOGICAL AND GEOCHEMICAL CHARACTERISTICS OF THE TWIN LAKES WATERSHED, OHIO

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I. INTRODUCTION

The Twin Lakes Watershed is a heavily urbanized ecosystem with three culturally eutrophic glacial lakes and four small upland, manmade ponds (Cooke, et al. 1973). In 1973, sewage (septic tank) diversion was essentially completed. The Twin Lakes Project was established in late 1971 to measure the response of the two main lakes (East and West) to diversion, and to investigate the efficacy of phosphorus precipitation by aluminum sulfate as a means of accelerating recovery. Monitoring data for 1972-1974 from that project (EPA 16010 HCS, R801936) is reported here.

Methods of measurements for hydrologic, geological, and limnologic data are given in Section IV.

II. GEOGRAPHIC DESCRIPTION OF WATER BODY

A. Latitude and Longitude. These data are listed in Table 1.
B. Altitude Above Sea Level. These data are listed in Table 1.
C. Catchment Area. These data are listed in Table 1.

Table 1.
Morphological and Hydrological Data of the Twin Lakes Watershed

Latitude-Longitude	41° 12' N. Latitude, 81° 21' W. Longitude	
Area of Watershed (ha.)	334.5 (including lakes)	
Population Estimate (1975)	1510 (452/km.^{2})	
	West Twin Lake	East Twin Lake
Area (ha.)	34.02 (including canals & lagoons)	26.88
Maximum length (km.)	0.65	0.85
Maximum width (km.)	0.60	0.50
Volume (m^3) (V)	14.99×10^5 (including canals & lagoons)*	13.50×10^5
Maximum depth (m.)	11.50	12.00
Mean depth (m.)	4.34	5.03
Elevation (m.)	318.73	318.42
Water renewal time (yrs.) (=V/Q)	1.64 (1972) 1.81 (1973) 1.03 (1974)	0.79 (1972) 0.93 (1973) 0.58 (1974)
Area of other lakes in in sub-watershed	15	3

*These shallow areas are excluded from calculations of mean concentrations and amounts of nutrients.

D. General Climatic Data. Portage County has a humid-temperate continental type climate with an average frost-free season of 168 days. Average dates of spring and fall killing frosts are May 2 and October 17. Average January temperature is -3°C, the average July temperature 21.8°C. Temperature extremes are 39°C and -30°C (Ritchie and Powell, 1973). Insolation has not been measured.

Precipitation-evaporation data for 1972-74 is summarized in Tables 4 and 5. The highest occurred in September 1972 with 20.7 cms. and one storm of 9.1 cms. Highest evaporation occurs in June-August.

E. General Geological Characteristics. Geologic materials in the watershed are comprised of up to 45.7 meters of deposits overlying sandstone bedrock. The Twin Lakes are situated on the axis of a buried bedrock valley (Winslow and White, 1966), filled with outwash deposits of silt, sand and gravel derived from the Kent Ice advance, which occurred about 15,000 years ago. The western belt of the deposits left by the Kent moraine is composed of a high proportion of sand and gravel. Kettle holes are common. The deepest are sites of ponds and lakes, including Twin Lakes. Earth materials surrounding the lakes are sand and fine gravel on the uplands, silts and organic soils in the undrained depression areas. Underlying sand and gravel is gray silt varying in thickness from 3 to 10 meters. The silt forms a confining layer over coarser sand and gravel deposits which lie at depths from 12 to 20 meters below the surface. The deep sand and gravel serves as the principal aquifer for the wells of residents in the Twin Lakes area. Soils developed on the glacial materials are well drained and moderately permeable, except in lowlands. Erosion potential is low where the soils are protected by vegetative cover. Construction in the steep areas has caused severe erosion and sedimentation.

F. Vegetation. Open space in the watershed is comprised of small upland areas of oak, beech, hickory, and sugar maple woods, low poorly drained areas of elm, maple, and willow, and swampy areas with poison sumac, swamp maple, alder and sparse tamarack. No extensive open fields or pasture land are in the basin, except for large lawns.

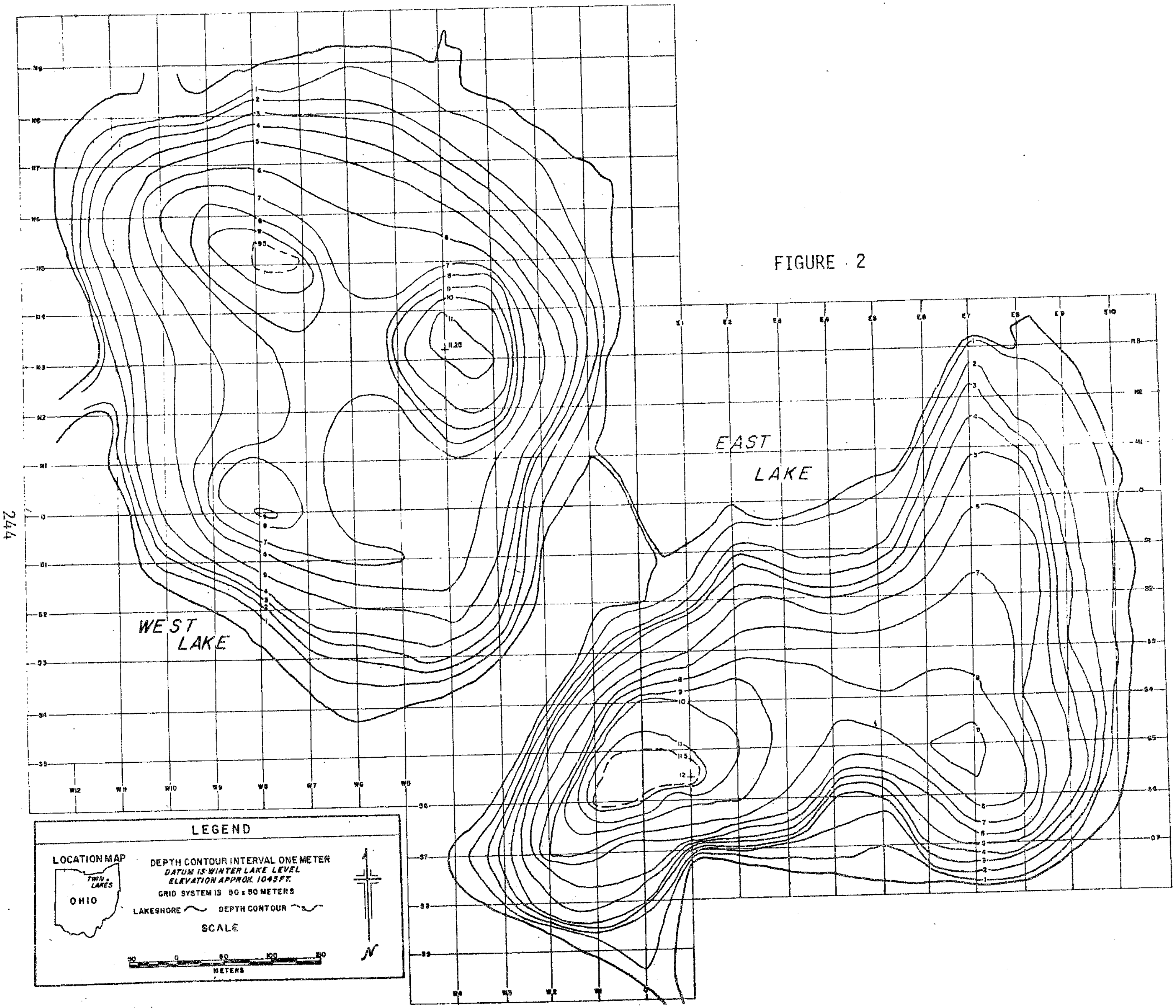
G. Population. There are approximately 1510 people living within the watershed in 430 houses.

H. Land Usage. The watershed contains two types of land: residential and open space. The major land use is single family residential.

I. Use of Water. The water in the lakes is used solely for recreation.

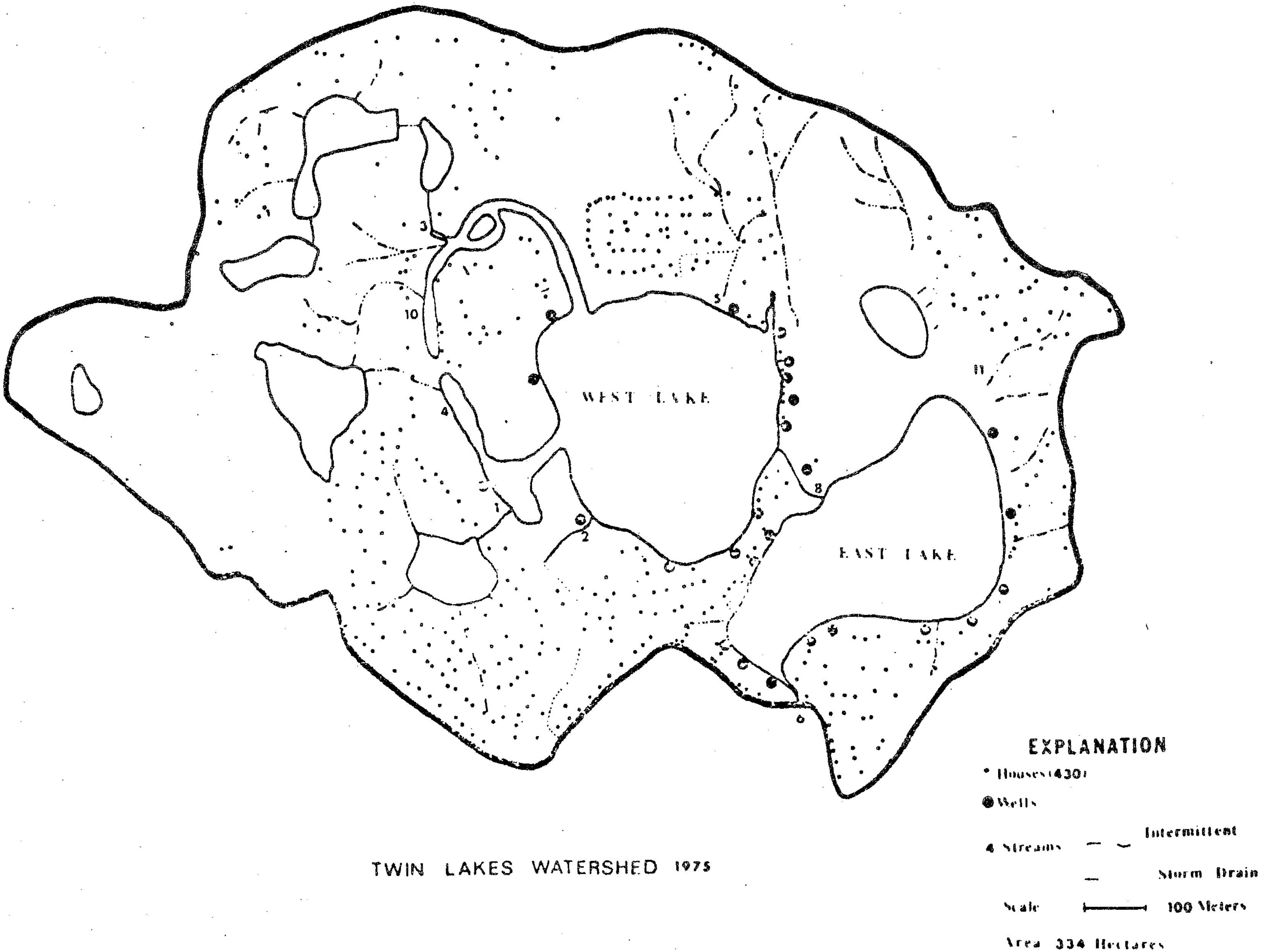
J. Sewage and Effluent Discharge. Until 1972, sewage was discharged into septic tanks and thence by groundwater and stream flow to the lakes. Sewage was diverted during late 1971 through 1972 to a package plant which discharges away from the watershed. All storm drainage enters the lakes. There is no industrial discharge.

FIGURE 2



EAST
LAKE
WEST
LAKE
LEGEND
LOCATION MAP
TWIN LAKES
OHIO
DEPTH CONTOUR INTERVAL ONE METER
DATUM IS WINTER LAKE LEVEL
ELEVATION APPROX. 1045 FT.
GRID SYSTEM IS 50 x 50 METERS
LAKESHORE
DEPTH CONTOUR
SCALE
METERS
N

FIGURE 1



TWIN LAKES WATERSHED 1975

III. MORPHOMETRIC AND HYDROLOGIC CHARACTERISTICS OF THE TWIN LAKES

The two lowermost lakes of the Twin Lakes Watershed, East (ETL) and West (WTL) Twin Lakes, are small eutrophic kettle-type lakes of similar morphology (Figure 2). WTL is slightly larger in area and volume and lower in mean depth, due primarily to the construction of a lagoon and canals on the west and northwest sides of the lake (Figure 1).

A. Surface Area, Length, Width. See Table 1.

B. Volume of Water and Regulation. Lake volumes and volume-area relationships are given in Tables I and 2. The lakes receive water from precipitation, outflow of small, upland, man-made ponds, storm flow, small spring-fed woodland streams, and groundwater. Water is lost by evaporation, and by outflow from ETL. Rate of outflow is partially controlled by a small marsh and golf course pond. WTL and Dollar Lake flow into ETL.

C. Maximum and Average Depths. See Table 1.

D. Exceptional Depths and Ratio of Surface Area of Deep and Shallow Waters. See Table 1 for depths. Shallow waters are considered to be the zone of macrophyte growth. Using areas of Table 2 and areas of macrophytes (see Section IV, C. 7), the ratio of deep to shallow waters of WTL and ETL are 3.96 and 2.49 respectively.

Table 2. Volumes and Areas of Lake Strata

Depth or Top of Stratum	Volume (m^3)	Area (m^2)	Volume (m^3)	Area (m^2)
0	319,250	340,152	252,911	268,820
1	247,333	276,112	218,156	237,330
2	223,659	234,702	188,917	199,530
3	202,391	213,700	169,883	178,500
4	175,592	191,290	151,315	161,410
5	141,592	160,350	127,240	141,440
6	94,406	123,630	98,825	113,550
7	49,886	67,940	68,660	84,800
8	25,824	33,800	38,331	53,700
9	13,627	18,600	19,305	24,800
10	5,459	9,200	10,217	14,290
11	203	2,440	3,389	6,630
Total	1,499,222m^3		1,350,568m^3	

E. Ratio of Epi- Over Hypolimnion. The principal metalimnetic strata were identified from temperature data. Table 3 catalogs this feature for all observation days. Table 11 shows average extents, volumes, and volume-ratios of the epilimnion and hypolimnion. During 1971-74, the metalimnion has tended to occur deeper in each lake.

TABLE 3

Catalog of Observation Dates. Thermal conditions indicated in parentheses: Ice = ice present; --- = unstratified; Str or numbers = metalimnion present, with numbers indicating depths (m) at which the bounds of the metalimnion occurred.

East Twin Lake

1971 — Total Visits 35 days

Apr	5	(---)
	9	(---)
	14	(---)
	17	(2,3)
	21	(1,4)
May	5	(7,8)
	12	(1,4)
	19	(2,5)
	28	(Str)
Jun	4	(Str)
	8	(1,7)
	18	(Str)
	23	(2,7)
	29	(2,6)
Jul	7	(2,7)
	13	(2,7)
	20	(3,8)
	27	(3,7)
Aug	3	(3,7)
	10	(3,7)
	17	(3,7)
	24	(3,8)
Sep	2	(3,8)
	8	(2,8)
Oct	1	(2,8)
	11	(5,8)
	14	(6,9)
	21	(6,9)
	28	(6,9)
Nov	5	(8,10)
	11	(---)
	15	(---)
	18	(---)
	26	(---)
Dec	7	(---)

1972 — Total Visits 47 days

Jan	31	(Ice)
Feb	10	(Ice)
	17	(Ice)
	24	(Ice)
Mar	2	(Ice)
	9	(Ice)
	24	(---)
	28	(---)
Apr	4	(---)
	11	(---)
	18	(2,6)
	25	(5,7)
May	2	(1,8)
	9	(3,8)
	16	(2,8)
	23	(2,8)
	30	(2,9)
Jun	6	(3,9)
	13	(2,8)
	20	(1,8)
	27	(1,9)
Jul	5	(2,9)
	11	(1,8)
	18	(1,8)
	25	(2,7)
Aug	1	(3,7)
	8	(3,7)
	15	(2,7)
	22	(2,7)
	29	(2,8)
Sep	5	(3,8)
	12	(4,8)
	19	(4,9)
	26	(4,10)
Oct	3	(6,10)
	10	(6,10)
	17	(6,8)
	24	(8,9)
	31	(9,11)
Nov	7	(9,10)
	14	(---)
	21	(---)
	28	(---)
Dec	5	(---)
	12	(Ice)
	19	(Ice)
	26	(Ice)

1973 — Total Visits 43 days

Jan	2	(Ice)
	16	(Ice)
Feb	13	(Ice)
	20	(Ice)
	27	(Ice)
Mar	6	(Ice)
	13	(---)
	20	(---)
	27	(---)
Apr	3	(---)
	10	(---)
	17	(8,9)
	24	(Str)
May	1	(2,8)
	8	(2,9)
	15	(5,9)
	29	(3,9)
Jun	5	(2,8)
	12	(2,7)
	19	(2,8)
	26	(2,9)
Jul	3	(2,9)
	10	(3,8)
	17	(Str)
	24	(2,8)
	31	(Str)
Aug	7	(3,8)
	14	(3,8)
	21	(3,8)
	28	(1,8)
Sep	4	(2,7)
	11	(3,7)
	18	(4,8)
	25	(4,8)
Oct	2	(5,8)
	9	(5,8)
	16	(5,8)
	23	(7,9)
	30	(Str)
Nov	6	(Str)
	13	(---)
	27	(---)
Dec	11	(---)

1974 — Total Visits 47 days

Jan	7	(Ice)
	28	(Ice)
Feb	11	(Ice)
	27	(Ice)
Mar	6	(---)
	18	(---)
Apr	1	(---)
	9	(---)
	16	(7,9)
	24	(4,6)
	30	(2,5)
May	7	(4,6)
	17	(2,7)
	24	(1,8)
	31	(2,8)
Jun	7	(2,7)
	14	(3,7)
	20	(3,7)
	26	(3,7)
Jul	2	(3,7)
	11	(2,8)
	18	(2,7)
	25	(Str)
	30	(3,8)
Aug	6	(3,8)
	13	(3,8)
	20	(Str)
	27	(2,8)
Sep	3	(4,8)
	10	(4,9)
	17	(4,8)
	24	(5,8)
Oct	1	(6,8)
	8	(7,9)
	15	(4,8)
	22	(Str)
	29	(9,10)
Nov	5	(Str)
	12	(10,11)
	19	(---)
	26	(---)

West Twin Lake

1971 — Total Visits 24 days

Apr	20	(2,5)
	28	(8,9)
May	15	(2,5)
Jun	8	(2,6)
	22	(2,6)
	28	(1,7)
Jul	6	(2,6)
	12	(2,6)
	19	(2,6)
	26	(2,6)
Aug	2	(2,6)
	9	(3,6)
	16	(2,6)
	23	(2,5)
Sep	1	(2,6)
	7	(1,6)
	29	(4,6)
Oct	12	(6,7)
	19	(5,6)
	26	(5,7)
Nov	4	(8,9)
	11	(---)
	18	(---)
	26	(---)

1972 — Total Visits 47 days

Jan	11	(Ice)
Feb	8	(Ice)
	15	(Ice)
	22	(Ice)
	29	(Ice)
Mar	7	(Ice)
	24	(---)
	31	(---)
Apr	6	(---)
	13	(4,5)
	20	(4,6)
	27	(5,8)
May	4	(2,7)
	11	(Str)
	18	(3,7)
	25	(0,8)
Jun	1	(3,8)
	8	(4,7)
	15	(3,9)
	22	(3,7)
	29	(3,7)
Jul	6	(4,8)
	13	(2,9)
	20	(2,7)
	27	(1,8)
Aug	3	(4,8)
	10	(4,8)
	17	(3,7)
	24	(3,9)
	31	(2,7)
Sep	7	(4,9)
	14	(3,9)
	21	(4,7)
	28	(5,8)
Oct	5	(5,8)
	12	(6,9)
	19	(8,10)
	26	(---)
Nov	2	(---)
	9	(---)
	16	(---)
	22	(---)
	30	(---)
Dec	7	(---)
	14	(Ice)
	21	(Ice)
	29	(Ice)

1973 — Total Visits 45 days

Jan	4	(Ice)
	11	(Ice)
	18	(Ice)
Feb	15	(Ice)
	22	(Ice)
Mar	1	(Ice)
	8	(Ice)
	15	(---)
	22	(---)
	29	(---)
Apr	5	(---)
	12	(---)
	19	(5,6)
	26	(2,6)
May	3	(5,8)
	10	(5,8)
	17	(7,8)
	24	(6,7)
	31	(4,9)
Jun	7	(4,8)
	14	(3,8)
	21	(3,8)
	28	(3,9)
Jul	5	(2,9)
	12	(3,8)
	19	(2,7)
	26	(3,8)
Aug	2	(3,8)
	9	(3,8)
	16	(3,7)
	23	(4,7)
	30	(1,9)
Sep	6	(2,7)
	13	(4,7)
	20	(5,9)
	27	(5,8)
Oct	4	(5,9)
	11	(5,8)
	18	(8,9)
	25	(9,10)
Nov	1	(---)
	8	(---)
	15	(---)
	27	(---)
Dec	11	(---)

1974 — Total Visits 40 days

Jan	7	(Ice)
	28	(Ice)
Feb	11	(Ice)
	27	(Ice)
Mar	6	(---)
	18	(---)
Apr	1	(---)
	12	(---)
	19	(3,4)
	27	(Str)
May	3	(3,6)
	14	(4,7)
	21	(2,7)
	28	(3,7)
Jun	4	(2,7)
	11	(3,6)
	18	(4,7)
	25	(3,7)
Jul	1	(4,7)
	9	(1,7)
	18	(2,7)
	25	(3,8)
	30	(3,6)
Aug	6	(3,7)
	13	(3,8)
	20	(Str)
	27	(2,7)
Sep	5	(4,7)
	12	(2,7)
	19	(4,7)
	26	(5,7)
Oct	3	(7,10)
	10	(8,9)
	17	(Str)
	24	(---)
	31	(---)
Nov	7	(---)
	14	(---)
	21	(---)
	27	(---)

F. Duration of Stratification. Both lakes are dimictic second class lakes. For purposes of discussion an annual cycle of four stadia is defined: WINTER (ice present), SPRING (unstratified, SUMMER (metalimnion present), and FALL (unstratified). Table 3 catalogs the occurrence of these conditions for all days the lakes were visited. For convenience in this report, the start of winter has been defined as January 1.

Ice usually appears in December and disappears in early March. The lakes commonly thaw and refreeze at least once during this interval. Ice thickness is usually 7-10 cm.; thickness of 30 cm.+ has been reported.

The 1971-74 average onset of summer was April 14 for both lakes. The 1971-74 average summer lasted 211 days in ETL (to November 11) and 196 days in WTL (to October 27). During 1971-1974, summer conditions have lengthened in ETL from 207 to 217 days and have shortened in WTL from 206 to 188 days.

G. Nature of Lake Sediments. Sediment characteristics of the surficial (upper 2 cm.) muds of the littoral, metalimnion (sublittoral) and hypolimnion (profundal) were determined in 1972-73 by Lardis (1973) (Table 4). Littoral sediments contain mainly decaying vegetation, shell fragments, and allochthonous debris. Sublittoral muds are brown to black-gray with lesser and varying amounts of decaying vegetation. Profundal zone muds are dark gray-black in ETL, brownish-black in WTL, and are much blacker during anoxic periods; rusty-brown above a gray-black layer during oxygenated periods. Profundal sediments throughout the year are very fluid and easily disturbed. Both lakes exhibit an increased amount of organic phosphorus (method of Mehta et al. 1954) with depth of overlying water. Mean phosphorus content of ETL sediment is significantly less than WTL (0.66 mg.P/g. vs. 0.85 mg.P/g.), eventhough loading to ETL indicates ETL to be more enriched. Lardis attributec this to the organic phosphorus added to WTL during the dredging of the canals in 1969.

The mean phosphorus content of the littoral zones are similar and vary little from season to season. The sublittoral of WTL has considerably more phosphorus than ETL and exhibits a decline from fall to winter. The greatest difference between lakes is in profundal samples. In both lakes, the phosphorus content of profundal surficial muds increases from fall to spring, then declines after onset of summer anoxic conditions (13% mean decrease in WTL, from spring levels), suggesting that the increase in dissolved ortho phosphate in hypolimnetic waters may in part be from this decrease in sediment-interstitial water phosphorus.

The organic content of ETL and WTL surficial sediments increases with depth of sample; WTL profundal samples have more organic matter than ETL. The percent organic content of dry sediment samples ranges from 14% in littoral to 39% in profundal in ETL, 14% in littoral to 41% in profundal in WTL. The water content of surficial sediments ranged from 71.5 to 97%; highest values were found in profundal samples. Most samples were 94-96% water.

Table 4. Means of Total Organic and Dissolved Inorganic Phosphorus in the Sediments of Each Limnetic Zone (±25 $\bar{x}$; n = number of samples; PO_4-P in mg.P/g. dry sediment) from Lardis (1973).

	Fall	n	Winter	n	Spring	n	Summer	n
West Twin Lake								
Littoral	.54±.08	19	.54±.09	8	.53±.09	8	.57±.12	7
Sublittoral	.89±.03	9	.78±.05	8	.79±.05	8	.77±.06	8
Profundal	1.19±.08	21	1.26±.13	9	1.27±.11	9	1.04±.03	9
East Twin Lake								
Littoral	.43±.07	19	.53±.13	9	.52±.10	9	.53±.08	9
Sublittoral	.65±.06	9	.65±.05	7	.67±.06	8	.67±.06	7
Profundal	.80±.03	20	.83±.06	9	.87±.07	9	.77±.04	9

H. Seasonal Variation of Preciptation and Evaporation

Table 5. Precipitation and Evaporation. Twin Lakes Watershed.

	1972			1973			1974		
Mo.	Precip. (m)	Volume (m^3)	Evapo. (m)	Precip. (m)	Volume (m^3)	Evapo. (m)	Precip. (m)	Volume (m^3)	Evapo. (m)
J	.0356	118904		.0417	139278		.0907	302938	
F	.0511	170674		.0483	161322		.0706	235804	
M	.1001	334334	.0527	.0623	208082	.0247	.1044	348696	.0660
A	.1628	543752	.0606	.0875	292250	.0529	.1270	424180	.0889
M	.0955	318970	.0970	.1270	424180	.0896	.1155	385770	.1092
J	.1018	340012	.1356	.1028	343352	.0929	.0767	256178	.1499
J	.0823	274882	.1577	.0655	218770	.1074	.0723	241482	.1727
A	.0612	204408	.1516	.0726	242484	.1119	.1750	584500	.1372
S	.2070	691380	.1233	.0708	236472	.1180	.0558	186372	.0805
O	.0386	128924	.0880	.1143	381762	.1000	.0695	232130	.0559
N	.0996	332664		.0657	219438		.1270	424180	
D	.0886	295924		.0617	206078		.0589	196726	
Total	1.1242	3754828	0.8665	0.9202	3073478	0.6974	1.1434	3818956	0.8603
Mean	0.0937	312902	0.1083	0.0767	256123	0.0872	0.0953	318246	0.1075

Area of watershed = 334 hectares. Volume to lakes obtained by multiplying lake area (m^2) by precipitation (m).

I. Inflow-Outflow of Water

Table 6. Water Inflow-Outflow ($m^3 x 10^3$)

	1972	1973	1974
A. West Twin Lake			
surface streams	145.45	189.79	332.45
groundwater	307.37	321.33	307.33
precipitation on lake	382.35	313.01	401.96
runoff	382.96	362.56	598.60
total inflow	1218.12	1177.68	1640.33
evaporation	343.61	273.53	270.33
outflow (to ETL)	916.84	826.24	1461.58
B. East Twin Lake			
surface streams	1057.31	956.12	1479.02
groundwater	246.02	247.02	246.02
precipitation on lake	251.36	246.61	317.66
runoff	379.66	332.93	416.23
total inflow	1934,34	1678.13	2458.93
evaporation	268.70	227.69	223.68
outflow (out of watershed	1700.01	1444.92	2307.49

J. Water Currents. No investigations of water currents in the Twin Lakes have been made.

K. Water Renewal Time. Water renewal times (years) are listed in Table 1.

IV. LIMNOLOGICAL CHARACTERIZATION SUMMARY

Methods

1. Limnological Methods. Unless otherwise noted, all limnological observations were made from a water column over the deepest point in each lake at depths 0.1, 2, 4, 7, and 10 meters. Table 3 catalogs all days on which the lakes were visited. Visits were generally weekly from late spring through early fall, but less frequently otherwise. The list of features monitored was complete for most but not all visits. An annotated list of quantitative methods is given below:

a. Physical
   - (1) temperature--at one-meter intervals; Whitney resistance thermometer.
   - (2) transparency--20 cm. diameter, alternating black-white quadrants, Secchi Disc.
   - (3) light--Whitney LMD-8A photometer with sea and deck cells.
   - (4) conductance--in the laboratory; YSI Model 31 conductivity bridge.

b. Chemical
   - (1) pH--in laboratory; Corning Model 7 meter and combination electrode.
   - (2) alkalinity--titration with 0.02N H_2SO_4; endpoint pH 4.5.
   - (3) dissolved oxygen--at one-meter intervals; titration with 0.0125N sodium thiosulfate, azide modification.
   - (4) sulfate--turbidmetric, using Hach Chemical Co. reagents; standard curve prepared in our laboratory.
   - (5) nitrate--cadmium reduction using Hach Chemical Co. reagents; standard curve prepared in our laboratory.
   - (6) ammonia--direct nesslerization, using Hach Chemical Co. reagents; standard curve prepared in our laboratory.
   - (7) ortho PO_4-P--at one meter intervals; ascorbic acid-ammonium molybdate, on 0.45 μ Millipore filtered samples.
   - (8) total PO_4-P unfiltered--at one-meter intervals; persulfate-sulfuric acid digestion. PO_4-P determined as in (7).
   - (9) total PO_4-P filtered--persulfate-sulfuric acid digestion of 0.45 μ filtered samples. PO_4-P determined as in (7).

c. Biological
   - (1) phytoplankton--25 ml. samples filtered on 0.45 μ Millipore filters, dried, cleared with immersion oil, counted at 140 x, 11 Whipple fields; of dominant species, using appropriate geometric shapes to calculate cell volume (McNabb, 1960. Limnol. Oceanogr. 5:57).
   - (2) Chlorophyll A--500 ml. sample filtered through GF/A filter, extracted with 90% buffered acetone, using tissue grinder; equations (trichromatic) of Parsons and Strickland, no acid correction (Long and Cooke, 1971. Limnol. Oceanogr. 16:990).

(3) zooplankton--vertical tow, #20 net, lake bottom to top of hypolimnion (or 6 m. when unstratified) and bottom of metalimnion to surface (6 m. to surface when unstratified), using rim line and weighted bucket to close net. 10 ml. aliquots counted in duplicates.
(4) macrophytes--outer limit of plant distribution from shore measured at several points around lake perimeter. Percent cover estimated, and plant samples obtained by use of SCUBA. Dry weight per area multiplied by percent cover and area of macrophyte community. Net yield estimated by difference.
(5) potential plankton metabolism--plankton samples were incubated in the laboratory at 5000 lux. Metabolism was measured by the pH method in light and dark bottles after 4 hours of incubation.

2. Surface Water Measuring Methods.

Twin Lakes Stream

Station	Measuring Method	Frequency of Measure
1	90° V notch weir and stilling well	Continuous recorder
2	90° V notch weir and stilling well	Continuous recorder
3	3'H flume, Agriculture Research Service design	Continuous recorder
4	15" Culvert discharge, current meter	Daily
5	24" Culvert discharge, current meter	Daily
6	Stage-Discharge Rating Curve, stilling well	Continuous
7	2 submerged culverts 15" diameter, current meter	Daily
8	Culvert, bucket	Daily
9	90° V notch weir, bucket	Daily
Dollar Lake Stream Station		
1	Culvert discharge, bucket	Daily
2	Culvert discharge, bucket	Daily
3	60° V notch weir and stilling basin	Daily
4	Culvert discharge, bucket	Daily
5	90° V notch weir, and stilling well	Daily

3. Surface Run-off. Land runoff or storm flows were computed from lake level increases, as recorded by limnographs, in excess of that from direct precipitation and stream inflows.

4. Ground Water Methods. Twenty-eight shallow wells were installed around the perimeter of the lakes and a flow net was constructed. Specific discharge was determined from the hydraulic gradient and field measurement of permeability. Average cross-sectional discharge depth between these wells was assumed to be 3.0 meters (range 1-6 meters). Wells were sampled monthly for water chemistry. A deep piezometer nest beside WTL was used to estimate the upward hydraulic gradient and discharge into the lakes of deep ground water. Ground water inflow and loading is the sum of shallow and deep groundwater discharge.

5. Precipitation-Evaporation. Precipitation was measured with a recording Leupold-Stevens type Q 6 weighing bucket, located at WTL. Rain and snow samples were collected at the University (8 km. south of watershed) for chemical analysis. These samples included dry fallout. Evaporation was measured in 1972 from daily temperature data, using the Blaney-Criddle equation, and in 1973-74 with a U.S. Weather Bureau Class A Evaporating Pan and a Weather Measure Recording Evaporimeter.

Results. Averaged data are displayed in tabular form as indicated in the annotated list below. Wherever appropriate in averaging, values for limnological features were weighted with respect to time in days. Where appropriate, both unweighted (COL) and volume-weighted (LAKE) values were averaged. For features expressed as concentrations, average total amounts in the lake may be found by multiplying the LAKE averages by the total lake volumes (Table 1). The surface densities of these average total amounts may be found by further dividing by total lake areas (Table 1).

A. Physical

1. Temperature. Tables 7 and 8.

2. Conductance. Tables 9 and 10.

3. Light. Secchi disc transparency is summarized in Table 12. Depths of light extinction at 10%, 1% and 0.1% of surface light intensity are presented in Table 13 for selected dates in 1971 and 1972.

4. Color. No measurements of color have been made.

5. Solar Radiation. No measurements of incident solar radiation have been made.

B. Chemical.

1. pH. Tables 14 and 15.

2. Dissolved Oxygen. Tables 16 and 17.

3. Phosphorus. Total phosphorus concentrations are summarized in Tables 18 and 19. Filterable total "soluble" phosphorus concentrations are summarized in Tables 20 and 21. Filterable ortho phosphorus concentrations are summarized in Tables 22 and 23.

4. Nitrogen. "Total" nitrogen (ammonium and nitrate nitrogen only) concentrations are summarized in Tables 24 and 25. The two fractions recorded are summarized in Tables 26, 27, 28, and 29.

5. Alkalinity. Tables 30 and 31.

6. Electrolytes. Sulfate concentrations only are summarized in Tables 32 & 33.

7. Trace Metals. No measurements of trace metal concentrations were made.

Table 7. TEMP COL: Average Temperatures in the Column (^{o}C).

West Twin Lake Year	Winter Season	Spring Season	Summer Season	Epilim Only	Hypolim Only	Fall Season	All Seasons
1971	-	-	14.146	20.533	10.713	6.737	12.902
1972	2.296	3.776	13.405	18.831	8.240	5.155	8.657
1973	2.346	6.469	15.643	20.506	11.095	7.377	10.743
1974	3.179	6.209	13.853	19.446	9.766	7.488	10.042
East Twin Lake							
1971	-	6.845	13.737	20.945	8.930	4.234	11.347
1972	2.163	3.538	12.559	19.156	7.027	3.848	8.649
1973	2.011	5.049	14.539	20.496	9.552	6.875	10.011
1974	3.208	6.075	12.732	18.241	8.001	6.055	9.740

Table 8. TEMP LAKE: Average Temperatures in the Lake (^{o}C).

West Twin Lake							
1971	-	-	17.125	20.554	10.853	6.784	15.388
1972	2.161	3.782	17.200	18.844	8.389	5.150	10.586
1973	2.312	6.661	18.707	20.501	11.289	7.440	12.398
1974	3.116	6.333	17.041	19.462	9.943	7.550	11.773
East Twin Lake							
1971	-	7.232	17.083	20.972	9.065	4.262	13.833
1972	1.930	3.729	15.952	19.171	7.147	3.788	10.604
1973	1.992	5.322	17.679	20.504	9.663	6.950	11.770
1974	3.189	5.895	15.618	18.271	8.119	6.154	11.520

Table 9. COND COL: Average Conductances ($20^{o}C$) in the Column (μmho)

West Twin Lake							
1971	-	-	-	-	-	-	-
1972	-	-	415.68	376.85	435.07	400.81	412.39
1973	428.02	430.52	412.47	391.00	430.49	-	417.89
1974	-	420.95	413.31	378.80	449.44	400.75	414.12
East Twin Lake							
1971	-	-	-	-	-	-	-
1972	-	-	375.47	357.69	384.72	373.34	375.12
1973	429.29	400.76	381.54	362.45	413.14	-	393.59
1974	-	399.72	398.61	373.29	417.63	376.21	397.10

Table 10. COND LAKE: Average Conductances ($20^{o}C$) in the Column (μmho)

West Twin Lake							
1971	-	-	-	-	-	-	-
1972	-	-	403.01	377.09	434.40	390.45	400.23
1973	428.19	426.54	400.08	390.97	428.77	-	409.22
1974	-	420.95	390.37	378.10	466.45	403.18	403.23
East Twin Lake							
1971	-	-	-	-	-	-	-
1972	-	-	363.78	357.36	384.38	373.56	365.36
1973	420.53	393.54	367.94	363.23	409.29	-	381.74
1974	-	400.24	382.58	372.83	415.45	378.05	388.46

Table 11. Average Dimensions of Epilimnion and Hypolimnion.

A. Lowest Extents of Epilimnion (m).

	1971	1972	1973	1974	All
West Twin Lake	3.114	3.375	4.179	3.604	3.562
East Twin Lake	2.944	3.367	3.600	4.044	3.496

B. Highest Extents of Hypolimnion (m).

	1971	1972	1973	1974	All
West Twin Lake	6.243	7.821	7.964	7.114	7.275
East Twin Lake	7.167	8.369	8.250	7.906	7.926

C. Volumes of Epilimnion (m^3).

	1971	1972	1973	1974	All
West Twin Lake	715866	796415	937229	829430	818380
East Twin Lake	609036	665854	725323	770224	693600

D. Volumes of Hypolimnion (m^3).

	1971	1972	1973	1974	All
West Twin Lake	186099	68646	58244	108873	106405
East Twin Lake	158646	72691	64412	97291	98001

E. Ratios of Volumes, Epilimnion/Hypolimnion.

	1971	1972	1973	1974	All
West Twin Lake	3.847	11.602	16.091	7.618	7.691
East Twin Lake	3.839	9.160	11.261	7.917	7.077

Table 12. SECCHI TRANSPARENCY: Average Secchi Disc Depths (m).

West Twin Lake

Year	Winter	Spring	Summer	Fall	All
1971	-	-	1.622	2.031	1.707
1972	2.728	1.050	1.437	4.034	2.170
1973	2.519	1.181	2.948	3.282	2.754
1974	1.593	1.432	2.177	3.883	2.323

East Twin Lake

Year	Winter	Spring	Summer	Fall	All
1971	-	-	2.261	1.701	2.146
1972	1.384	1.043	1.655	2.188	1.623
1973	1.858	1.590	2.402	2.988	2.304
1974	1.886	1.535	1.903	1.962	1.866

Table 13. LIGHT EXTINCTION
Depths at Intensities .100, .010 and .001 of Surface.

Light Intensity	West Twin Lake		East Twin Lake	
	1971	1972	1971	1972
.100	1.4679	2.4286	2.0250	1.7600
.010	3.5357	5.3571	4.6393	3.6800
.001	4.9536	6.7833	5.6857	5.1600

Table 14. pH COL: Average pH Values in the Column.

Year	Winter Season	Spring Season	Summer Season	Epilim Only	Hypolim Only	Fall Season	All Seasons
West Twin Lake							
1971	-	-	-	-	-	-	-
1972	-	-	-	-	-	-	-
1973	-	-	7.4575	7.7643	7.3466	7.6872	7.4955
1974	8.0271	8.4577	7.4127	7.8504	7.2345	7.2826	7.5018
East Twin Lake							
1971	-	-	-	-	-	-	-
1972	-	-	-	-	-	-	-
1973	-	-	7.3699	8.0661	7.2452	6.8887	7.2460
1974	7.0596	8.2168	7.3405	7.7181	7.1212	7.4599	7.4699

Table 15. pH LAKE: Average pH Values in the Lake.

West Twin Lake							
1971	-	-	-	-	-	-	-
1972	-	-	-	-	-	-	-
1973	-	-	7.6143	7.7824	7.3455	7.6890	7.6285
1974	8.0449	8.4584	7.6337	7.8672	7.2464	7.2674	7.6299
East Twin Lake							
1971	-	-	-	-	-	-	-
1972	-	-	-	-	-	-	-
1973	-	-	7.5462	8.0850	7.2416	6.8445	7.3271
1974	8.0074	8.2128	7.5601	7.7505	7.1344	7.4586	7.6464

Table 16. [02-0XY] COL: Dissolved Oxygen Gas, Average Concentrations in the Column ($\mu g\ O_2/l$)..

West Twin Lake							
1971	-	-	3363.9	9234.8	544.0	5005.8	3617.3
1972	11670.7	14396.2	3702.5	7776.7	950.2	8483.0	7184.1
1973	10522.9	10932.5	3952.3	8247.5	454.7	8739.4	6735.2
1974	12770.6	12153.7	3864.8	8318.2	466.4	6991.4	6910.3
East Twin Lake							
1971	-	12983.3	3733.1	9045.1	614.2	8347.8	5029.5
1972	8970.3	12143.0	3582.9	8782.8	530.9	8057.1	5925.1
1973	9859.0	10674.6	4357.0	8622.2	722.6	9056.6	6705.0
1974	11318.1	11735.9	4191.1	8096.5	569.9	8150.9	6631.8

Table 17. [02-0XY] LAKE: Dissolved Oxygen Gas, Average Concentrations in the Lake ($\mu g\ O_2/l$).

West Twin Lake							
1971	-	-	5840.3	9287.6	566.9	5592.7	5802.1
1972	13265.3	14250.1	6070.5	7853.0	1016.1	8476.0	8766.1
1973	12018.9	11079.3	6579.9	8309.8	509.5	8676.6	8419.6
1974	13262.9	12245.5	6554.7	8405.7	621.6	7479.7	8533.6
East Twin Lake							
1971	-	13920.8	6135.2	9124.9	717.7	8598.6	6909.8
1972	11792.9	12596.2	5865.9	8827.7	588.7	8657.4	7930.5
1973	11563.9	11029.9	6752.5	8650.6	733.8	9038.5	8403.0
1974	12457.3	11986.7	6551.3	8226.9	669.4	8299.4	8330.7

Table 18. [TOT-P] COL: Unfiltered Total Phosphorus, Average Concentrations in the Column (μg P/l).

West Twin Lake Year	Winter Season	Spring Season	Summer Season	Epilim Only	Hypolim Only	Fall Season	All Seasons
1971	-	-	417.76	37.63	779.76	122.86	343.04
1972	152.79	170.82	257.09	57.21	494.35	132.20	204.85
1973	125.17	85.12	277.26	53.02	616.97	135.52	206.28
1974	122.31	78.04	229.90	48.35	512.02	125.79	173.67
East Twin Lake							
1971	-	-	289.24	29.76	687.23	66.81	226.56
1972	100.68	96.42	181.26	43.95	377.55	97.45	145.96
1973	112.54	77.53	187.98	38.07	430.73	72.27	147.77
1974	97.41	65.60	189.06	45.30	466.55	94.11	148.79

Table 19. [TOT-P] LAKE: Unfiltered Total Phosphorus, Average Concentrations in the Lake (μg P/l).

West Twin Lake							
1971	-	-	161.44	35.94	756.21	122.86	151.67
1972	134.84	171.55	108.63	55.55	483.66	127.53	122.17
1973	111.02	90.46	99.78	52.44	598.30	133.87	106.49
1974	136.72	79.78	78.66	47.17	483.40	121.08	96.83
East Twin Lake							
1971	-	-	104.39	28.94	671.17	66.78	93.79
1972	81.27	98.18	79.48	43.27	362.22	95.66	83.70
1973	95.44	75.74	74.28	37.58	416.93	76.41	78.61
1974	81.23	66.88	73.52	44.46	447.11	93.83	76.12

Table 20. [TOT-P DISS] COL: Filterable Total Phosphorus, Average Concentrations in the Column (μg P/l).

West Twin Lake							
1971	-	-	-	-	-	-	-
1972	-	-	-	-	-	-	-
1973	98.18	43.32	236.75	44.02	520.30	95.79	168.70
1974	84.13	26.73	210.88	31.88	492.15	112.12	147.97
East Twin Lake							
1971	-	-	-	-	-	-	-
1972	-	-	-	-	-	-	-
1973	79.59	26.05	195.21	69.85	414.34	105.43	145.08
1974	46.63	18.69	160.05	27.04	438.62	48.09	112.04

Table 21. [TOT-P DISS] LAKE: Filterable Total Phosphorus, Average Concentrations in the Lake (μg P/l).

West Twin Lake							
1971	-	-	-	-	-	-	-
1972	-	-	-	-	-	-	-
1973	80.82	43.82	88.57	40.58	494.71	99.27	84.40
1974	84.03	28.14	65.34	28.81	434.21	100.98	70.26
East Twin Lake							
1971	-	-	-	-	-	-	-
1972	-	-	-	-	-	-	-
1973	61.27	25.42	100.03	68.70	391.48	102.71	85.43
1974	30.32	19.82	49.56	24.74	399.16	46.58	42.25

Table 22. [P04-P DISS] COL: Filterable Ortho-Phosphate Phosphorus, Average Concentrations in the Column (μg P/l).

Year	Winter Season	Spring Season	Summer Season	Epilim Only	Hypolim Only	Fall Season	All Seasons
West Twin Lake							
1971	-	-	207.12	18.26	256.00	-	207.12
1972	106.66	14.64	181.69	11.02	405.53	84.16	134.40
1973	75.56	22.83	219.32	16.42	516.28	98.07	152.81
1974	57.78	6.53	178.85	12.01	451.15	84.52	119.20
East Twin Lake							
1971	-	-	142.62	17.40	326.36	-	142.62
1972	23.91	21.45	129.27	9.69	329.86	48.70	84.43
1973	54.05	14.40	147.30	8.20	381.35	35.04	101.83
1974	31.95	4.99	139.85	9.52	404.12	26.24	93.47

Table 23. [P04-P DISS] LAKE: Filterable Ortho-Phosphate Phosphorus, Average Concentration in the Lake (μg P/l).

West Twin Lake							
1971	-	-	69.76	13.32	392.12	-	69.76
1972	82.73	14.10	51.58	10.82	393.39	82.80	62.69
1973	56.63	18.92	58.64	15.75	507.92	96.93	60.47
1974	52.10	6.32	35.44	10.02	419.48	78.48	42.20
East Twin Lake							
1971	-	-	52.09	17.33	310.74	-	52.09
1972	8.25	23.59	36.12	9.59	316.84	48.20	29.73
1973	36.29	11.46	39.31	7.71	371.01	33.75	35.08
1974	13.82	5.09	31.55	8.54	385.34	25.58	24.62

Table 24. [TOT-N] COL: Total Nitrogen, Average Concentrations in the Column (mg N/l).

West Twin Lake							
1971	-	-	3.9208	-	-	-	3.9208
1972	0.4152	0.2431	1.8772	0.4150	3.8550	1.2810	1.2471
1973	0.9693	0.3599	2.0837	0.3788	4.7491	1.5051	1.5979
1974	-	-	-	-	-	-	-
East Twin Lake							
1971	-	-	3.0609	-	-	-	3.0609
1972	0.2332	0.1835	1.5944	0.3861	3.2371	1.2768	1.0622
1973	1.0470	0.5944	1.7621	0.3131	3.6645	1.1608	1.4239
1974	-	-	-	-	-	-	-

Table 25. [TOT-N] LAKE: Total Nitrogen, Average Concentrations in the Lake (mg N/l).

West Twin Lake							
1971	-	-	1.9289	-	-	-	1.9289
1972	0.4041	0.2616	0.7854	0.3987	3.6490	1.3300	0.7291
1973	0.8834	0.3187	0.7507	0.3528	4.4626	1.3716	0.8326
1974	-	-	-	-	-	-	-
East Twin Lake							
1971	-	-	1.3446	-	-	-	1.3446
1972	0.1832	0.1821	0.7089	0.3602	3.0889	1.2251	0.5825
1973	0.9542	0.5647	0.8065	0.3206	3.4919	1.1118	0.8395
1974	-	-	-	-	-	-	-

Table 26. [NH4-N] COL: Ammonium Nitrogen, Average Concentrations in the Column (mg N/l).

West Twin Lake Year	Winter Season	Spring Season	Summer Season	Epilim Only	Hypolim Only	Fall Season	All Seasons
1971	-	-	3.8199	0.4381	6.2037	-	3.8199
1972	0.1364	0.0808	1.8204	0.3816	3.7723	1.1561	1.1070
1973	0.7975	0.2440	2.0288	0.3296	4.6866	1.3330	1.4960
1974	1.0024	0.3390	2.1483	0.3184	4.4126	0.5185	1.4520
East Twin Lake							
1971		-	2.9723	0.3350	6.1336	-	2.9723
1972	0.1153	0.0734	1.5287	0.3429	3.1380	1.1605	0.9716
1973	0.8422	0.4023	1.7011	0.2627	3.6008	0.9450	1.3090
1974	0.8494	0.2751	1.6661	0.2971	4.7045	0.0776	1.2076

Table 27. [NH4-N] LAKE: Ammonium Nitrogen, Average Concentrations in the Lake (mg N/l).

West Twin Lake							
1971	-	-	1.8383	0.4371	5.8251	-	1.8383
1972	0.0995	0.0951	0.7440	0.3654	3.5696	1.2061	0.5886
1973	0.7023	0.2191	0.7009	0.3051	4.4019	1.2031	0.7337
1974	0.9996	0.3129	0.7647	0.3178	3.8305	0.4882	0.6994
East Twin Lake							
1971	-	-	1.2649	0.3342	5.8155	-	1.2649
1972	0.0652	0.0731	0.6594	0.3175	2.9913	1.1113	0.5004
1973	0.7360	0.3668	0.7517	0.2694	3.4298	0.8886	0.7247
1974	0.6193	0.2883	0.5082	0.2978	4.1861	0.0757	0.4589

Table 28. [NO3-N] COL: Nitrate Nitrogen, Average Concentrations in the Column (mg N/l).

West Twin Lake							
1971	-	-	.03363	.07084	.00089	-	.03363
1972	.27878	.16230	.05676	.03340	.08273	.12494	.14008
1973	.17181	.11592	.05490	.04922	.06246	.17208	.10192
1974	-	-	-	-	-	-	-
East Twin Lake							
1971	-	-	.02953	.06117	.00000	-	.02953
1972	.11789	.11016	.06567	.04318	.09910	.11631	.09064
1973	.20480	.19212	.06095	.05043	.06369	.21583	.11489
1974	-	-	-	-	-	-	-

Table 29. [NO3-N] LAKE: Nitrate Nitrogen, Average Concentrations in the Lake (mg N/l).

West Twin Lake							
1971	-	-	.05917	.07100	.00265	-	.05917
1972	.30458	.16650	.04142	.03327	.07937	.12391	.14045
1973	.18106	.09957	.04980	.04767	.06072	.16846	.09887
1974	-	-	-	-	-	-	-
East Twin Lake							
1971	-	-	.04908	.06126	.00000	-	.04908
1972	.11804	.10898	.04953	.04274	.09759	.11383	.08211
1973	.21822	.19791	.05478	.05115	.06206	.22323	.11478
1974	-	-	-	-	-	-	-

Table 30. ALK COL: Average Alkalinities in the Column (mg $CaCO_3$/l).

Year	Winter Season	Spring Season	Summer Season	Epilim Only	Hypolim Only	Fall Season	All Seasons
West Twin Lake							
1971	-	-	-	-	-	-	-
1972	-	-	-	-	-	-	-
1973	-	-	122.29	102.38	149.17	116.76	121.16
1974	120.36	116.54	118.91	92.32	150.33	105.05	116.54
East Twin Lake							
1971	-	-	-	-	-	-	-
1972	-	-	-	-	-	-	-
1973	-	-	120.04	95.80	144.83	107.65	118.02
1974	115.79	110.61	115.87	95.69	141.31	104.49	114.14

Table 31. ALK LAKE: Average Alkalinities in the Column (mg $CaCO_3$/l).

West Twin Lake							
1971	-	-	-	-	-	-	-
1972	-	-	-	-	-	-	-
1973	-	-	108.07	102.30	146.70	116.08	109.70
1974	119.55	116.98	100.16	91.99	147.24	104.53	106.29
East Twin Lake							
1971	-	-	-	-	-	-	-
1972	-	-	-	-	-	-	-
1973	-	-	104.79	95.41	142.40	105.67	104.94
1974	114.26	110.87	102.06	95.36	138.80	104.43	105.42

Table 32. [SO4] COL: Sulfate, Average Concentrations in the Column (mMoles/l).

West Twin Lake							
1971	-	-	-	-	-	-	-
1972	-	0.294	0.399	0.446	0.320	0.374	0.375
1973	0.490	0.416	0.362	0.433	0.252	0.433	0.405
1974	0.523	0.533	0.357	0.423	0.278	0.357	0.408
East Twin Lake							
1971	-	-	-	-	-	-	-
1972	-	0.227	0.315	0.390	0.232	0.262	0.290
1973	0.444	0.387	0.331	0.398	0.221	0.388	0.366
1974	0.470	0.450	0.327	0.359	0.260	0.284	0.363

Table 33. [SO4] LAKE: Sulfate, Average Concentrations in the Lake (mMoles/l).

West Twin Lake							
1971	-	-		-	-	-	-
1972	-	0.305	0.448	0.446	0.324	0.376	0.507
1973	0.480	0.416	0.417	0.434	0.263	0.434	0.433
1974	0.526	0.527	0.400	0.424	0.290	0.374	0.436
East Twin Lake							
1971	-	-	-	-	-	-	-
1972	-	0.232	0.362	0.391	0.237	0.260	0.322
1973	0.440	0.380	0.376	0.399	0.229	0.383	0.390
1974	0.470	0.450	0.353	0.360	0.269	0.278	0.378

C. Biological

1. Phytoplankton

a. Chlorophyll. Chlorophyll A concentrations are summarized in Tables 34 and 35.

b. Primary Production. We have adopted polarographic and titrimetric methods of measuring phytoplankton potential productivity rather than *in situ* methods (Long, 1971). Maximum potential productivity *in situ* was estimated by correcting laboratory data for phytoplankton density, day length, and epilimnetic temperature (assuming a Q_{10} of 2) (Talling, 1957, 1965). In 1970, at the height of the bluegreen algal bloom the maximum productivity *in situ* was estimated to be 3400 mg. C/m^2/day for ETL. In 1974 the potential productivity estimate of each lake was lower than that found in 1970 (Table 36).

The production of macrophytes was estimated in 1972 (Rogers, 1974) by the harvest method, using SCUBA. The rate, from 15 April to 1 July was (mg.C/m^2/day) 375 for ETL, 267 for WTL, or about 10% to 30% of the maximum rate of the plankton. Reduced growth rates were correlated with plankton blooms. No measurements for 1974 are available; we estimate that macrophyte production at least equalled the 1972 rate, thus bringing it up to about 50% of the net community metabolism of the lakes.

Oxygen deficits have been used to estimate productivity. The contributions of allochtonous and autochthonous production are not easily separated, and, as pointed out by Edmondson (1966) sedimentation during periods of blue-green blooms is not rapid. In the Twin Lakes, some of this production may leave the lakes because of the low residence time. The deficits have declined since diversion, particularly in ETL which does not have the canals and the heavy import from that area of the watershed (Table 37).

c. Algal Assays. We have monitored acid and alkaline phosphatase in limnetic waters of both lakes from 1972-74. *Aphanizomenon flos-aquae* appears to produce it adaptively, particularly in late summer, and both cell volume and potential productivity increase following the appearance of alkaline phosphatase (Heath and Cooke, 1974). This alga (the dominant species in each year) appears to be phosphorus-limited in August.

Levels of total PO_4-P at spring circulation are high (Table 40) and the relationship to mean euphotic zone Chlorophyll A is not as strong as most lakes reported in Dillon (1974), particularly in 1972-73. In 1974, the summer chlorophyll was much more closely related to spring phosphorus levels.

We conclude that these lakes have been phosphorus-limited primarily in late summer, as evidenced by the phosphatase studies. They appear to be moving towards more general phosphorus limitation as loading declines.

d. Identification and Count. See Table 38 and Figure 3.

Table 34. [CHLOR-A] COL: Chlorophyll a, Average Concentrations in the Column (mg Chl/m^3).

Year	Winter Season	Spring Season	Summer Season	Epilim Only	Hypolim Only	Fall Season	All Seasons
West Twin Lake							
1971	-	-	31.280	28.018	37.436	-	31.280
1972	30.107	66.324	67.838	21.849	107.011	11.462	47.584
1973	14.126	34.870	53.641	12.246	97.310	8.165	36.619
1974	20.083	36.916	42.741	21.651	59.256	7.967	32.328
East Twin Lake							
1971	-	-	28.986	10.162	37.201	-	28.986
1972	15.183	40.992	26.364	23.500	16.044	19.649	23.201
1973	11.072	27.372	25.988	16.021	21.747	15.083	22.017
1974	19.369	30.661	33.000	19.076	26.969	23.567	29.527

Table 35. [CHLOR-A] LAKE: Chlorophyll a, Average Concentrations in the Lake (mg Chl/m^3).

Year	Winter Season	Spring Season	Summer Season	Epilim Only	Hypolim Only	Fall Season	All Seasons
West Twin Lake							
1971	-	-	26.791	28.249	38.737	-	26.791
1972	53.903	68.742	37.770	21.551	108.878	11.233	39.971
1973	19.101	43.987	24.888	12.110	94.954	8.257	22.939
1974	28.250	40.030	29.944	22.106	62.435	9.164	27.655
East Twin Lake							
1971	-	-	20.712	10.191	40.111	-	20.712
1972	20.719	41.840	28.555	23.521	17.222	20.208	26.120
1973	15.006	33.400	23.011	16.010	24.507	14.958	21.636
1974	24.199	31.443	28.869	19.429	31.556	21.530	27.757

Table 36. Estimated Net Plankton Community Photosynthesis (mg C/m^2/day).

Date	West Twin Lake	East Twin Lake
27 June 1974	1758.8	117.2
4 July 1974	298.1	777.2
11 July 1974	135.1	459.6
20 July 1974	439.8	844.6
2 Aug. 1974	387.5	284.2
9 Aug. 1974	433.8	362.3
mean	575.5	474.2

Table 37. Oxygen Deficit (mg $O_2/cm.^2$/day)

Date	West Twin Lake	East Twin Lake
1970	0.0525	0.0400
1971	--	0.0740
1972	0.0523	0.1150
1973	0.0558	0.0362
1974	0.0223	0.0300

Table 38. Major Phytoplankton Species.

1972

	West Twin Lake - 0.1m.	East Twin Lake - 0.1m.
Summer	Aphanizomenon flos-aquae* Sphaeroecystis Schroeteri Cyclotella sp. Melosira granulata	Aphanizomenon flos-aquae* Anabaena limnetica
Fall	Aphanizomenon flos aquae* Sphaeroecystis Schroeteri Asterionella formosa	Aphanizomenon flos-aquae* Asterionella formosa
Winter	Asterionella formosa	Aphanizomenon flos-aquae
Spring	Asterionella formosa* Aphanizomenon flos-aquae Miscellaneous greens	Asterionella formosa* Aphanizomenon flos-aquae Fragilaria crotonensis

1974

	West Twin Lake	East Twin Lake
Summer	Aphanizomenon flos-aquae* Anabaena limnetica* Microcystis aeruginosa	Aphanizomenon flos-aquae* Anabaena limnetica* Microcystis aeruginosa Stephanodiscus niagarae
Fall	Aphanizomenon flos-aquae Anabaena limnetica*	Aphanizomenon flos-aquae*
Winter	Asterionella formosa* Fragilaria crotonensis	Stephanodiscus niagarae* Aphanizomenon flos-aquae* Asterionella formosa
Spring	Fragilaria crotonensis* Asterionella formosa	Aphanizomenon flos-aquae* Stephanodiscus niagarae Fragilaria crotonensis

*=dominants

Table 39. Species of Microcrustacea Identified from the East and West Twin Lakes, 1969-1970.

Leptodora kindtii (Focke)**
Diaphanosoma leuchtenbergianum Fischer
Daphnia ambigua Scourfield
D. galeata Sars Mendotae Birge
D. retrocurva Forbes
D. pulex Leydig, Richard**
Simocephalus exspinosus (Koch)**
S. serrulata (Koch)**
Ceriodaphnia reticulata (Jurine)

Bosmina longirostris (O. F. Muller)

Camptocercus rectirostris (Schodler)*
Leydigia quadrangularis (Leydig)**
Alona guttata Sars

A. Costata Sars**
A. quadrangularis (O. F. Muller)**
Pleuroxus procurvas Birge
P. denticulatus Birge*
Chydorus sphaericus (O. F. Muller)
Diaptomus reighardi Marsh
Orthocyclops modestus (Herrick)
Eucyclops speratus (Lilljeborg)**
Tropocyclops prasinus mexicanus Kuefer
Cyclops bicuspidatus thomasi S. A. Forbes
Mesocyclops edax (S. A. Forbes)
Ergasilus chautauquaensis Fellows

* = East Twin Only
** = West Twin Only

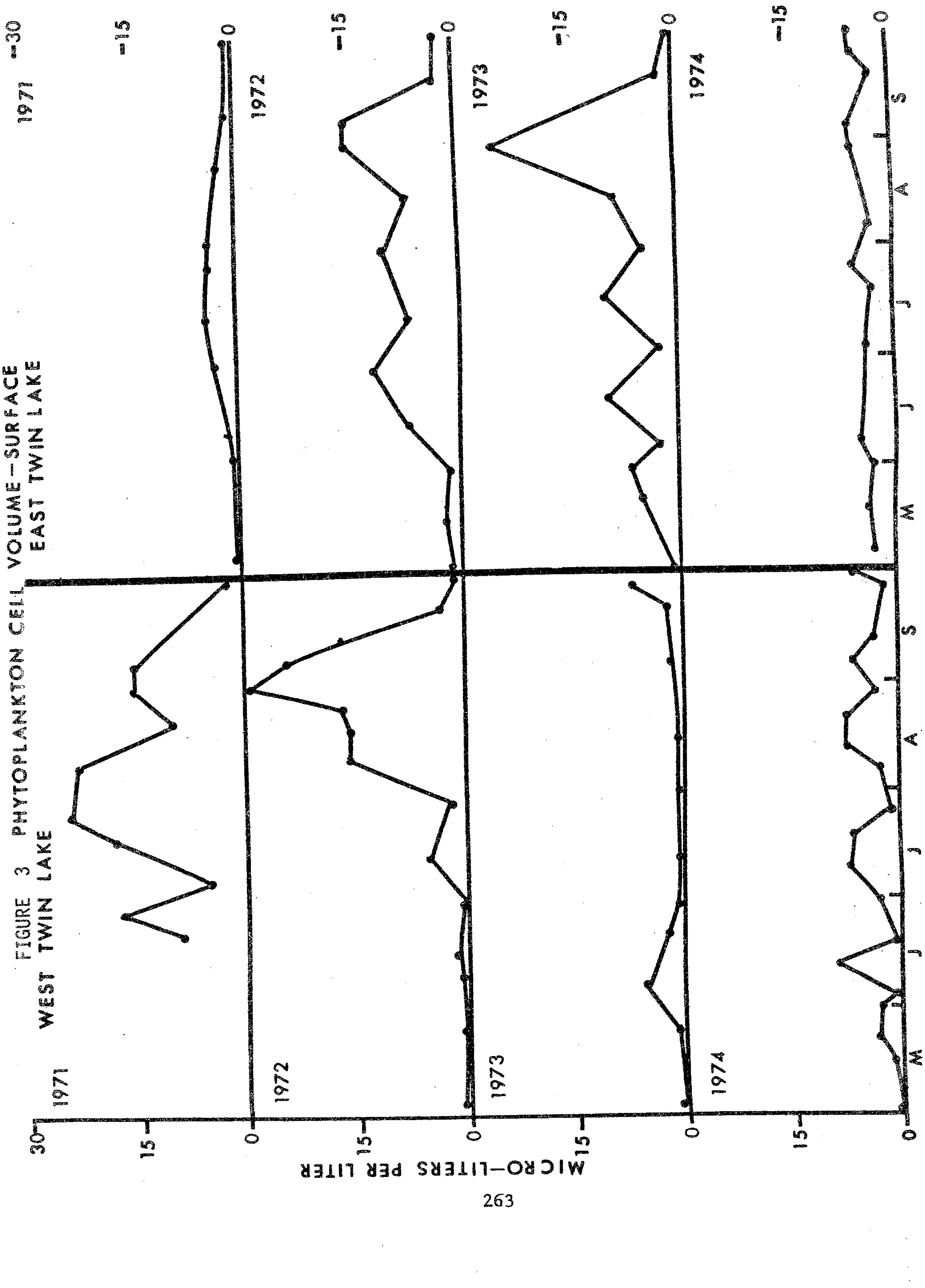


FIGURE 3 PHYTOPLANKTON CELL VOLUME—SURFACE
WEST TWIN LAKE
EAST TWIN LAKE
1971
1972
1973
1974
30
15
0
-30
-15
M
J
J
A
S
MICRO-LITERS PER LITER

2. Zooplankton. Limnetic microcrustacea were sampled on 26 dates between May 1969-May 1970 by Heinz (1971). The species are listed in Table 38. Epilimnetic density ranged from 85/liter in May to 1/liter in late summer. The periods of greatest density were May and December (WTL) or January (ETL), with a small bloom in August. The dominant species were *Daphnia galeata*, *Bosmina longirostris*, *Cyclops bicuspidatus thomasi*, and *Mesocyclops edax*. The species composition of ETL and WTL strongly resembled that of littoral communities, and the mean number of species/sample in limnetic waters was 2-3 times that of other lakes, indicating in both instances that the littoral had a strong influence on the limnetic waters. Species composition and abundance were most alike at spring circulation when *Cyclops* dominated. The lakes diverged in summer: WTL was dominated by *D. galeata*, *Bosmina*, and *Diaptomus reighardi*, but ETL by just *D. galeata*. Fall circulation was dominated by *Daphnia* and *Cyclops*, and winter stagnation primarily by *Cyclops* in both lakes.

3. Bottom Fauna. Bottom fauna are rare, presumably due to the long anoxic period. Macroinvertebrates of ETL, identified primarily to genus, as available in Wilbur (1974).

4. Fish. The fish are dominated by Centrarchidae, primarily bluegill, black crappie, pumpkinseed, and largemouth bass. Fish size has declined in recent years.

5. Bacteria. Fecal coliform bacteria in surface waters fell from 200 colonies/100 ml. (swim beach on WTL was closed in 1970 and 1971) before diversion to near 0 in surface waters to 10 colonies/100 ml. in deep water in 1972 and 1973. Total bacteria/100 ml. ranged from 600-44,000 in both lakes in 1972, with the highest counts in the metalimnion and at bottom. Surface inflows were highly contaminated with fecal coliforms before diversion, particularly those flowing into WTL, where samples contained 90,000 colonies/100 ml. or more (1971 and 1972). In 1973, counts dropped to 0-600. Groundwater samples were not as contaminated as surface drainage samples, except for wells located directly below leach fields where colony counts/100 ml. ranged from 30-5000.

6. Bottom Flora. No studies of bottom flora have been conducted.

7. Macrophytes. This community ranks with at least equal importance to algae as a nuisance. The distribution and biomass was surveyed in 1972, using SCUBA (Rogers, 1974). The species are: *Ceratophyllum demersum* L., *Najas guadulapensis* (Spreng) Magnus, *Elodea canadensis* Michx., *Nuphar advenum* (Ait) Ait., *Potamogeton crispus* L., and *Chara vulgaris* L (Chlorophyta). Macrophytes covered about 28% of the lake area in ETL and 23% in WTL. *P. crispus* was dominant in early summer, *N. guadulapensis* (Etl) and *C. demersum* (WTL) during the rest of the season. Total dry weight in ETL declined from 200 kg. (July) to 100 kg. (September); in WTL it declined from 80 kg. (July) to 50 kg. (August). The total PO_4-P content in ETL was about 2.5 kg., in WTL about 1.5 kg. *Nuphar* contained about 6 kg. PO_4-P.

IV. NUTRIENT BUDGETS SUMMARY

A. Phosphorus

1. West Twin Lake

source	1972 (kg./year)	1973 (kg./year)	1974 (kg./year)
Waste Discharges	0.00	0.00	0.00
Land Runoff	50.97	50.08	35.92
Precipitation	7.45	6.60	9.11
Ground Water	6.96	20.93	32.12
Surface Streams	55.14	25.43	13.73
Total Inflow	120.52	103.04	90.87
Total Outflow	106.16	79.25	131.69*

2. East Twin Lake

	1972	1973	1974
Waste Discharges	0.00	0.00	0.00
Land Runoff	66.12	34.53	24.97
Precipitation	6.20	5.24	7.17
Ground Water	5.31	17.25	19.60
Surface Streams	114.05	81.52	132.86
Total Inflow	191.00	138.22	184.62
Total Outflow	123.40	132.80	144.47

*Apparently due in part to sewer pipe leak in outflow of WTL (inflow of ETL).

B. Nitrogen (Total Combined Inorganic) kg./year.

1. West Twin Lake

	1972	1973
Waste Discharges	0.0	0.0
Land Runoff	2067.5	1957.5
Precipitation	1146.5	937.6
Ground Water	1301.9	1439.1
Surface Streams	937.4	763.9
Total Inflow	5453.3	5098.1
Total Outflow	3845.6	2048.6

2. East Twin Lake

	1972	1973
Waste Discharges	0.0	0.0
Land Runoff	2714.2	1797.3
Precipitation	897.5	737.0
Ground Water	979.0	963.0
Surface Streams	3845.0	1688.2
Total Inflow	8435.7	5185.5
Total Outflow	6408.6	4371.4

VI. DISCUSSION

East and West Twin Lakes are early eutrophic and mesotrophic, respectively, with the trend in both (except ETL in 1974 after the sewer leak) towards mesotrophy after sewage diversion. Evidence for this is based primarily upon changing characteristics of the plankton. If macrophytes are included, the lakes are eutrophic. Briefly the basis for this is:

1. The oxygen deficits are lower than often found in eutrophic lakes (Table 37).
2. While Aphanizomenon flos-aquae now dominantes the plankton, an increasing fraction of the community is diatoms. Mean cell volume (from Figure 3) for 1972-74 for WTL ranged from 1.05-5.86; for ETL 3.44-6.59 μl./l. Vollenweider (1968) suggests that 3-5 μl./l. might be the borderline between mesotrophy and eutrophy. Mean summer photic zone chlorophyll A (Table 34) is on the low end of Sakamoto's (1966) range of 5-140 mg. ChlA/m^3 for eutrophic lakes. Maximum net plankton community photosynthesis (Table 36) has dropped, since diversion, from 3400 (ETL) to a mean of 474 (ETL) and 575 mg.C/m^2/day (WTL). These latter values are in the range of borderline eutrophic lakes (Vollenweider, 1968).
3. Secchi disc transparency (Table 12) averages are like those of moderately eutrophic lakes.

How well does the degree of eutrophy, as assessed above, compare to that predicted by the loading models of Vollenweider (1968, 1973) and Dillon (1974)? Data for the models are summarized in Table 40. The log phosphorus loading--log mean depth (1968) model indicates the lakes to be more eutrophic than they are, based on plankton data. The 1973 model (log phosphorus loading-log mean depth/water residence (T_W)) indicates the lakes to be moving towards mesotrophy, with WTL now (1974) mesotrophic and ETL eutrophic. This position is supported by the evidence about plankton presented above, and is due in large part to the low water residence time. Dillon's model places both lakes well into the eutrophic range, which they are not if only plankton-based indicators are employed.

The Vollenweider (1973) model accurately predicts the degree of eutrophication of the Twin Lakes, as described by characteristics related primarily to plankton production. However, nearly half the productivity of the lakes is due to macrophytes, and 25% of the area is littoral. The lakes are in fact of poorer quality, particularly from the view of the lake user than might be indicated by the mesotrophic label. For planning or management purposes for lakes and watersheds of this type, models based primarily on plankton characteristics may not be applicable, or are at least insensitive to the effects of very low mean depth. We suggest a fruitful approach will include estimates of total community productivity for lakes with a mean depth less than 10 m. and a ratio of deep to shallow areas of less than 10 into classification models. Perhaps the 1968 model, which includes a factor primarily related to plankton biomass or productivity (phosphorus loading) and a factor primarily related to macrophyte growth (mean depth) is most applicable to shallow lakes, and the other models most applicable to deeper lakes.

Table 40

Summary Hydrological and Limnological Data for Lake Classification Models

A. West Twin Lake	1972	1973	1974
1. Lake Area (A_o), ha.	34.015	34.015	34.015
2. Lake Volume (V), m^3	14.99×10^5	14.99×10^5	14.99×10^5
3. Mean Depth ($\overline{Z}$), M	4.34	4.34	4.34
4. Annual Outflow (Q_o), m^3	916835	826235	1461576
5. Annual Inflow (Q_i), m^3	1218121	1177684	1640332
6. Water Residence Time (T_w), yr. V/Q	1.64	1.81	1.03
7. Flushing Rate (p), $Yr.^{-1}$ $1/T_w$, Q/V	0.61	0.55	0.98
8. Areal Water Loading ($\overline{Z}/T_w$)	2.65	2.40	4.23
9. Phosphorus Loading (L), gms. $P/m^2/yr.$	0.354	0.303	0.267
10. Outflow Phosphorus Amt. ($[P]_a$), kg./yr.	106.16	79.29	131.69*
11. Inflow Phosphorus Amt. ($[P]_i$), kg./yr.	120.52	103.04	90.87
12. Retention Coefficient (R_{exp}) $R_{exp} = 1 - \frac{Q_o[P]_a}{Q_i(P]_i}$	0.337	0.460	1.291
13. Ice Out, Mean Total PO_4-P Conc. (mg./m^3)	212.0	118.0	94.0
14. Spring Circulation Period, Mean Total PO_4-P Conc. (mg./m^3)	137.0	85.0	78.0
15. Mean Summer Photic Zone Chlorophyll A Conc. (mg/m^3)	28.58	18.57	23.54
16. $\frac{L\ (1 = R_{exp})}{p}$	0.205	0.297	-0.079*
17. Area of Land Drainage (Ad), ha.	184	184	184
18. Basin population (C)	-	-	1124
19. Per Capita Phosphorus Discharge (Ec)	-	. -	0.08kg/ person/year
20. Total Phosphorus Import ($[P]_i$)/Ad/year (mg. $P/M^2/yr$)	65.5	56.0	49.30

*Apparently an artifact. $[P]_a$ [Pi] in 1974 was partly caused by leaking sewer line which crosses lake outlet.

Table 40

Summary Hydrological and Limnological Data for Lake Classification Models

	B. East Twin Lake	1972	1973	1974
1.	Lake Area (A_o), ha.	26.88	26.88	26.88
2.	Lake Volume (V), m^3	13.50×10^5	13.50×10^5	13.50×10^5
3.	Mean Depth ($\bar{Z}$), M	5.03	5.03	5.03
4.	Annual Outflow (Q_o), m^3	1700006	1444921	2307490
5.	Annual Inflow (Q_i), m^3	1934340	1678127	2458930
6.	Water Residence Time (T_w), Yr. V/Q	0.79	0.93	0.58
7.	Flushing Rate (p), $Yr.^{-1}$ $1/T_w$, Q/V	1.26	1.07	1.71
8.	Areal Water Loading ($\bar{Z}/T_w$)	6.37	5.41	8.63
9.	Phosphorus Loading (L), gms. $P/m^2/yr.$	0.711	0.514	0.687
10.	Outflow Phosphorus Amt. ($[P]_a$), kg./yr.	123.4	132.8	144.5
11.	Inflow Phosphorus Amt. ($[P]_i$), kg./yr.	191.0	138.2	184.6
12.	Retention Coefficient (R_{exp}) $R_{exp} = 1 - \frac{Q_o\,[P]_a}{Q_i\,[P]_i}$	0.432	0.173	0.265
13.	Ice Out, Mean Total PO_4-P Conc. (mg./m^3)	118.0	75.0	77.0
14.	Spring Circulation Period, Mean Total PO_4-P Conc. (mg./m^3)	94.0	65.0	65.0
15.	Mean Summer Photic Zone Chlorophyll A Conc. (mg./m^3)	26.08	19.14	18.57
16.	$\frac{L\,(1 - R_{exp})}{p}$	0.321	0.397	0.295
17.	Area of Land Drainage (Ad), ha.	255	255	255
18.	Basin Population (C)			1510*
19.	Per Capita Phosphorus Discharge (Ec)			.122kg/ capita/year
20.	Total Phosphorus Import ($[P]_i$)/Ad/yr. (mg.P/M^2/yr)	74.9	54.20	72.39

*Includes West Twin Lake sub-watershed (184 ha.) since WTL drains into East Twin Lake. Drainage areas obtained by subtracting lake areas from watershed area. Small lakes of watershed = 18 hectares. Watershed area = 334 hectares.

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SECTION V - OREGON

WALDO LAKE, OREGON

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INTRODUCTION

Waldo Lake, the second largest lake in Oregon, is one of the most pristine lakes on record. Located near the summit of the Cascade Mountains, the lake was accessible only by foot or by a primitive road system until 1969, when a paved road was constructed linking it with the Willamette Highway. Three large campgrounds have been developed on the east side of the lake by the U.S. Forest Service, and the lake has become subject to greatly increased summer recreational use over the past six years. The Environmental Protection Agency began limnological studies in 1969 to investigate possible effects of development on this unique lake. Except for the summers of 1969 and 1970, work has been confined to one annual visit, in August or September, from 1970 to 1974. Results from 1969 and 1970 have been reported by Malueg et al. (1972).

GEOGRAPHICAL DESCRIPTION OF WALDO LAKE

Waldo Lake is located at latitude 43°43'N, longitude 122°03'W, 1650 m above mean sea level on the western slope of the Cascade Mountains (Fig. 1). Precipitation amounts are moderately heavy, occurring for the most part in the non-summer months (Table 1). Average yearly precipitation is approximately 180 cm. Evaporation is not measured at the lake, but is estimated as approximately 109 cm annually from NOAA measurements in Detroit and Wickiup Reservoir. Between 1969 and 1973, yearly extreme temperatures varied between -30° and 38°C. Mean temperature for the period 1969-1972 was 6.0°C. (All precipitation, evaporation, and temperature information is from U.S. Department of Commerce, NOAA, Environmental Data Service, Climatological Data).

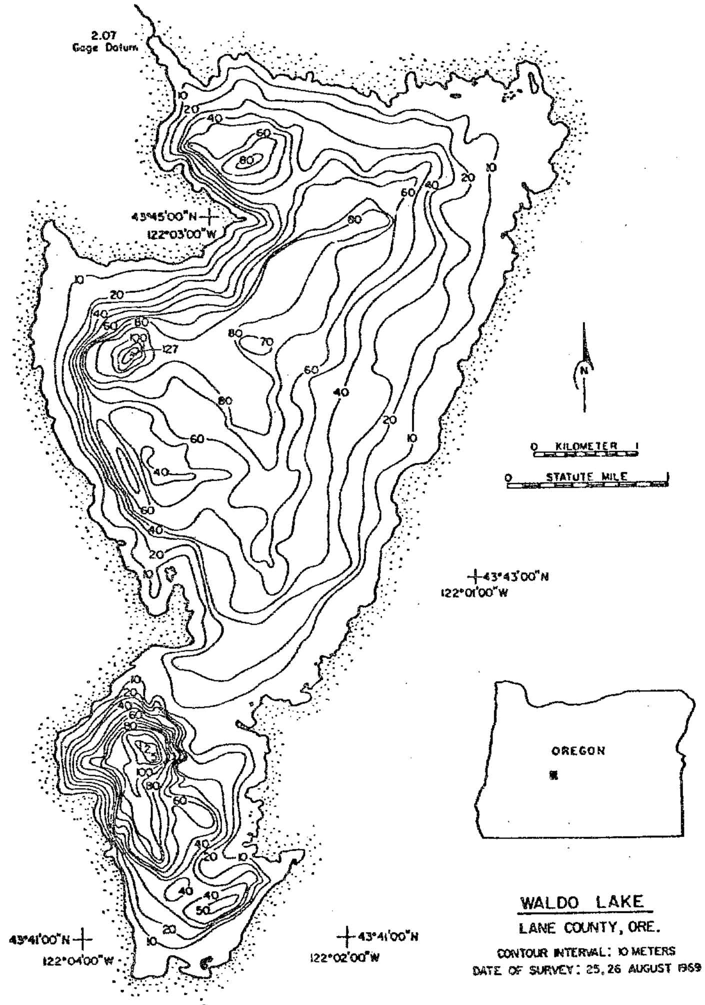

Figure 1. Bathymetric map of Waldo Lake.

Table 1

Precipitation Record, Waldo Lake

Date	Ppt'n Since Last Reading Inches	Cm	Date	Ppt'n Since Last Reading Inches	Cm
Aug. '65	Gage installed		July '69	76.85	195.2
July '66	54.20	137.7	Oct.	---	---
Aug.	0.20	0.5	Apr. '70	47.35	120.3
Nov.	12.80	32.5	June	10.20	25.9
July '67	---	---	July	0.45	1.1
Oct.	3.70	9.4	Sept.	3.35	8.5
Jan. '68	18.80	47.8	April '71	79.70	202.4
May	25.10	63.8	Aug.	9.95	25.3
Aug.	5.15	13.1	Oct. 7	3.00	7.6
Sept.	---	---	Oct. 20	1.80	4.5
Oct.	4.05	10.3	June '72	74.25	188.6

The lake is surrounded by coniferous forest, predominantly Douglas fir, pine, and hemlock. A large meadow lies at the south end. The soil mantle is generally less than 1 m thick, consisting of moderately weathered volcanic materials and glacially rounded boulders up to 1.5 m in diameter. Underlying bedrock is principally hard basalt. Numerous intermittent streams, unchannelled runoff, and direct precipitation constitute the lake's principal sources of water.

No permanent human population exists around the lake; however, vacationers utilize camping facilities developed by the U. S. Forest Service on the east side and the numerous hiking trails which radiate from the area. This use is for the most part confined to the period July 15-September 15. Fish production is low, and fishing is of relatively minor importance. In 1973 the Forest Service estimated a total of 27,900 vistor days for the campsites and 2100 additional, non-camping visitor days by boaters and swimmers. Figures for 1972 were 16,400 and 2,500 visitor days. They estimate that use during 1971, 1970, and 1969 (when the campgrounds were opened) was comparable to 1972. Drinking water for two of the campgrounds is taken from the lake. Estimated daily water usage during the 1973 season was 45 m^3, with a season's total of 2700 m^3.

Sewage and effluent discharge is via septic tank drain fields and drain seepage from outdoor faucets. The discharge volumes are not measured and the quantity and chemical quality of the ground water entering the lake is not known. Ground water and effluent movement away from one septic tank drain field was measured in the summer of 1970 (Tilstra et al, 1973), but direct entrance of the effluent into the lake was not demonstrated.

MORPHOMETRY AND HYDROLOGY

The combined area of Waldo Lake and its watershed is 7900 ha (79 km^2). The maximum length of the lake is 9.6 km on an approximate N-S axis; maximum width is 4.3 km, and the surface area is 2700 ha. Maximum depth is 127 m, mean depth 35.6 m, and the volume, 9.5×10^8 m^3 (0.95 km^3). The greatest depth (127 m) occurs in a restricted hole in the northwest part of the lake; this is closely matched by a 125 m depression near the south end.

Thermal stratification has been observed each year, with an epilimnion of 5-10 m thickness. The ratio of epilimnion to hypolimnion (E/H) is roughly 0.3. Sufficient data are not at hand to permit determination of the duration of stratification; it has been estimated at five months.

Little information exists on the nature of the lake sediments. A great deal of the bottom is rocky. Sediments taken from the 127 m location contained 0.2% total P, 0.9% total N, and 5.1% total C (dry wt) (Malueg et al). However, the areal extent of the sediments is not known.

There are no permanent influent streams. The U. S. Geological Survey maintains a recording gage on the outlet, the origin of the North Fork of the Willamette River. The average outflow for the period 1969-1973 was 44.7×10^6 m^3/yr (1.42 m^3/sec). The retention time of the lake, calculated as volume/outflow, is 21.2 years.

LIMNOLOGICAL CHARACTERIZATION

Limnological observations are made at nine stations, including the two deep holes. Water quality differences from station to station are slight, and in this report only data from the North Hole (the deepest point in the lake) are reported.

PHYSICAL CHARACTERISTICS

As noted previously the lake stratifies thermally. Midsummer surface temperatures range between 14° and 18° C; minimum deep water temperatures of 3.9° and 3.8° C were observed in 1972 and 1974, respectively. Temperatures from the North Hole for 1969-1974 are listed in Table 2.

Table 2
North Hole
Temperature, °C
August

Depth, m	1969	1970	1971	1972	1973	1974
0	16.6	18.0	17.5	14.1	16.5	---
5	16.6	17.9	16.8	13.9	16.0	15.4
10	13.1	17.2	12.0	13.8	13.6	11.7
15	9.8	11.6	10.0	10.6	10.5	8.6
20	7.9	9.3	8.0	7.3	9.1	7.3
25	6.7	7.7	7.8	7.1	8.0	6.1
30	6.1	6.5	6.4	6.4	7.4	5.6
40	5.3	5.6	5.5	5.2	6.3	5.0
50	5.0	5.1	5.1	4.3	5.7	4.3
60	4.6	4.8	4.9	4.1	5.3	4.0
70	4.4	4.6	4.6	3.9	4.8	4.0
80	4.3	4.6	4.5	3.9	4.6	3.8
90	4.2	4.5	4.5	3.9	4.4	3.8
100	4.1	4.4	4.4	3.9	4.4	3.8

Specific conductance of the lake waters is extremely low, ranging between 2.0 and 5.0 μmhos/cm at 25°C (Table 3). This reflects the very dilute concentration of all solutes for which determinations have been made; total solids as determined by Malueg et al were nearly undetectable at 3 mg/l.

Table 3
Specific Conductance, μmhos/cm @25°C
August

Depth, m	1969	1970	1971	1972	1973	1974
0	3.2	--	3.4	3.0	4.0	5.0
20	3.2	--	3.1	--	3.9	4.0
40	3.0	--	2.9	3.0	3.8	4.0
60	2.9	--	2.9	3.0	3.6	--
80	3.0	--	2.9	3.0	3.5	4.0
100	2.9	--	2.8	3.0	3.5	4.0

Measurements taken with a white 20-cm secchi disc have shown considerable variation during our period of record. In 1969 values from 24.0 to 32.5 m were obtained between June and September. Observations since 1969 have been as follows:

1970 -- 27.5 m
1971 -- (missing)
1972 -- 25.0 m
1973 -- 23.0 m
1974 -- 35.0 m

Fluctuations in secchi disc transparency appear to be caused by meteorological conditions and coniferous pollen rather than by the presence of phytoplankton.

CHEMICAL CHARACTERISTICS

Total alkalinity (Table 4) ranges between 1.0 and 3.0 mg/l (as $CaCO_3$), with essentially uniform distribution from surface to bottom. Accurate determinations of pH are difficult because of the extremely low buffering capacity and dissolved solids content of the water. Levels of pH are consistently less than 7.0 except for the 1972 measurements, which are suspect (Table 5). Measurements in 1974 were made with a Hydrolab Surveyor Model 6D in situ water quality analyzer, and would be expected to be of greater accuracy than earlier determinations made in vitro.

Table 4
North Hole
Total Alkalinity
mg/l $CaCO_3$
August

Depth, m	1969	1970	1971	1972	1973	1974
0	2.0	2.0	--	2.0	1.0	1.0
20	1.0	2.0	2.0	--	1.0	2.0
40	1.0	2.0	2.0	2.0	1.0	2.0
60	1.0	2.0	2.0	2.0	1.0	3.0
80	1.0	2.0	1.0	2.0	3.0	2.0
100	1.0	2.0	2.0	2.0	1.0	3.0

Table 5
North Hole
pH
August

Depth, m	1969	1970	1971	1972	1973	1974
0	5.5	6.6	6.3	7.1	--	6.4
20	5.4	6.3	6.3	--	--	5.6
40	5.3	6.2	6.3	7.1	--	6.0
60	5.3	6.2	6.3	7.2	--	5.0*
80	5.2	6.4	6.1	7.1	--	5.3*
100	5.2	6.3	6.0	6.8	--	5.1*

*Data from South Hole

Dissolved oxygen exhibits an orthograde distribution as would be expected in such an extremely unproductive lake. Epilimnetic values are usually about 2 mg/l lower than at greater depths. Percent saturation varies between 89 and 114, and is usually very near 100 percent. Dissolved oxygen distribution is summarized in Table 6.

Table 6
North Hole
Dissolved Oxygen, mg/l
August

Depth, m	1969	1970	1971	1972	1973	1974
0	8.1	7.7	8.2	8.5	8.5	8.1
20	10.3	10.4	9.8	--	10.8	10.2
40	11.2	10.8	10.4	10.5	11.2	10.8
60	10.8	10.7	10.2	10.5	11.2	10.1
80	10.9	10.8	10.7	11.0	11.2	10.0
100	10.7	9.4	10.2	10.6	11.2	9.5

Phosphorus measurements have consisted of total and orthophosphate phosphorus. Concentrations of both forms are consistently below 5 μg/l, and significant differences or trends cannot be distinguished within the limits of the analytical technique. Nitrite, nitrate, and ammonia nitrogen are almost invariably below this laboratory's minimum detection limit of 10 μg/l and apparent differences are probably due to analytical limitations.

Chlorophyll _a_ determinations have been made on North Hole samples for each year of the study. The reliability of the data for 1971 and 1972 are uncertain, although the expected very low pigment levels were indicated. Chlorophyll _a_ was not detectable in the 1973 samples. Measurements for 1969, 1970, and 1974 are given in Table 7. Values are consistently below 1.0 μg/l, and exhibit no trends over the five year period of record.

Table 7
North Hole
Chlorophyll a
μg/l

Depth m	Sept. 1969	August 1970	August 1974
0	0.4	0.1	0.2
20	0.6	0.1	0.1
40	0.2	0.4	0.1
60	0.2	0.6	0.2
80	0.6	0.7	0.2
100	0.5	0.2	0.4

Primary productivity measurements by Larson and Donaldson (1970) showed an average carbon uptake rate in the summer of 1969 of 38 mg $C/m^2/day$. Powers et al (1972) showed carbon uptake rates in the summer of 1970 ranging between 0.03 and 0.10 mg $C/m^3/hr$. Both sets of data indicate extremely low productivity rates.

Laboratory algal assay tests were conducted on Waldo Lake water by Miller et al (1974). Autclaved-filtered water did not support growth beyond 0.06 mg dry wt/l, even with the addition of 1.0 mg N/l and 0.05 mg P/l. However, in the in situ primary productivity experiments carried out by Powers et al, addition of 0.05 mg P/l alone increased photosynthetic rate on three of four occasions. The influence of phosphorus plus nitrogen was not significantly different from the effect of phosphorus alone.

Summaries of algal cell counts and group identifications are presented in Table 8. Clump counts were made on a Sedgwick-Rafter cell, using concentrated samples prepared by settling 500 ml to 50 ml over a 12-day period. In the clump count method, all unicellular, colonial, and aggregated organisms are tallied as single units, and have equal numerical weight. Samples obtained in 1973 and 1974 have not been processed.

Table 8

North Hole

Phytoplankton

organisms/ml

Group	0	20	40	60	80	100
			1969 Depth, m			
Diatoms			10		10	10
Greens	10	30	10	30		10
Blue-greens						
Dinoflagellates	20	40	90	150	190	120
Unknown						
Total	30	70	110	180	200	140
			1970			
Diatoms				10		
Greens			10	10	10	
Blue-Greens						
Dinoflagellates	21	10	10			
Unknown						
Total	21	10	10	10	10	
			1971			
Diatoms	2	2				
Greens	7	4	20	184	177	100
Blue-Greens						
Dinoflagellates	94	19	2	2	10	1
Unknown						
Total	103	25	22	186	187	101
			1972			
Diatoms		2		2	3	
Greens	67	197	594	857	379	728
Blue-Greens						
Dinoflagellates	19		5			
Unknown						
Total	86	199	599	859	381	728

Although Larson and Donaldson (1970) reported an average of 4.5 organisms per #6 net tow near shore, Malueg et al reported no zooplankton. Repeated vertical tows from the deep stations and near shore horizontal tows, using a 0.5 m #10 plankton net, have failed to produce a single zooplankter during our entire study.

The extreme clarity of the lake is emphasized by the presence of the hepatic, Jungermannia triris Nees, and a moss Hygrohypnum (molle?), at the bottom of the North Hole at 127 m.

NUTRIENT BUDGETS

Sources of nutrients to Waldo Lake are precipitation (principally snow), intermittent surface runoff, and ground water. Septic tank drainage from campgrounds is a presumed source, although the 1973 study did not demonstrate transport of effluent to the lake. There are no permanent tributaries. The 30,000 visitor days estimated by the Forest Service for 1973, when prorated over an entire year, are equivalent to a permanent population of 82 persons (30,000/365 = 82). Assuming an average phosphorus loading rate of 1.1 kg P/capita/yr, this amounts to 93 kg P/yr, or 0.003 g P/m^2/yr to the lake.

Lacking measurements of surface and ground water contributions, it is not possible to measure directly the nutrient loadings to the lake. Phosphorus and nitrogen budgets have therefore been calculated by several different indirect methods. Constants used in the calculations include:

Average annual precipitation = 181.4 cm
Estimated annual evaporation = 109 cm
Catchment area of lake (including lake surface) = 7900 ha
Surface area of lake = 2700 ha
Average outflow from lake = 45 x 10^6 m^3/yr
Average total P concentration of outflow = 3.5 μg/l
Average total P concentration of lake = 3.5 μg/l
Average total P concentration of precipitation on catchment area = 5 μg/l (after Malueg et al)
Average N concentration of precipitation on catchment area = 83 μg/l (after Malueg et al).

PHOSPHORUS

1. Using information from Vollenweider (Input-Output Models), assume that P loading is three times the measured lake concentration and also (in this case) three times the measured phosphrous flowing out of the lake:

Measured P out = 157.5 kg/yr

P in = 3 x 157.5 = 472.5 kg/yr = <u>0.0175 g P/m^2/yr</u>.

2. Using unpublished data of Miller, assume from the innate characteristics of the watershed that P loading to Waldo Lake is the same as that from undisturbed forest land in the Upper Klamath Lake, Oregon, drainage (5.25 kg/km^2/yr):

Waldo Lake watershed = 5200 ha

5200 x 0.052 kg P/ha/yr = 270 kg P/yr = <u>0.01 g P/m^2/yr.</u>

3. Using average annual precipitation for the Waldo Lake watershed, and snow analyses of Malueg et al:

(a) Assume that all precipitation onto the watershed eventually enters the lake, and that the total P content of the precipitation is 5 mg P/m^3:

(143.4 x 10^6 m^3 water) (5 mg P/m^3) = 716.5 kg P/yr to lake = <u>0.027 g P/m^2/yr</u>.

(b) Assume that only that part of the precipitation equal to the measured outflow plus the estimated evaporation from the lake actually enters the lake:

measured outflow = 45×10^6 m^3/yr
est. evaporation = $\underline{29 \times 10^6 \ m^3/yr}$
runoff to lake = 74×10^6 m^3/yr, $(74 \times 10^6)(5 \text{ mg P})$
= 370 kg P/yr = $\underline{0.014 \text{ g P/m}^2\text{/yr}}$.

4. Using information from Vollenweider and Dillon (1974), Tables 5 and 6, assume a total P soil export factor of 0.010 g total P/m^2 of land/year. This is the value used by Patalas (1972) for Lake Superior (igneous forested land).

Area of Waldo Lake watershed = 5200 ha
$5200 \times 0.01 = 5.2 \times 10^5$ g/m^2 P from watershed soil/year.

Assume remainder of P loading is via direct precipitation onto lake surface:

(1.81 m^3 ppt'n/yr)(27×10^6 m^2 lake surface)
= 48.87×10^6 m^3 ppt'n onto lake surface x 5 mg P/m^3
= 2.4×10^5 g P.
5.2×10^5 g P from soil + 2.4×10^5 g P from ppt'n
= 7.6×10^5 g P to lake = $\underline{0.028 \text{ g P/m}^2\text{/yr.}}$

NITROGEN

Total nitrogen loading to the lake has been estimated by methods 2, 3a, and 3b (above). Method 1, in which measured output was related to input in the phosphorus estimates, has not been attempted for nitrogen because estimates of nitrogen retention in lakes are even more tenuous than for phosphorus. Method 4 could not be used because of lack of information on soil loading.

Method 2. Using unpublished data of Miller for the Upper Klamath Lake watershed, assume N loading = 22 kg/km^2/yr:

Waldo Lake watershed = 5200 ha

5200 x 0.22 kg N/ha/yr = 1144 kg N/yr = <u>0.042 g N/m^2/yr</u>.

Method 3a. Using average annual precipitation for the Waldo Lake watershed, and snow analyses of Malueg et al:

(a) Assume that all precipitation onto the catchment area eventually enters the lake, and that total N content is 83 mg N/m^3: (143.3 x 10^6) (83 mg N/m^3) = 11,894 kg N/yr to lake = <u>0.44 g N/m^2/yr</u>.

(b) Assume that runoff to lake = 74 x 10^6 m^3/yr (outflow plus evaporation):

(74 x 10^6)(83 mg N) = 6142 kg N/yr = <u>0.23 g N/m^2/yr</u>.

DISCUSSION

All available limnological criteria confirm the extremely pristine state of Waldo Lake. Comparisons with Crater Lake and Lake Tahoe, two other well-known ultraoligotrophic lakes, show that Waldo Lake's specific conductance is one to two orders of magnitude less and its total dissolved solids an order of magnitude less. Secchi disc values for Waldo fall within the range for Tahoe and Crater. Based on our 1969 and 1970 measurements, primary productivity in Waldo is significantly less than in the other two, as are phytoplankton numbers. As stated earlier, zooplankton have not been found at any time during our investigations of Waldo Lake.

Based on the calculated loading rates for phosphorus and nitrogen, Waldo Lake falls near the extreme lower end of the "Vollenweider scale." Using the highest rates yielded by the several estimates,

$$P = 0.028 \text{ g P/m}^2\text{/yr}$$
$$N = 0.44 \text{ g N/m}^2\text{/yr}$$
(N/P loading ratio = 15.7).

The ratio of mean depth to retention time is 1.68. Because of this low value, the lake, when entered on a plot of P loading vs mean depth/retention time, falls near the lower left portion of the diagram in the critical part of the oligotrophic region, implying that a relatively slight increase in phosphorus loading could strongly alter the trophic status. Such an implication appears to be substantiated by the primary productivity experiments of Powers et al where phosphorus was shown to stimulate photosynthetic activity. However, Miller et al were unable to increase algal production with an addition of phosphorus alone or phosphorus plus nitrogen, indicating that nutrients in addition to nitrogen and phosphorus were limiting to algal growth. This could well be the case in a lake where all dissolved constituents are in very low concentration. The relative importance of phosphorus in Waldo Lake is therefore uncertain, but there is no question that introduction of nutrient or polluting materials of any kind to such a unique resource should be held to a minimum. Increased concentrations of micronutrients could result in a condition where slight increases in phosphorus loading could significantly change the trophic state.

SUMMARY

Waldo Lake, in the Cascade Mountains of Oregon, is extremely oligotrophic, ranking amount the most pristine lakes of the world. The recent development of access roads and campground facilities has raised

questions concerning the possible response of the lake to the pressures of increased recreational use, and was the primary reason for the inception of our studies. The lake has no permanent tributaries, and the hydrologic and nutrient budgets are not amenable to accurate measurement. Several different methods of estimation place the rates of phosphrous and nitrogen loading at 0.028 and 0.44 g/m^2yr, respectively. The N/P loading ratio is 15.7. On the "Vollenweider scale" the lake is definitely oligotrophic, but lies in that area of the diagram where relatively small increases in P loading are significant. However, the relative importance of phosphorus in Waldo Lake is uncertain because of the very low concentrations of all measured nutrients.

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SECTION VI - WASHINGTON

LAKE WASHINGTON

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I. INTRODUCTION

A. History. In its natural state, Lake Washington drained from the south end through the Black River into the Duwamish estuary and Puget Sound. It had one major inlet, the Sammamish River from Lake Sammamish, and about a dozen small streams. In the 1890s a small cut was made between Union Bay of Lake Washington and Portage Bay of Lake Union to permit passage of logs to a sawmill. Later this cut was enlarged and a canal with locks made, between Lake Union and Puget Sound. It opened in 1916, at which time the level of the lake was lowered by about 3.3 m (10 feet) and the Cedar River was diverted into the south end of the lake. In the 1940s and 1950s small amounts of salt water entered Lake Washington and formed a transitory layer of very dilute sea water in the deepest parts. The latest intrusion was in 1952.

II. GEOGRAPHIC DESCRIPTION

A. Latitude 47° 38' N. Longitude 122° 14.5' W.

B. The level of the lake is regulated between 6.1 and 6.7 m above mean low water in Puget Sound except in unusually dry years. The lowest level, 5.6 m, occurred in 1958.

C. The catchment area of land including Mercer Island in the lake is 1588 km^2. The water area is 88 km^2, total 1676 km^2.

D. General climatic data (1931-1960). Monthly mean air temperatures vary from 5.1°C (41.2°F) in January to 18.67°C (65.6°F) in July.

Rainfall varies from a monthly mean of 1.6 cm (0.63 in.) in July to 13.77 cm (5.42 in.) in December. Yearly mean 86.61 cm (34.1 in.), range 48.58 cm to 114.07 cm.

In general, winds are from the southerly directions most frequently in winter, northerly in summer. The strongest winds come from southwest in spring or early summer. The mean velocity at Sand Point is 11.1 km/hr (6.9 mph).

Total evaporation is about the same as the rainfall with an average excess of rainfall of about 3 cm. Net monthly evaporation varies from -14.1 cm to +13.1 cm.

The lake never freezes across. In the most severe winters thin ice can develop in the bays, but this is a rare occurrence.

E. General geological characteristics. The lake occupies a deep, narrow through sculptured by the Vashon ice sheet. Most of the upland area is occupied with glacial till covered with a few feet of weathered soil. In the lowland valleys, alluvial deposits of clay are prevalent with sand and gravel deposits in places. Erosion appears not to be a major problem.

F. Vegetation.

The original vegetation was a thick forest dominated by Douglas fir (*Pseudotsuga menziesii*), red cedar (*Thuja plicata*) and western hemlock (*Tsuga heterophylla*). Spruce (*Picea sitchensis*) and fir (*Abies grandis*) were less common (Scott, 1962). Red alder (*Alnus rubra*) and cottonwood (*Populus trichocarpa*) were the only abundant deciduous trees. They grew on river floodplains and as pioneer trees on other disturbed sites.

The second- or third-growth forests currently around Seattle have a different distribution of species. In cut-over areas, red alder has become much more abundant, sometimes being the dominant species for a time. Alder dominance last 30-50 years, until conifers regenerate and overtop the alder. The success of alder varies, depending on soils and fire. Douglas fir grows with alder in many areas, cottonwood is sometimes common and willow is an important pioneer on other sites. Burning of slash and brush after clear-cutting, a frequent occurrence in the early years of this century, destroyed the conifer seed in the soil, encouraging the growth of alder, which has abundant and easily transported seeds. Large areas from which conifers had been removed completely were without a source for conifer seed and went over completely to alder forest and brush (modified from Davis, 1973).

The Cedar River watershed is under control of the Seattle Water Department. While it has been intensively managed, the characteristics of the soil are such that erosion into the Cedar River is not a serious problem.

G. Population. The lowland area is heavily urbanized, but most of the Cedar River watershed is fenced and uninhabited because it is the major source of water for the metropolitan area. The city of Seattle borders much of the west side of the lake and several small towns are at the ends and on the east side. The human population within the Lake Washington watershed is approximately 525,000.

H. Land use. In the Lake Washington watershed, land use includes residential, commercial and industrial, but large areas are undeveloped. Intensive lumbering takes place in the Cedar River watershed, which is otherwise largely undeveloped. A relatively small amount of agriculture is done, mostly in the Sammamish River area.

Shoreline use: residential, 64.5%; recreation, 19.0; undeveloped, 7.1; public service, 3.7; industrial, 2.8; commercial, 1.6; private club, 0.8; circulation and utilities, 0.5.

I. Use of water. To a large extent Lake Washington is used as a recreational amenity for boating, fishing and swimming. Commercial traffic on the lake consists mainly of rafted logs and of barges of sand and gravel in transit to construction companies. A commercial flying service at the north end of the lake and about thirty private planes use the lake for landing.

A Naval Air Station at Sand Point was partly deactivated in 1970, and recently a large part of the area has been released for two developments. One will be a public park, the other will be an establishment at which NOAA will station its ships and have some research and administrative activities.

The lake itself is no longer used as a general source of drinking water, but the Cedar River is a primary source of water for Seattle and a number of smaller towns in the area. It is also a spawning area for an important run of sockeye salmon, so there is public pressure to maintain an adquate flow.

An unusual feature of Lake Washington is the two multilane floating bridges that carry more than 85,000 vehicle crossings per day.

J. Sewage. The maximum input of treated secondary effluent took place in 1962, for in March 1963 a program of diversion was put into effect. The amount of sewage was progressively decreased from about 76,000 m^3 per day (20 million gallons), and the project was finished in 1968. Seattle has had combined sewer systems with storm overflows into Lake Washington. At the present, a project of sewer separation is being carried out. There has been no major source of industrial waste, although the Boeing Company put a waste rich in phosphate into the Cedar River in the 1950s. In the late 1950s some of the streams carried septic tank overflow, but this has been greatly reduced by local sewerage projects.

III. Description of Lake Washington.

A. Area 87.615 km^2
Length 21 km
Width: maximum 5.5 km, mean about 3

B. Volume 2885.3 million m^3

C. Maximum depth 62.5 m (approx)
Average depth 32.9 m

D. The deepest parts are shallow grooves or troughs between the middle and sides. About 83% of the lake is deeper than 10 m.

E. Typically the epilimnion is 10 m thick, and the E:H ratio then is 0.387.

F. The lake is monomictic. Secure stratification is usually established about the middle of May, although in calm years transitory stratification occurs in April and in windy years stratification may be delayed until June. Maximum temperature occurs in August. The lake begins to cool then and the epilimnion thickens progressively until homothermal conditions are established in November or December. During very cold weather, cold water masses form in the shallower bays and slide down to the bottom of the lake and out toward the middle, causing decreases in temperature that cannot be accounted for by mixing.

G. The deep sediments are a black planktogenic gyttja, dominated by diatom frustules. In shallow water most of the bottom is covered with boulders, gravel and sand.

H. At Seattle the maximum mean monthly rainfall is 13.8 cm (5.42 in) in December, the minimum 1.6 cm (0.63 in) in July. The extremes for individual months have been a trace in July and 38.9 cm (15.33 in) in January 1880. The maximum during the period of recovery from eutrophication was 25.6 cm (10.07 in) in December 1968. Annual total rainfall has varied from 143.4 cm in 1879 to 49.6 cm in 1952 (56.44 in to 19.52 in). Rainfall is greater in the upper part of the Cedar River watershed and heavy snows occur in some years.

I. Inflow and outflow of water. According to calculations by a hydrological model developed for METRO, the mean volume of water entering through all inlets in the period 1942-1972 was 1211 million m^3 per year. The minimum was 466.9 million m^3 in 1944 and the maximum 1681.8 million m^3 in 1950. The Cedar River is responsible for about half the total flow. Of the rest, the Sammamish River contributes about 72%, the other 28% being brought in by the various small streams around the lake. Thus, the two main rivers account for about 86% of the total inflow.

The mean rainfall of 0.8661 m amounts to about 75.9 million m^3 falling directly on the lake. The volume of the lake is 2885.3 million m^3.

Nothing quantitative is known about ground water.

J. Surface water currents have been studied by METRO using dye patches and streaks. No consistent current pattern exists. The drift from inlets to outlet is masked by wind-blown currents, but the movement of water is not clearly and directly related to momentary wind direction, evidently because of the constraints of the shores, and because of delayed effects of previous wind conditions.

K. Water renewal time. By dividing the volumes of inflow listed in Part I above into the volume of the lake, the following retention times for calendar years are obtained: mean 2.38 years, maximum 6.18 years (1944), minimum 1.72 years (1950). For 1957 when the first loading was calculated, it was 2.97 years by the model, 3.32 by calculations using U.S.G.S. gauge data.

The reciprocals of the numbers given above, the renewal rates per year are, in order: 0.420, 0.162, 0.583, and for 1957 (model) 0.336.

IV. Limnological characterization.

A. Physical

1. Temperature. Surface temperature in the open water varies from about 6° to about 25° C. The maximum temperature at the bottom in summer is about 8°.

2. Conductivity in recent years has varied between 76 and 87 µmhos.

3. Light. Some measurements of light penetration have been made with a photometer and many Secchi disc measurements of transparency taken. During the period of eutrophication at a time when the Secchi disc transparency was 1.1 m, 10% of the surface light intensity occurred at 2.4 m. During the period of de-eutrophication, Secchi values have increased, and the maximum value ever observed, 7.5 m, occurred in February 1975. The largest summer value ever observed, 5.5 m, occurred in July, 1975.

4. Lake Washington has no measureable humic color. When the lake is clearest in the winter, the Secchi disc appears green.

5. Solar radiation. During June-September, the mean daily solar radiation has varied between 391 and 468 langleys.

B. Chemical

1. The maximum pH occurs during the spring and summer when primary production is maximum. The highest value in 1933 was 8.6.

During the eutrophic years it got as high as 9.9 in 1963 and progressively decreased over the years, getting down to 8.5 in 1973, although it went back to 8.96 in 1974.

2. Dissolved oxygen. Although large volumes of the lake did not become anoxic during the period of eutrophication, values less than 5.0 were prevalent in late summer. Since diversion of sewage, oxygen concentrations in the hypolimnion remain in the order of 8 mgm/l.

3. Total phosphorus has varied greatly over the years with different degrees of eutrophication. The maximum annual mean was 65.7 μg/l in 1963, minimum 16.8 in 1973. Mean dissolved inorganic phosphate P in January-March was 56.9 μg/l in 1964, 8.8 in 1972.

4. Nitrogen has varied considerably, but not as much as phosphorus. The mean in January-March of inorganic N was 495 μg/l in 1965, 313 in 1973.

5. Alkalinity has not varied a great deal, being about 20-30 μg/l expressed as $CaCO_3$.

6. Few complete ion analyses are available for Lake Washington. A typical analysis, from 1969, is:

Ca	8.8	HCO_3	40.0	TDS	54
Mg	3.3	SO_4	8.2		
Na	4.6	Cl	3.1	(all as mg/l)	
K	1.1	SiO_2	8.6		

7. Few trace metal analyses has been published for the water of Lake Washington, although there is considerable interest in the sediments. The lake has been relatively enriched in a number of trace elements by emissions from a smelter near Tacoma 40 miles to the south.

C. Biological

1. Phytoplankton

a. Chlorophyll. The mean chlorophyll in summer was 41.0 μg/l in 1964, 4.8 in 1973.

b. Primary production (see Table 1).

c. Algal assays. In recent years, the natural population of phytoplankton has tended to respond to addition of phosphate more than to addition of nitrate in bottle tests. In the mid 1960s when the lake was still enriched with sewage, it tended not to be responsive to added phosphate.

d. Lake Washington has characteristically had a spring bloom of diatoms dominated by *Stephanodiscus*, *Fragilaria*, *Melosira* and *Asterionella*. In 1933 and 1950, the summer population was mostly a small mixture of species of green algae and some flagellates. During the period of eutrophication this basic pattern had superimposed on it a dense population of blue-green algae in the summer. The blue-greens included *Oscillatoria rubescens*, *O. agardhii*, *Microcystis*, *Anabaena* and *Aphanizomenon*.

2. Zooplankton. The most abundant zooplankton include *Diaptomus ashlandi*, *Epischura lacustris*, two species of *Cyclops*, *Diaphanosoma leuchtenbergianum* and *Bosmina longirostris*. Several species of rotifers become prominent, the most prevalent being *Keratella cochlearis* and *Kellicottia longispina*.

3. The bottom fauna is dominated by a variety of chironomids, with lesser numbers of tubificids and small molluscs (*Pisidium*).

4. Fish. A variety of species of fish live in the lake. Of special interest is the sockeye salmon (*Onchorhynchus nerka*) which became abundant in 1964 and is heavily fished.

5. Bacteria. Dr. James Staley is studying the bacteria with special attention to *Metallogenium* and *Caulobacter*.

6. Bottom flora and macrophytes. No systematic study appears to have been made. Genera growing in shallows include *Potamogeton*, *Myriophyllum*, *Najas*, *Anacharis*, and *Ceratophyllum*. Emergent plants include *Scirpus*, *Typha* and *Sagittaria*.

V. Nutrient Budget Summary.

The nutrient input to the lake varied greatly with the increase of sewage and then with the diversion. Data are summarized in Table 2.

Table 1. Primary production in Lake Washington

A. Gross oxygen production, $g/m^2/day$ in 24 hour runs

	July-Aug.	June-Sept.	Year
1958	---	4.2	2.1
1963	4.2	4.0	1.9
1964	3.0	3.8	2.0
1965	4.8	4.6	3.3
1966	5.2	4.7	3.7
1967	---	3.3	2.2
1968	4.6	3.7	2.3
1969	2.7	2.7	2.0
1970	3.3	3.0	1.9
1971	0.9	1.1	0.9
1972	1.2	1.4	1.2
1973	1.8	2.0	1.3
1974	1.4	1.4	1.3
1975	1.8	1.7	1.4
1976	1.2	1.0	0.8

B. ^{14}C fixation in 24 hour runs

	Annual		June-Sept.
	Mean $mg/m^2/day$	Total $gm/m^2/year$	$mg/m^2/day$
1972	---	---	246
1973	198	72	265
1974	282	103	353
1975	371	135	493
1976	187	68	219

Note: Measurements of carbon uptake rates were started in 1963, but were not done often enough to permit calculation of means in the earlier years.

Table 2. Nutrient income to Lake Washington, kg/year

A. Income

1957

Phosphorus

	Total P	Dissolved P
Streams	42,600	36,000
Sewage plant effluent	42,100	37,900
Industrial waste (est.)	7,800	7,800
Septic tank drainage (est.)	8,600	7,800
Combined sewer overflow (est.)	7,100	6,500
Total wastes	65,600	60,000
Total (full)	108,200	96,000
Total - septic tanks	99,600	88,200
- combined sewers	92,500	81,700

Nitrogen

	Total N	Dissolved N	Nitrate-N
Streams	1,471,000	1,331,000	253,100
Sewage plant effluent	172,600	133,300	19,600
Industrial waste (est.)	---	10,600	---
Septic tank drainage (est.)	---	16,100	---
Combined sewer overflow (est.)	---	18,600	---
Total wastes	---	178,600	---
Total (full)	1,688,900	1,509,600	---
Total - septic tanks	1,672,800	1,493,500	---
- combined sewers	1,654,200	1,474,900	272,700

Data from Hollis M. Phillips, Seattle Department of Engineering. Stream values based on measurements of concentration and flow. Flow data for two rivers and two small streams from U.S.G.S. Other flow data estimated from drainage area. Septic tank drainage, combined sewer overflow and industrial waste estimates from Brown and Caldwell. The full total of all items listed is probably an overestimate since some of the septic tank drainage would have entered the streams and appeared in the measurements there. The underlined totals are probably the best to use.

1962

Phosphorus

	Total P
Streams	80,900
Sewage plant effluent	128,300
Combined sewer overflow	21,600
Total	230,800

This was the year of maximum sewage input but no measurements were made. Diversion started in March 1963. To obtain the figures listed, sewage plant effluent was calculated by proportion with the populations served by the treatment plants in 1962 and 1964 (see below). Combined sewer overflow was estimated by proportion with the estimate of 1957 and the sewage plant effluent. Septic tank drainage was ignored since many of

Table 2. (Continued)

the formerly unsewered areas were now sewered. A sewage treatment plant served the town of Bothell on the Sammamish River from 1959 to March 1967, and the effluent went into the river where the nutrients would appear in the stream analyses. The population served by the plant was 2,460 in 1962 and 2,600 in 1964.

1964

Phosphorus

	Total P
Streams	80,900
Sewage plant effluent	103,900
Combined sewer overflow	17,500
Total	202,308

Nitrogen

	Total Organic N	Inorganic N	Sum
Streams	380,500	527,000	907,500
Sewage plant effluent	271,000	33,000	304,000
Combined sewer overflow	45,700	5,600	51,300
Total	697,200	565,600	1,262,800

Data from Municipality of Metropolitan Seattle. By 1964, three of the sewage treatment plants had been diverted. In 1957 they had contributed 34.2% of the dissolved P and 33.2% of the total P. However, the population served by each plant had increased. Combined sewer overflow was calculated as for 1962. Inorganic N means nitrate and nitrite.

1970-1974

Streams only

	Total P	Dissolved P*	Total Organic N	Nitrate-N
1970	43,700	---	---	442,600
1971	37,600	---	401,600	559,800
1972	91,200	21,240	647,800	719,900
1973	26,800	14,000	807,200	398,400
1974	41,300	16,700	386,900	453,300
1975	66,300	7,300	607,900	640,500

*Perchloric acid digestion of filtered sample

Sewage diversion started in 1963 and was finished early in 1968, although most had been diverted by 1967. Floods in the Cedar River in early 1972 and in winter 1975-1976 brought in much silt, accounting for the elevated phosphorus input in those years. The 1972 flood was accompanied by more erosion and landslides than the later one.

Table 2. (Continued)

B. Loading

The values in Part A were used to calculate the annual loading figures. The area of the lake is 87,615 thousand m^2, the volume is 2,853.0 million m^3, and the mean depth 32.9 m.

	Inflow Thousands m^3	Total P $g/m^2 \cdot year$	Dissolved P $g/m^2 \cdot year$	Inflow / Lake Volume	Hydraulic loading
1957	973,600	1.2	1.1	0.338	11.1
1962	964,400	2.6	---	0.334	11.0
1964	1,554,061	2.3	---	0.539	17.7
1970	1,207,800	0.5	---	0.419	13.8
1971	1,539,706	0.43	---	0.534	17.6
1972	1,513,606	1.0	0.24	0.525	17.3
1973	898,300	0.31	0.16	0.311	10.2
1974	1,329,300	0.47	0.19	0.461	15.2
1975	1,479,740	0.76	0.08	0.513	16.9

All the values in this table involve a certain amount of estimation and extrapolation since measurements of flow and concentration were not made in all the small inlet streams each year. There is more than one way to approximate some of the values, as by proportion with watershed area, by regression of one stream that has been gauged only part of the time on one that has a complete record, or by hydrological calculation. The most elaborate study of the small streams was made in 1957 by the Seattle Engineering Department (Hollis M. Phillips, personal communication; see Edmondson 1972). In 1957, 10 small permanent streams contributed 8.8% of the water, 13.5% of the total phosphorus, 24.7% of the phosphate, 30.3% of the nitrate, and 15.2% of the Kjeldahl nitrogen. The chemical content of the small streams is more like that of the Sammamish River than of the Cedar, and the following proportions of the Sammamish input were used for calculating stream input for later years when all the streams were not measured: water 24.5%, total phosphorus 13.5%, phosphate 50.6%, nitrate 43.8%, Kjeldahl nitrogen 38.6%. In 1957 the volume of sewage effluent was 8,608 thousand m^3, less than 1% of the streamflow. The maximum rainfall in the years listed was 104.5 cm in 1972, amounting to 91,557 thousand m^3 on the lake, about 3% of the volume of the lake. These volumes have been ignored in calculating inflow which is limited to stream flow.

The loading calculations for 1970-1974 do not include flow from Seattle's storm sewers nor overflow from the remaining combined sewers that occurs during rainy periods. In 1976, this amounted to about 7,000 kg/year of total phosphorus, or 0.08 g/m^2, about equally divided between the two sources (personal communication, Glen Farris and John Buffo of METRO). The calculations also do not include overland drainage or inflow from temporary streams.

Some of the differences between this table and previously published values are accounted for by improvements in the information available and in the calculations. Some values in this table for years after 1971 may be revised in future calculations as more information becomes available, but any changes are expected to be small. Phosphorus and water loading for any year are unlikely to be increased by as much as 20% over the values presented here.

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NUTRIENT LOADING AND TROPHIC STATE

OF LAKE SAMMAMISH, WASHINGTON

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INTRODUCTION

Lake Sammamish is best characterized as mesotrophic and sediment core analyses indicate that its status has remained relatively constant for more than the past 100 years. The lake has been studied continuously since late 1969 with only two previously recorded studies; a nearly two-year study by the Municipality of Metro Seattle in 1964-65 and a one day survey in 1913 (Kemmerer et al., 1924). In addition to continuous monitoring of limnological characteristics since late 1969 to the present, special studies of secondary production (zooplankton and fish), nutrient exchange rates between sediment and water, phytoplankton uptake of nutrients, feeding rates of zooplankton, profundal bottom fauna, and dynamic modeling of the phosphorus cycle have continued, as well as a careful evaluation of the nutrient (particularly P) income.

Most of this effort has been for the purpose of defining the processes that have permitted the lake to remain mesotrophic in spite of alteration of the P loading. Because the lake was thought to be showing early signs of eutrophication (Isaac et al., 1969), the Municipality of Metropolitan Seattle (Metro) diverted the secondary effluent from the town of Issaquah and waste from a dairy processing plant in the fall of 1968. This diversion was subsequently shown to have amounted to one-third of the lake's P loading. The lake's internal sediment-water interchange mechanism controlled by iron can resist P loading changes over a range of at least 0.66-1.0 g P/m^2 year. This allows the available water column P content to remain remarkably stable and is probably the main cause for the lake's lack of response to diversion. However, stability could not be expected to persist over a much greater range in loading and when viewed over the range of trophic state and loading that exists in the world's lakes, the range examined in Sammamish appears rather small.

GEOGRAPHIC DESCRIPTION OF LAKE SAMMAMISH

The waning of the Wisconsin glaciation (14,000 BP) left the Puget Sound lowlands dominated by striated hills, rolling uplands, and deeply cut troughs. Today one trough is occupied by Lake Sammamish, a second by Lake Washington with the meandering Sammamish River connecting the two. A mild, maritime climate now prevails, annually producing 90 centimeters of precipitation and a mean monthly temperature of 11.5°C (52.7°F). Direct sunshine is present 45 percent of the daylight hours. Table I provides a summary of the pertinent geographic conditions.

Parameter	Lake Sammamish
Location	
Altitude (meters above mean sea level)	12
Longitude	122°05'W
Latitude	46°36'N
Size of Drainage Basin (km^2)	253
Duration of ice cover	none
Evapotranspiration (cm)	23.7
Evaporation (cm)	5.1
Precipitation (cm)	90
Maximum monthly precipitation (cm	39
Minimum monthly precipitation (cm)	0

Table I. Summary of Basin Geography

The predominant surface stratum of the drainage basin is a light-gray till. This till is a hard unsorted mixture about 46 meters thick, consisting of clay, sand silt, and gravel. Although the till is relatively impermeable, thin beds of sand and gravel commonly yield small quantities of perched water.

Aquifers transect the basin, with several artesian wells surfacing within the basin (Liesch, *et al.*, 1963). Coal seams are located in the southern half of the watershed, while high quality sand and gravel, refractory grade clay, quarry basalt and cinnebar deposits are scattered throughout the basin (Livingston, 1971).

A geologic cross-section cutting through Issaquah in an east-west direction shows base rock consisting of marine sedimentary rocks on the west side of the Lake Sammamish valley. On the east side is volcanic rock with overlying layers of clay, advanced stratified drift, till and sedimentary deposits (Liesch, *et al.*, 1963).

Prior to the arrival of European settlers in 1862, the Lake Sammamish basin was covered in a climax formation of Western Red Cedar (*Thuja plicata*), Western Hemlock (*Tsuga heterophylla*), and Douglas Fir (*Pseudotsuga taxifolia*) (Hansen,1938). Heavy logging around the turn of the century left the basin in second growth forest. Today 80% of the watershed remains in second growth, primarily red alder (*Alnus oregona*) with scattered maple (*Acer*, *sp.*) and willow (*Salix sp.*). Hence the impact of land erosion upon the lake is minimized.

The population of the basin has grown from three families in 1862 to the present 40,000, the majority of the growth coming in the last 10 years. The only sizeable concentration is located in the town of Issaquah, population 4,500. The town is comprised of the small businesses required to support a residential community. The only industrial development is a dairy processing plant and a state salmon hatchery. Within the watershed are several gravel operations and a county sanitary landfill. Large residential developments have been built throughout the entire west side of the lake. The east side is dotted with small farms, but the mjaor portion of the land remains in second-growth. A narrow strip of land along the east shore of the lake has been subdivided into residential tracts. The upper valley drained by Issaquah and Tibbetts Creeks is primarily forested with scattered farms and small clusters of houses (Fig. 1).

The primary point sources of wastewater within the basin were the town of Issaquah, the milk processing plant, and the fish hatchery. Since 1968, the effluent from the town's trickling filter plant (568 m^3/d) and the milk plant (284 m^3/d) have been diverted out of the basin. Today only the milk plant cooling water (227 m^3/d from groundwater) and the hatchery passthrough water, which originally comes from Issaquah Creek, are discharged through Issaquah Creek, to Lake Sammamish. Only the sparsely settled east side and upper valley sections of the watershed remain on septic tanks.

MORPHOMETRIC AND HYDROLOGIC DESCRIPTION OF LAKE SAMMAMISH

Lake Sammamish occupied a 13 km section of the Sammamish River Valley after the Wisconsin glaciation when the retreating Vashon glacier left a terminal maraine blocking the valley. Today the lake level is controlled by a weir at the head of the Sammamish River. Morphometric and hydrologic data on the lake are summarized in Table II. Sixty five percent of the lake surface has a depth greater than 15 m. The ratio of epilimnion to hypolimnion volume is 1.0.

The preliminary mapping of surface lake sediments completed to date include particle size distribution and mineralogical composition, cation exchange capacity and chemical analysis shown below (Horton, 1972):

Properties	Mean Value	Properties	Mean Value
CEC, meq/100g	23.9	Chemical Analysis, mg/g dry weight	
Size distribution, %		C	5.1
Sand	13	N	4.8
Silt	60	P	1.3
Clay	27	Fe	52.0
		Mn	1.1
		Ca	8.1
		Mg	15.7
		Na	21.2
		K	2.9

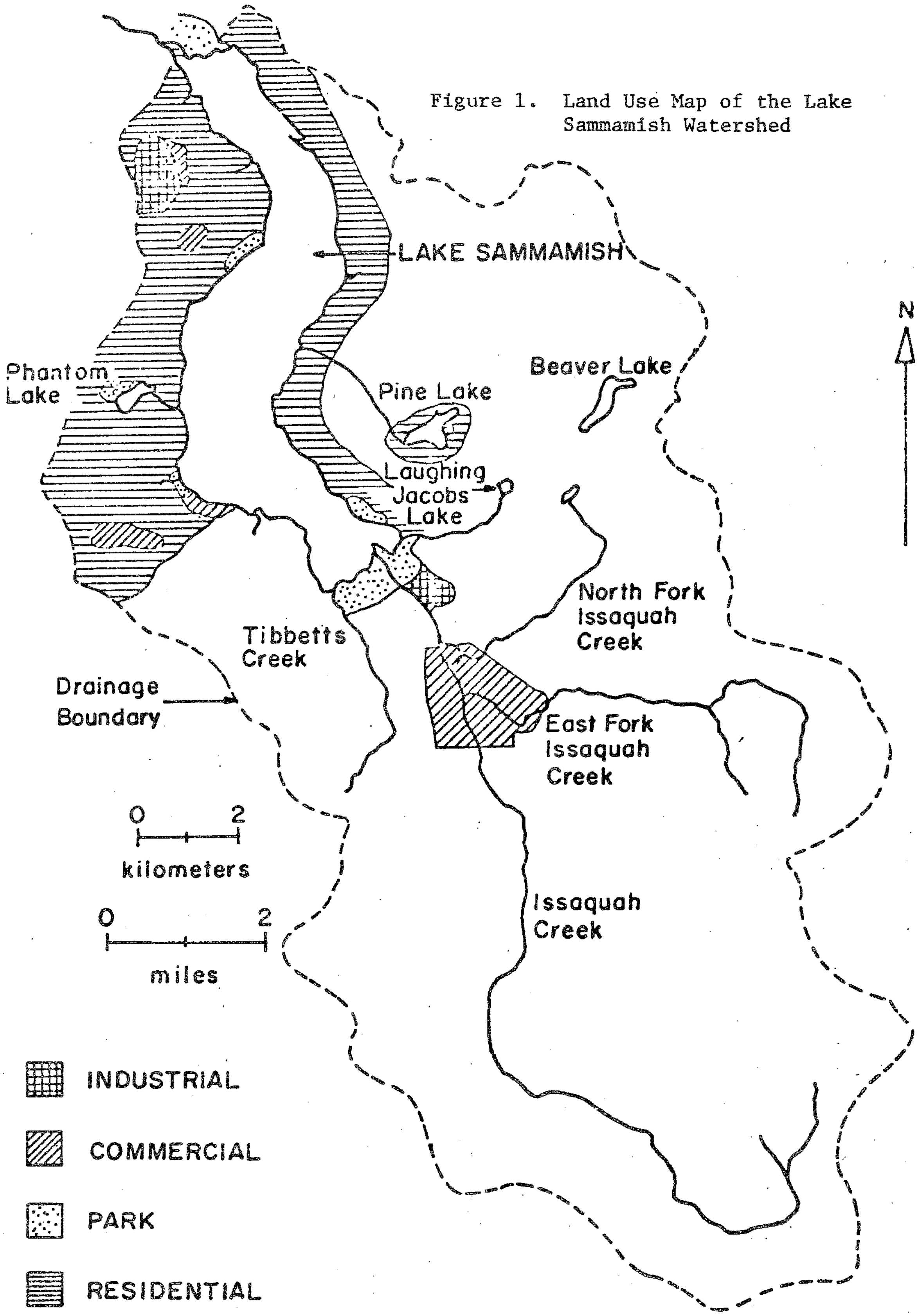

Figure 1. Land Use Map of the Lake Sammamish Watershed

Parameter	Lake Sammamish
Drainage Area (km^2)	253
Surface Area of lake (km^2)	19.8
Lake Volume (km^3)*	0.35
Depth	
Mean (m)	17.7
Maximum (m)	32.0
Epilimnion (m)	8.8
Euphotic (m)	7.3
Width	
Mean (km)	1.5
Maximum (km)	2.4
Length of Lake (km)	13.0
Length of Shoreline (km)	34.0
Water Retention Time (yrs)	2.2
Stream Inflow (km^3/yr)	0.167
Stream Outflow (km^3/yr)	0.162
Groundwater Infiltration (km^3/yr)	0.0
Groundwater Exfiltration (km^3/yr)	0.01
Duration of Stratification (mos)	7

*influenced by wier

Table II. Summary of Pertinent Hydrologic and Morphometric Characteristics for Lake Sammamish

The study of water currents has been limited to the movement of Issaquah Creek water in the lake (Moon, 1973). During the period of winter mixing the creek water dispersal was primarily influenced by wind direction and velocity. The water was sufficiently dispersed at a distance of 500 m to make the tracer undetectable. Similar studies made during thermal stratification showed the creek water plunging into the metalimnion (9-12 m) and dispersing in a fan-like pattern.

LIMNOLOGICAL CHARACTERIZATION

The limnological monitoring of Lake Sammamish has been continuous since 1970. Prior to 1970, Metro monitored the lake for a 1.5 year period in 1964 and 1965. Monitoring has been conducted largely at one centrally located station which has been shown to represent the limnetic area, and at a frequency of usually twice per month.

Physical

Temperature

The lake is monomictic and begins thermal stratification in May. Maximum water column stability occurs by late August and destruction of the thermocline is complete by late November. The surface temperature range is from a minimum 5.5°C to a maximum 25.5°C. The bottom water remains below 7°C.

Light Penetration

The depth of visibility has been determined by means of the Secchi disc. The annual mean for the six years of data is 3.3 m. The lowest seasonal readings (3.0 m) occur in the winter due to turbidity from the winter mixing and runoff. The springtime mean is only slightly higher (3.1 m), but the low values are due to the diatom pulse. The light penetration increases during the summer (3.5 m) and reaches its highest mean value in autumn (3.6 m).

Light extinction was determined by a submarine photometer. The bottom of the euphotic zone was considered to be at a depth receiving 1% of the surface light intensity. The mean depth of the euphotic zone is 7.3 m, while the range is from 5.0 to 12.5 m.

Solar Radiation

Insolation was determined at the University of Washington campus with an integrating Epply pyranometer. The ten year mean insolation is 3000 $kcal/m^2/day$ and the range is 700 $kcal/m^2/day$ in December to 5700 $kcal/m^2/day$ in August.

Chemical

pH and Alkalinity

The pH ranges from 6.3 to 9.6 due largely to biological activity. Correspondingly the alkalinity as $CaCO_3$ ranges from 26 mg/l (0.52 meq/l) to 42 mg/l (0.84 meq/l), while the mean is 33.3 mg/l (0.67 meq/l).

Dissolved Oxygen

During the winter, the oxygen content essentially remains at an air saturation level, approximately 12 mg O_2/l, due to continual circulation. The development of thermal stratification in early May results in a clinograde O_2 curve that approaches zero oxygen content (0.1 mg O_2/l) in the bottom waters by mid-August. The hypolimnetic oxygen deficit continues to increase until early October. By this time the entire hypolimnion (below 15 meters) has less than 1 mg O_2/l. Oxygen levels start to increase with the coming of the autumnal circulation.

Phosphorus

Total and ortho-phosphorus have been measured from 1969 to the present. The five-year mean concentrations for the 7.3 m photic zone are as follows:

	yearly	winter (Dec.-Feb.)	growing season (March-Aug.)
Total-P (μg/l)	26	30	25
Soluble ortho-P (μg/l)	6	13	4

The winter total P content in 1975 remained identical to the previous five-year mean - 30 μg/l. The winter mean total phosphorus concentration for the entire water column was 36 μg/l. This reflects the higher total phosphorus concentrations found in the hypolimnion. The mean total P content in the water column prior to fall turnover is greater - 40 μg/l. The range in total and ortho P over the five years has been 10-90 and 1-21 μg/l, respectively.

Nitrogen

Although all forms of N have been determined for only the past year, inorganic nitrogen (NO_2+NO_3-N) data are available for the past five years. The five year annual mean for the photic zone is 180 μg N/l and 275 μg/l for the growing season (March-Aug.).

Annual surface water values have been computed for calendar year 1973 and are summarized below:

	Mean Annual Concentration (μg N/l)	Range (μg N/l)
Organic N	225	60 - 403
NH_3-N	41	5 - 125

The 1973 inorganic nitrogen (NO_2+NO_3-N) mean compares favorably with the four year photic zone mean (191 vs 180 μg N/l), indicating that the single year's data may be representative of the long-term nitrogen concentrations.

Metals

Only preliminary data are available for metals. Neither $SO_4^{=}$ nor Cl^- have been measured, while the only trace metals measured have been Mn^{++}, Zn^{++} and Pb^{++}. The results from a single central station survey during the stratified period are shown as follows:

location	Ca	Mg	Na	K	Fe	Mn	Zn	Pb
	---------- mg/l ----------				------------ µg/l ------------			
Lake Surface								
Surface	12.80	3.42	8.43	1.01	40	19	376	0.5
8 m			8.47	0.98	63	40	300	0.5
16 m	8.40	3.44	8.15	0.94	280	600	35	0.8
25 m	8.95	3.68	8.17	1.00	1020	1660	34	0.8
Inflows								
Issaquah Ck.	12.40	3.70	9.31	0.94	450	35	318	0.9
Tibbetts Ck.	24.70	8.15	14.59	1.52	110	20	150	4.2
Outflow								
Sammamish River	6.05	3.00	8.16	1.13	25	9	7	0.6

Biological

Phytoplankton

Phytoplankton has a peak in the spring composed primarily of diatoms. The dominating genera during winter and spring are *Melosira* and *Stephanodiscus*. During the summer and fall, *Fragilaria*, *Synedra*, *Melosira*, *Rhizosalenia* and *Asterionella* are the major diatoms. The bluegreen algae are comprised predominantly of *Aphanocapsa*, *Microcystis*, *Coelosphaerium*, *Anabaena* and *Aphanizomenon*. In 1973-74 the appearance of *Aphanizomenon* has been less pronounced than earlier while the abundance of an Oscillatoriaceae species has increased. Predominant chlorophyseans are *Oocystis*, *Sphaerocystis*, *Closteriopsis*, *Chlamydomonas* and *Staurastrum*. Also predominant in the phytoplankton of the lake is the chrysomonad *Mallomonas*.

Mean values for phytoplankton chlorophyll *a* and primary productivity are given in Table III. Peak values during the years 1970-74 were 25.1, 28.3, 7.7, 12.2 and 13.9 mg/m^3 for chlorophyll *a* and 1257, 1061, 1730, 1581 and 2389 mg C/m^2·day for primary productivity, respectively. The bluegreen algae have decreased in importance over the 1970-74 period compared to the pre-sewage-diversion period in 1964-65. The average decrease has been nearly 40%.

Year	Chlorophyll a (mg/m^3)		Primary Productivity (mg C/m^2·day)	
	yearly	growing season	yearly	growing season
1970	5.7	7.7	711	899
1971	6.6	10.9	467	575
1972	4.3	4.8	799	952
1973	4.0	4.7	496	545
1974	6.0	6.8	789	904
Average	5.3	7.0	652	775

Table III. Annual and growing season means of phytoplankton chlorophyll a (weighted means for the euphotic zone) and daily rate of primary productivity in 1970-1974.

Zooplankton

Vertical net hauls was the procedure used to collect zooplankton at frequencies varying from twice weekly to once per month with the least frequency at periods of low reproductive activity. The zooplankton fauna were dominated in 1972-73 by copepods, among which *Diaptomus ashlandi* was the most abundant species. The following species of zooplankton have been found (*indicates common species):

Copepods
- **Diaptomus ashlandi*
- **Epischura nevadensis*
- **Cyclops bicuspidatus*

Cladocerans
- **Daphnia thorata*
- **D. schødleri*
- **Bosmina longirostris*
- **Diaphanasoma leuchtenbergianum*
- *Leptodora kindtii*
- *Scapholeberis kingi*

Rotifers
- **Kellicottia longispina*
- *K. bostoniensis*
- **Polyarthra sp.*
- *Keratella cochlearis*
- *K. quadrata*
- **Conochilus unicornis*
- *Collotheca mutabilis*
- *C. pelagica*
- *Notholca squamula*
- *Ploesoma hudsoni*
- *Gastropus sp.*
- *Synchaeta sp.*
- *Trichocerca sp.*
- *Filinia sp.*

The mean biomass and production rate of zooplankton in the lake during the two-year period 1972-73 was:

	annual mean	growing season
Biomass (mg/m^3 dry wt.)	44.3	46.1
Production rate ($mg/m^3/day$ dry wt.)	.98	1.26

The growing season secondary productivity was only 4% of the primary productivity. Discussion of this low efficiency is given by Pederson, et al. (in press).

Bottom Fauna

A survey of the bottom fauna was made in July 1974. The macro fauna was dominated by chironomids in sublittoral (5 m) and deep profundal areas (25 m). Oligochaetes were dominating in the upper profundal (15 m) (Table IV). *Cladotanytarsus* and *Tanytarsus* were the dominating chironomid genera at 5 m depth. *Chironomus* larvae of the salinarius type (probably identical with *Ch. atritibia*), followed by *Phaenopsectra* were the most abundant forms at 15 m. The high density of *Chironomus* at 25 m depth was almost exclusively larvae of the *Ch. salinarius* type. The growth and development of this species is closely correlated with phytoplankton production and the duration of anoxic conditions in the deep profundal (Bissonnette, 1974).

	5 m (7 Ekmans)	15 m (10 Ekmans)	25 m (8 Ekmans)
Chironomidae	8330	3040	22640
Oligochaeta	1870	7010	4960
Mollusca	310	270	40
Crustacea	10	60	-
Others	50	-	-

Table IV. Abundance of major groups in the bottom fauna in July 1974 (ind/m^2; 0.4 mm mesh size)

Bottom Flora and Macrophytes

Meager data exist on this topic. The littoral zone is not extensive in the lake but moderate sized areas of submergent macrophytes do occur at either end of the lake. Periphyton growth occurs at some points adjacent to stream or storm water inflows, which is a topic presently under study.

NUTRIENT LOADING

Phosphorus

Earlier estimates of the nutrient loading to Lake Sammamish were made difficult by the quick response to rainfall in the tributaries, particularly Issaquah Creek, which contributes 70% of the surface water and 72.5% of the total phosphorus to the lake (Emery, *et al.*, 1973). For example, as much as 5% of the total annual phosphorus load has been calculated to enter the lake in one day due to a combination of high flow and high nutrient concentrations. The installment of an automatic sampler in the main tributary in 1973 has permitted accurate estimates of phosphorus loading for the last two years. Through comparison of similar hydrological years and the results of earlier monthly samples, the decrease in loading through sewage diversion was estimated. Limited rainwater analyses during water year 1971 established an atmospheric input to the lake surface. Groundwater input was considered insignificant relative to the other sources because the water balance was roughly explainable from a consideration of surface inputs and outputs. The loading rate of phosphorus from three sources is shown below:

Source	Percentage		kg P/yr	
	before div.	after div.	before div.	after div.
Waste Discharges	37	3	7,500	500
Land Runoff	58	89	11,500	11,500
Precipitation	5	8	1,000	1,000
Groundwater	0	0	0	0
Total	100		20,000	13,000

Nitrogen

The data for a nitrogen loading are not as extensive as for phosphorus. The contribution from ground water has been assumed to be zero, while the contribution from precipitation has not been evaluated. On the basis of the following estimates before and after sewage diversion in 1968 there appears to be no significant change in nitrogen loading (Guttormsen, 1974):

	Organic N+NH_3-N kg/yr	NO_2+NO_3-N kg/yr	Total N kg/yr
1965 Water Year	69,000	174,000	243,000
March 1972-Feb. 1973	60,000	198,000	258,000

DISCUSSION

Limnological Characteristics

The outstanding characteristic in Lake Sammamish is its consistently high oxygen deficit rate of about 0.05 mg/cm^2·day and complete hypolimnetic anaerobiosis from August through October. This is caused more from the lake's morphometry than high productivity since the epilimnion-to-hypolimnion volume ratio is rather high at 1.0 and the growing season mean productivity is only about 700 mg C/m^2·day, which is more typical of mesotrophy. Iron and phosphorus content are inversely related to oxygen content in the hypolimnion and, thus, the process of phosphorus release and complexation and resultant availability is controlled by the lake's anaerobic character. Because the lake is monomictic a winter stagnation period does not exist.

The lake usually has one large phytoplankton maximum - a diatom outburst in April. In springs with lower light and slower onset of water column stability the maximum is less, is delayed until June and is mixed with green and bluegreen algae. However, mean growing season chl a content and productivity show much less variance from year-to-year ranging from 4.7-10.9 μg/l and 545-952 mg C/m^2·day, respectively. Secchi disk depth is similar from year-to-year, with a growing season mean slightly in excess of 3.3 m and the maximum exceeding 5 m at times.

The minimum nutrient content and chl a occur in August, but usually show a slight increase in September and October as the metalimnion is forced downward proceeding toward the November turnover. Total P is then maximum at overturn reaching 40 μg/l (in excess of 100 before sewage diversion), with the December through February mean remaining very constant at about 30 μg/l. Nitrate-N at this time is typically around 275 μg/l.

Trophic State

The present trophic state of the lake was determined by a comparison of the above mentioned limnological characteristics with criteria for eutrophication (National Academy of Sciences, 1972). It appears that Lake Sammamish can be considered as mesotrophic with respect to phytoplankton biomass (expressed as chlorophyll a), daily and annual primary productivity and the composition of the benthic communities. Oxygen deficit rate and nutrient concentrations are more indicative of mesotrophy-eutrophy. The loading of total phosphorus (0.66 g/m^2·yr) and total nitrogen (13 g/m^2·yr) are both considerably above the eutrophic danger limit of Vollenweider's (1968) guidelines. With his recent correction of mean depth for flushing time the loading rates are nearer the danger limit, however.

Paleolimnological studies of diatom profiles, phosphorus and organic content, and distribution of chironomid head capsules show no change in the trophic state of the lake during the last 120 years.

Trophic State vs. Nutrient Loading

The fact that both producers and consumers in the lake do not seem to respond to the eutrophic level of nutrient loading suggests that some internal factor(s) is controlling the availability of the incoming phosphorus to the phytoplankton (Welch, *et al.*, 1973). The evidence to support the hypothesis centers around the lake phosphorus content being controlled by iron. Horton (1972) has shown that total iron is closely correlated with total phosphorus as oxygen is exhausted in the hypolimnion during late summer. Although phosphorus increases in the surface waters following lake turnover in late November, P is rapidly complexed by what are probably ferric hydroxides. Much of the released phosphorus is thereby largely resedimented and rendered unavailable to the phytoplankton when light is adequate in April and May.

The pattern of response in the lake since diversion is shown in Fig. 2. Although considerable year-to-year variation has occurred in chl *a* content, the photic zone total P content has remained rather constant. The year-to-year variation in chl *a* observed was no doubt largely a response to light and extent of early spring stratification, but the constant P content is indicative of the lake's resistance to P loading change in the range of at least 0.66 to 1.0 g $P/m^2 \cdot yr$. However, when viewed over a wider range of loading known for lakes of varied trophic state its general significance is questionable. Fig. 3 shows a very strong correlation between volumetric P loading and "potential" chl *a* (chl *a* ÷ residence time - yr). Here one can see the controlling significance of P loading with respect to chl *a* accumulation (in so far as water residence time allows P utilization) over a wide range of loading. Also, the relatively small aberration in chl *a* that could be caused by the observed P loading change in Lake Sammamish is clear. With further loading change Sammamish might well be expected to conform to the linear relationship in Fig. 3.

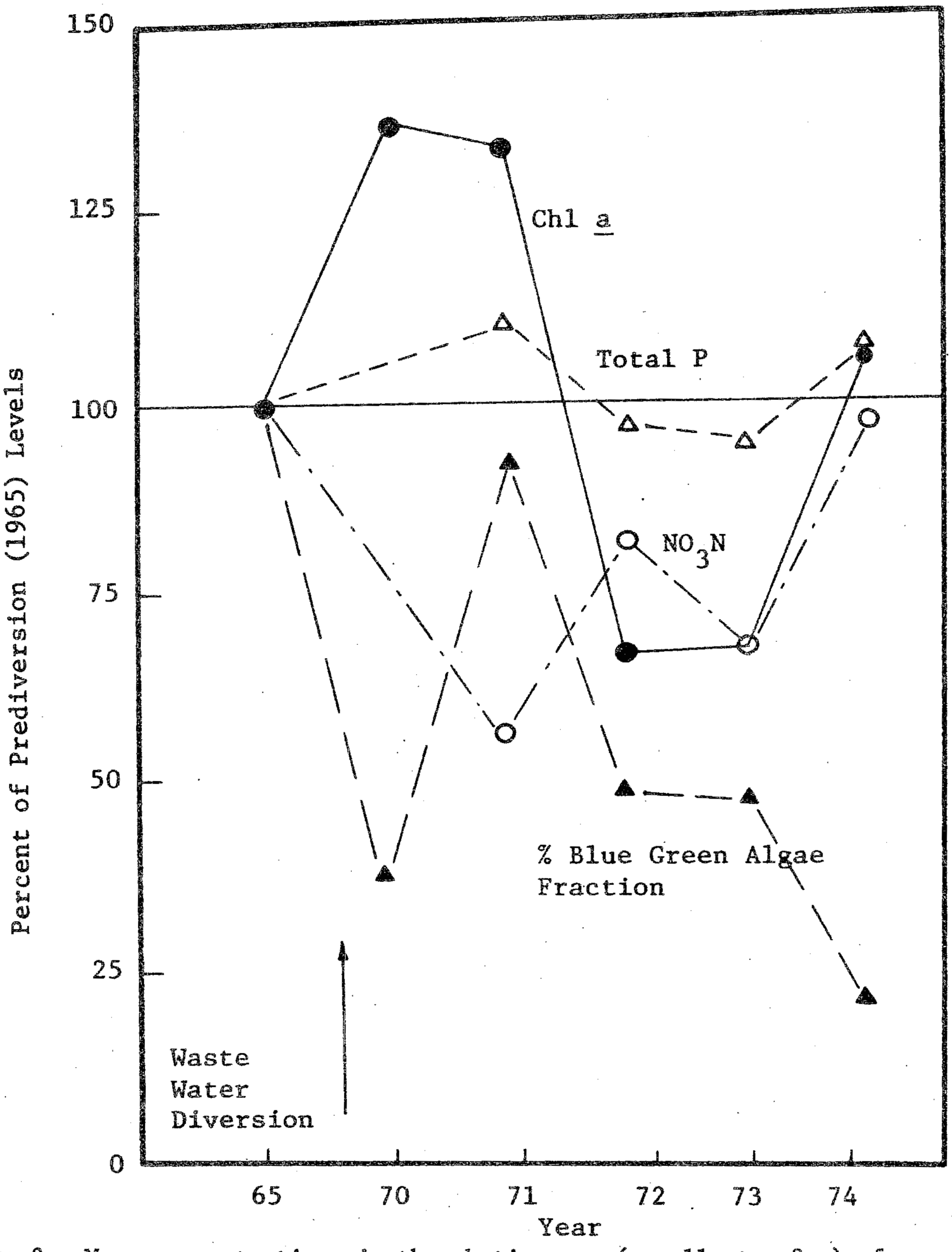

Figure 2. Mean concentrations in the photic zone (usually top 8 m) of growing season chl a (Mar-Aug), summer blue green algal fraction (June-Oct) and winter (Dec-Feb) total phosphorus and nitrate nitrogen relative to pre-diversion 1965 levels. The 1965 levels were: chl a 6.5 µg/1 (actually a mean of 1964 and 1965 data), total P 31 µg/1 and NO_3-N 390 µg/1.

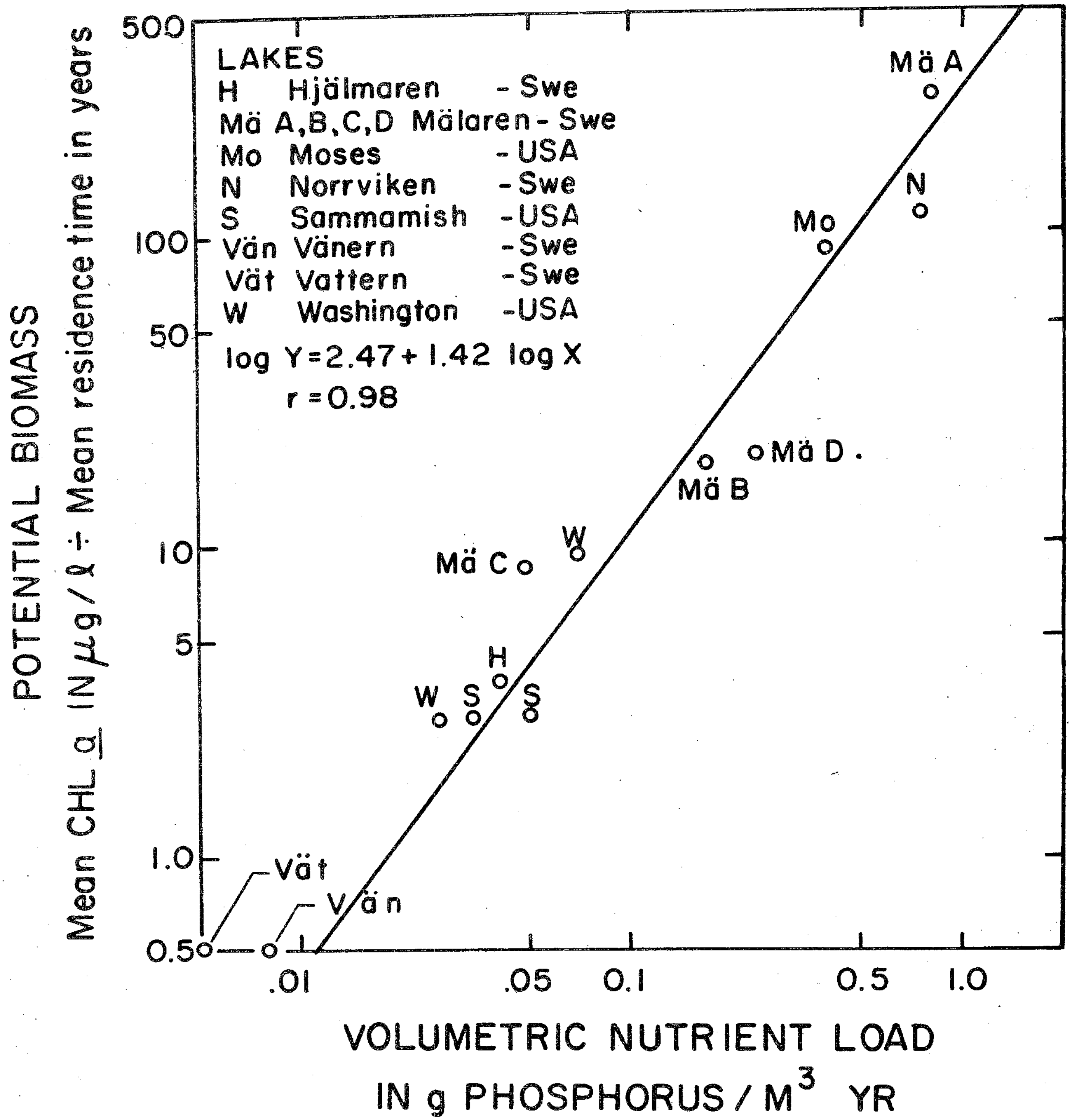

Figure 3. Relationship between volumetric nutrient loading and lake response in terms of potential biomass in a group of U.S. and Swedish lakes. The y values for Vän and Vät are 0.43 and 0.03, respectively and x for Vät is 0.003 (Welch, et al., 1975).

Acknowledgements

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SECTION VII - WISCONSIN

LAKE MENDOTA - NUTRIENT LOADS AND BIOLOGICAL RESPONSE

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INTRODUCTION

Lake Mendota is the largest of the Madison lakes which form a chain along the Yahara River in south-central Wisconsin. It is classified as a hard-water, eutrophic lake according to most standards. The drainage area of Lake Mendota is composed mostly of fertile farm land and the urban area. The hypolimnetic waters become devoid of oxygen during summer stratification. After fall reoxygenation, oxygen depletion again occurs in the bottom waters during late winter. Excessive weed growth and periodic algal blooms create offensive conditions during the summer months.

GEOGRAPHIC DESCRIPTION

Lake Mendota is located in Madison, Wisconsin, the latitude and longitude of the centroid of the water area being 43°7' N and 89°25' W. The surface of the lake stands at an altitude of 849 feet above sea level (Cline, 1965). The lake has a cumulative drainage area of 265 sq. miles (Lee, 1962). The climate of the basin is typically continental, the summers are hot and the winters are cold. The average annual temperature at Madison is 46.2°F and ranges from an average 72.7°F in July. During each of four winter months, December through March, the mean monthly temperature is below 32°F. The growing season extends generally from late April to mid-October and averages 175 days (Cline, 1965).

From 1852-1948, Lake Mendota showed an ice cover duration of 112 days (14 December to 4 April) on the average. Duration of ice cover ranged from 65 days in the winter of 1931-32 to 161 days in 1880-81. The earliest the lake has frozen over is 25 November 1857, and the latest it has thawed is 6 May 1957 (Frey, 1963).

The precipitation varies widely during the year. The maximum average monthly precipitation occurs in June, and the minimum average precipitation occurs in February. Generally 3 to 4 inches of precipitation per month occurs during May through September. Most of this precipitation is associated with thunderstorms. Between one and two inches of precipitation per month generally occurs during November through February. The total yearly precipitation averages 31.2 inches, which includes an average annual snowfall of 37.8 inches or about seven inches precipitation. The evapotranspiration rate from that part of the Yahara River basin covered by lakes and marshes is about equal to the precipitation (Cline, 1965).

Figure 1 is a schematic diagram of Lake Mendota showing depth contours and direction of the prevailing winds in summer. The maximum fetch is about 9 Km and occurs when the wind is out of the southwest.

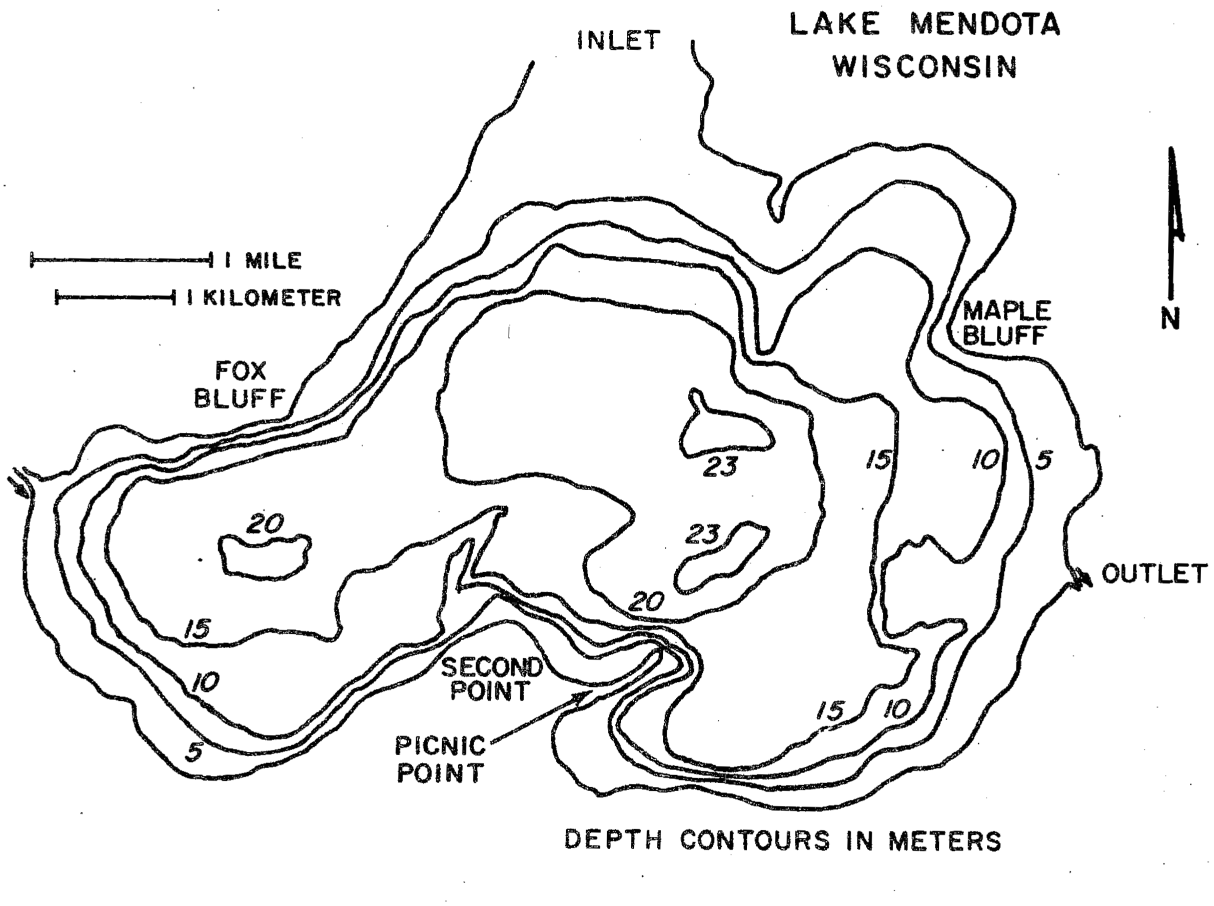

Figure 1. LAKE MENDOTA, WISCONSIN, SHOWING BATHYMETRY

Lake Mendota occupies a pre-glacial valley system excavated by streams in sandstones and sandy dolomites of upper Cambrian age. The lake was formed as a result of moranic damming during the most recent ice age (Twenhofel, 1933). Rocks of Cambrian age, principally sandstone and dolomite, were deposited in shallow seas on a surface of igneous and metamorphic rocks of Precambrian age. Dolomite and sandstone of Ordovician age were deposited on the Cambrian rocks. Glacial drift and loess overlie these formations (Cline, 1965).

Approximately 200,000 people live in Madison on the southeast shore of Lake Mendota. Land usage estimates for the Mendota basin provided by Sonzogni and Lee (1974) are shown in Table 1. A large area of the drainage basin is predominantly agricultural (dairy farms and mixed crops).

Table 1. ESTIMATE OF LAND USE WITHIN THE LAKE MENDOTA WATERSHED*

Land Use	Acres	Percent
Rural	115,000	83
Urban	16,000	12
Marshland	6,000	4
Woodland	1,000	1
Total	138,000	100

*After Sonzogni and Lee (1974)

Lake Mendota water is mainly used for sports, fishing and recreation. A limited amount of lake water is pumped into the University of Wisconsin water supply system. The municipal supply for the City of Madison comes almost entirely from ground water.

In 1958, discharges of treated sewage effluent were diverted around all Madison lakes. By 1973, waste water from several small communities was diverted from Lake Mendota tributaries (Sonzogni and Lee, 1974).

MORPHOMETRIC AND HYDROLOGIC DESCRIPTION

Lake Mendota has a surface area of 15.2 square miles ($39.4 Km^2$), a length of 5.9 miles (9.5Km) and a width of 4.6 miles (7.4Km). The shoreline is 20 miles (32.2Km) long. The water volume is approximately 128×10^9 gallons ($486 \times 10^6 m^3$). Maximum depth of the lake is 84 feet (25m) while the mean depth is 40 feet (12m) (Cline, 1965). Depths greater than 12m occur in about 50 percent of the surface area. Details of the hypsometry of the lake are given in Table 2. The stratification period in Lake

Mendota may extend from May to October. The volume ratio of epilimnion over hypolimnion varied from 0.93 in June to 5.34 in October, 1971. This ratio was 2.66 during August, the time of maximum stability (Stauffer and Lee, 1973).

Table 2. HYPSOMETRIC FACTORS FOR LAKE MENDOTA
SURFACE AREA OF LAKE MENDOTA 39.4 x 10^6 m^2

Depth Meters	Average % of Surface Area Within Interval	Volume Contained Within Interval x 10^{-7} m^3
0 - 0.5	98	1.88
0.5 - 1.5	92	3.60
1.5 - 2.5	87	3.40
2.5 - 3.5	82	3.20
3.5 - 4.5	77.5	3.03
4.5 - 5.5	73.5	2.87
5.5 - 6.5	71	2.78
6.5 - 7.5	69	2.70
7.5 - 8.5	66.5	2.60
8.5 - 9.5	64	2.50
9.5 -10.5	61.5	2.40
10.5 -11.5	58	2.27
11.5 -12.5	54	2.11
12.5 -13.5	51	1.99
13.5 -14.5	47	1.84
14.5 -15.5	45	1.76
15.5 -16.5	42	1.64
16.5 -17.5	39	1.52
17.5 -18.5	35	1.37
18.5 -19.5	29	1.13
19.5 -20.5	22	0.86
20.5 -21.5	15	0.59
21.5 -22.5	9	0.35
22.5 -23.5	4	0.16
23.5 -24.5	0.2	0.01
		48.6 x $10^7 m^3$

The hydraulic residence time for the lake is 4.5 years (Sonzogni and Lee, 1974). A water balance of Lake Mendota for October 1, 1948 to October 1, 1949, is given in Table 3.

Abrupt sedimentation changes have occurred in Lake Mendota in the recent past. Buff marl is overlain by black gyttja, gray-colored gyttja-marl forms the interface between buff marl and balck gyttja. The marl and gyttja differ in being high carbonate-low clastic and low carbonate-high clastic sediments, respectively.

Table 3. WATER BALANCE OF LAKE MENDOTA,
1 OCTOBER 1948 TO 1 OCTOBER 1949*

INFLOW	
a) Measured tributaries -- Stations 1-10 (192.27 mi^2 of drainage basin)	78.4 cfs
b) Unmeasured tributary area (31.87 mi^2 of drainage basin, by computation)	13.0 cfs
c) Precipitation onto lake surface (31.65 in.)	38.7 cfs
Total Inflow (4,100,000,000 ft^3)	
OUTFLOW	
a) Storage	3.5 cfs
b) Evaporation (51.17 in., by computation)	58.2 cfs
c) Outflow, Station 11	70.0 cfs
d) University pumpage	1.5 cfs
Total Outflow (4,200,000,000 ft^3)	133.2 cfs

Unaccounted for: 133.2-130.0 = 3.1 cfs

= 2.33 % of outflow

*After Rohlich, in Frey (1963)

The change in sedimentation is ascribed to increased deposition of clastic material in the lake as a consequence of farm and domestic practices (Murray, 1956). Cores of Lake Mendota show increased deposition of P, Fe, K, and Organic-C while carbonate-C has decreased in most recent periods (Bortleson and Lee, 1972).

CURRENTS

Bryson and Ragotzkie (1955) found that University Bay (Lake Mendota) was normally occupied by a clockwise gyre, the rotation rate of which is nearly constant (period=0.5 pendulum day) and independent of the current volocity. A jet of particularly high velocity extends out into the lake along the side of Picnic Point peninsula. Clarke and Bryson (1959) found that, following diminution of stress from the surface wind, a countercurrent rapidly develops below the surface as observed at Second Point Bar. Shulman and Bryson (1961) found that wind driven currents deviate to the right of the wind in a pattern which fits the logarithmic spiral hodograph of classical theory. Density currents were observed by Bryson and Suomi (1952) in Lake Mendota. Turbid runoff following periods of rain either flows along the bottom and spreads at the thermocline or moves deep into the hypolimnion as dictated by density relations.

LIMNOLOGICAL CHARACTERIZATION

In Lake Mendota the temperature of the water ranges from $0^{\circ}C$ to $27^{\circ}C$. Specific conductance varies from 250 to 350 umhos/cm at $20^{\circ}C$. True color of the lake water is from 5 to 15 mg/l chloroplatinate. During the summer months very little light penetrates below 4-6m. Turbidity caused by the biomass acts to absorb and scatter incident light. On May 13, 1971, light penetrated all the way to the bottom of Lake Mendota and 21 percent of the surface light reached 4m. Light continued to reach the bottom until mid-June and after July 14, 1971, less than 4 percent of the surface light reached 4m (Torrey, 1972).

Table 4 presents a typical chemical analysis of Lake Mendota, giving the range of value of the most important parameters of water quality (Lee, 1966).

Algal populations of Lake Mendota include the bloom-forming *Microcystis*, *Oscillatoria*, and *Lyngbya*. In addition, the acetylene-reducing genera *Anabaena*, *Aphanizomenon*, *Nostoc*, *Calothrix*, and *Gloeotrichia* are commonly observed (Torrey, 1972). Typical blue-green algae counts for the summer are on the order of 10^6 cells/liter. Total lake chlorophyll during the summer months averages 5,000Kg, with maximum values reaching 8,000Kg of chlorophyll. The average and maximum chlorophyll concentrations per unit area for the lake are 125 mg/m^2 and 200 mg/m^2, respectively (Stauffer and Lee, 1973). Primary production, calculated from light intensity and chlorophyll data (Ryther and Yentsch, 1957), is on the order of $4gC/m^2/day$ during the summer.

Table 5 summarizes identities and counts of common summer zooplankters in Lake Mendota. These include species of *Daphnia*, *Cyclops*, *Copepods*, *Diaptomus*, *Chydorus*, *Lecane* and *Asplanchna* (Frey, 1963).

Fish populations in Lake Mendota have changed since the turn of the century. Two major changes that have been found are the decrease in number and increase in average size of perch. Cisco, once a very abundant fish in Lake Mendota, has almost reached extinction; only rare occurrences have been reported in recent years. The disappearance of Cisco from Lake Mendota has been attributed to increase fertility of the lake. Conway (1972) has shown that the rate of dissolved oxygen depletion in the hypolimnetic waters of Lake Mendota has increased significantly from the early 1900's to the present. Since

Table 4. TYPICAL ANALYSIS OF LAKE MENDOTA, 1965-1966

Based on a 1.5 Year Study by Students and Staff of Water Chemistry Program, University of Wisconsin-Madison*

Water	Range**
Temperature $^{\circ}C$	0 - 27
Specific Conductance umhos/cm at 20 C	250 - 350
pH	6.5 - 9.2
Turbidity ppm SiO_2	10 - 50
Color (true) chloroplatinate mg/l	5 - 15
Sodium mg/l	4.5 - 8.0
Potassium mg/l	3.5 - 4.0
Magnesium mg/l	23 - 28
Calcium mg/l	26 - 30
Nitrate mg N/l	0 - 0.7
Nitrite mg N/l	0.0025 - 0.02
Ammonia mg N/l	0.04 - 0.9
Organic mg N/l	0.5 - 5.0
Total Phosphate mg/P/l	0.05 - 0.65
Orthophosphate mg/P/l	0.02 - 0.4
Dissolved Solids mg/l	200 ± 20
Filterable Solids mg/l	10 - 60
Silicon Dioxide mgSiO_2/l	0.1 - 1.5
Chloride mg/l	6.2 - 9.6
Iron mg/l	0.02 - 0.2
Manganese mg/l	0.005 - 0.5
Dissolved Oxygen mg/l	0 - 15
Chemical Oxygen Demand mg/l	7.0 - 20
Dissolved Organic Carbon mg/l	10 ± 1
Flouride mg/l	0.09 - 0.25
Alkalinity total mg/l of $CaCO_3$	140 - 193
Sulfate mg/l	18 - 30

*Compiled by Lee, 1966.

**Concentrations dependent on sampling location and date.

the hypolimnion of Lake Mendota has become completely anoxic each year since the early 1900's, it is felt that the Cisco inhabited a narrow layer of water just below the thermocline. The oxygen in this layer is maintained by diffusion through the thermocline. By the early 1940's, the rate of oxygen depletion in this region of the lake was sufficient to cause anoxic conditions immediately below the thermocline with the result that the Cisco were deprived of their niche and died in large numbers. Of the 61 species of fishes reported for Lake Mendota, 60 are listed among the 173 species in 29 families found in the Great Lakes drainage. The families Cyprinidae and Centrarchidae contribute the largest number of species. The Percidae and Ictaluridae are also well represented (McNaught, 1963).

Changes in the populations of aquatic macrophytes have also occurred. In the past 50 years *Myriophyllum spicatum* has invaded the lake and is presently the most abundant aquatic vascular plant in Lake Mendota. In 1921, the most abundant species, in descending order of abundance, were *Vallisneria spiralis*, *Najas flexidis*, *Potomogeton Richardsonii*, *P. zosteriformis* and *P. pectinatus* (Nichols and Mori, 1971).

Table 5. DISTRIBUTION OF COMMON SUMMER ZOOPLANKTERS IN LAKE MENDOTA (SUMMER, 1947)

Based on Clarke-Bumpus and Juday trap samples at 0-1 meter: m, mean number of organisms per liter; s^2, single haul variance.*

Species	Clarke-Bumpus		Juday Trap	
	m	$s^2(s^2/m)$	m	$s^2(s^2/m)$
LAKE MENDOTA				
Daphnia longispina	36	106(2.9)	26	67(2.6)
Diaptomus sp.	10	29(2.9)	11	38(3.4)
Cyclops viridis	6	4(0.7)	7	13(1.9)
Copepod nauplii	20	15(0.8)	39	180(4.6)
Chydorus sp.	26	63(2.4)	31	70(2.3)
Lecane sp. (Rotatoria)	24	75(3.0)	35	126(3.6)
Asplanchna sp. (Rotatoria)	9	8(0.9)	11	38(3.4)

*After Hasler, in Frey (1963).

NUTRIENT BUDGETS

Estimated nutrient sources for Lake Mendota after the 1971 diversion of wastewater discharges are listed in Table 6 (Sonzogni and Lee, 1974). A breakdown by chemical species of the nutrients input from each source is provided. From these data a generalized nutrients budget for the lake can be obtained (Table 7). The phosphorus and nitrogen loadings are 1.2 g $P/m^2/yr$ and 13 g $N/m^2/yr$, respectively.

Table 6. ESTIMATED NUTRIENT SOURCES FOR LAKE MENDOTA - 1972

Lbs/year (to nearest 1000 lbs)*

	Soluble O-P	Total P	NH_4^+-N	NO_3^--N	Org N	Total N
Waste water discharge	1,800	2,000	6,600+		3,000	6,900
Urban runoff	9,000	16,000	7,000	10,000	56,000	73,000
Rural runoff	34,000	69,000	320,000*		200,000	520,000
possible range		(12,000-115,000)	(58,000-690,000)*			(115,000-1,150,000)
Precipitation on lake surface	2,000	2,000	24,000	28,000	17,000	69,000
Dry fallout on lake surface	1,000	7,000	35,000	30,000	70,000	135,000
Ground water seepage	1,000	1,000		171,000		171,000
Base flow	12,000	12,000		135,000		135,000
Nitrogen fixation	-	-	-	-	88,000	88,000
Woodland runoff	0	0	0	0	0	0
Marsh drainage	0	0	0	0	0	0
Total	60,800	109,000	766,600		448,400	1,197,900

\+ Inorganic nitrogen
* After Sonzogni and Lee (1974)

Table 7. NUTRIENT SOURCES FOR LAKE MENDOTA*

A. Phosphorus	Kg/yr
Domestic Wastewaters	908
Urban Runoff	7,264
Rural Runoff	31,326
Precipitation	908
Dry Fallout	3,178
Ground Water	454
Base Flow	5,448
TOTAL	49,486

B. Nitrogen	Kg/yr
Wastewater	3,133
Urban Runoff	33,142
Rural Runoff	236,080
Precipitation	31,326
Dry Fallout	61,290
Ground Water	77,634
N-Fixation	39,952
Base Flow	61,290
TOTAL	543,847

*After Sonzogni and Lee (1974).

Despite the wastewater diversion, the mean phosphorus content of Lake Mendota has increased during the period between 1970-1973. Data collected by Sonzogni (1974) presented in Table 8, show the mean phosphorus content of the lake for this period. The calculated mean phosphorus lake concentration, based on these values and the total lake volume (486 x $10^6 m^3$), ranges from 0.12 to 0.15 mg/l. A mean value of 0.10 mgP/l is obtained if one divides the estimated phosphorus input by the volume of the lake. This is interesting in that it appears that one could predict the mean annual total phosphorus concentration for Lake Mendota based solely on input data.

Table 8. MEAN MONTHLY, WINTER AND ANNUAL PHOSPHORUS CONTENTS FOR LAKE MENDOTA*

Mean Content	1970-1971 kg x 10^{-4} DRP	TP	1971-1972 kg x 10^{-4} DRP	TP	1972-1973 kg x 10^{-4} DRP	TP
Aug	5.7	7.0	5.3	6.9	6.2	8.1
Sept	5.5	7.2	6.3	8.0	6.2	7.6
Oct	5.8	7.2	5.6	7.3	6.4	7.7
Nov	5.3	5.9	5.5	6.9	6.1	6.9
Dec	4.9	5.8	5.2	6.4	6.2	6.9
Jan	5.2	5.9	4.2	6.0	6.2	7.1
Feb	5.0	5.8	4.3	5.4	6.5	7.1
March	5.3	6.1	4.7	5.6	7.2	8.4
April	1.7	4.6	4.9	6.4	7.1	8.1
May	1.9	3.0	4.6	6.0	6.4	7.2
June	3.6	4.8	4.2	5.8	4.5	6.6
July	4.8	6.3	5.1	6.9	3.9	6.2
Winter[a]	5.1	5.8	4.6	5.9	6.3	7.0
Annual[b]	4.6	5.8	5.0	6.5	6.1	7.3

[a]Average for Dec, Jan and Feb

[b]Average for Aug through July

*After Sonzogni (1974)

DISCUSSION

A summary of available nutrient loading data is presented in Table 9. The Vollenweider (1974) loading curve values based on the data presented here, are 1.2 gP/m^2/yr for phosphorus loading and 2.7 m/yr for discharge height, q_s. When these values are plotted in the loading vs. q_s plot (Vollenweider, 1974), Lake Mendota falls in the eutrophic category (Figure 2). In addition, this plot shows Lake Mendota to be in a higher eutrophic state than was thought in 1965. It should be noted that Vollenweider (1974) uses a q_s value significantly smaller than the 2.7 m/yr which was an error in his original phosphorus loading lake response curve relationships.

Table 9. SUMMARY OF AVAILABLE NUTRIENT LOADING DATA

(1) Nutrient Loadings			
Total Phosphorus	49,500 Kg/yr	1.2 $g/m^2/yr$
Total Nitrogen	543,800 Kg/yr	13 $g/m^2/yr$
(2) Chlorophyl-a		
Average total in lake	5,000 Kg	125 mg/m^2
Maximum total in lake	8,000 Kg	200 mg/m^2
Average for euphotic zone		25 mg/m^2
Maximum for euphotic zone		40 mg/m^2
Average primary production (estimated)		4g $C/m^2/day$
(3) Physical Features of Lake Mendota		
Mean Depth	12 m	
Maximum Depth	23 m	
Area	39.4 Km^2	
Total Volume	486 x $10^6 m^3$	
Depth of Euphotic Zone	3 m	
Volume of Euphotic Zone	90 x $10^6 m^3$	
Hydraulic filling time	4.5 yr	
Discharge height, q_s	2.7 m/yr	

Examination of chemical characteristics through an annual cycle, and algal assay studies such as those of Walton and Lee (1972) as well as others at the University of Wisconsin at Madison (G. P. Fitzgerald), have shown the phosphorus concentration of the water is the key factor governing the excessive growth of planktonic and attached algae during the summer. Stauffer and Lee (1973) have demonstrated that many of the obnoxious blue-green algal blooms that occur in summer are caused by thermocline downward migration. This results in the transport of hypolimnetic phosphorus to the epilimnion.

According to Vollenweider (1974), the current phosphorus loading of Lake Mendota is about ten times the "permissible" loading. Because the lake receives its nutrients primarily from diffuse rural sources (Table 7), reduction of phosphorus loading to the "permissible" level does not appear to be likely. In fact, it appears that technical, economic and political factors involved would make significant reduction of these loads difficult, if not impossible (Lee, 1972).

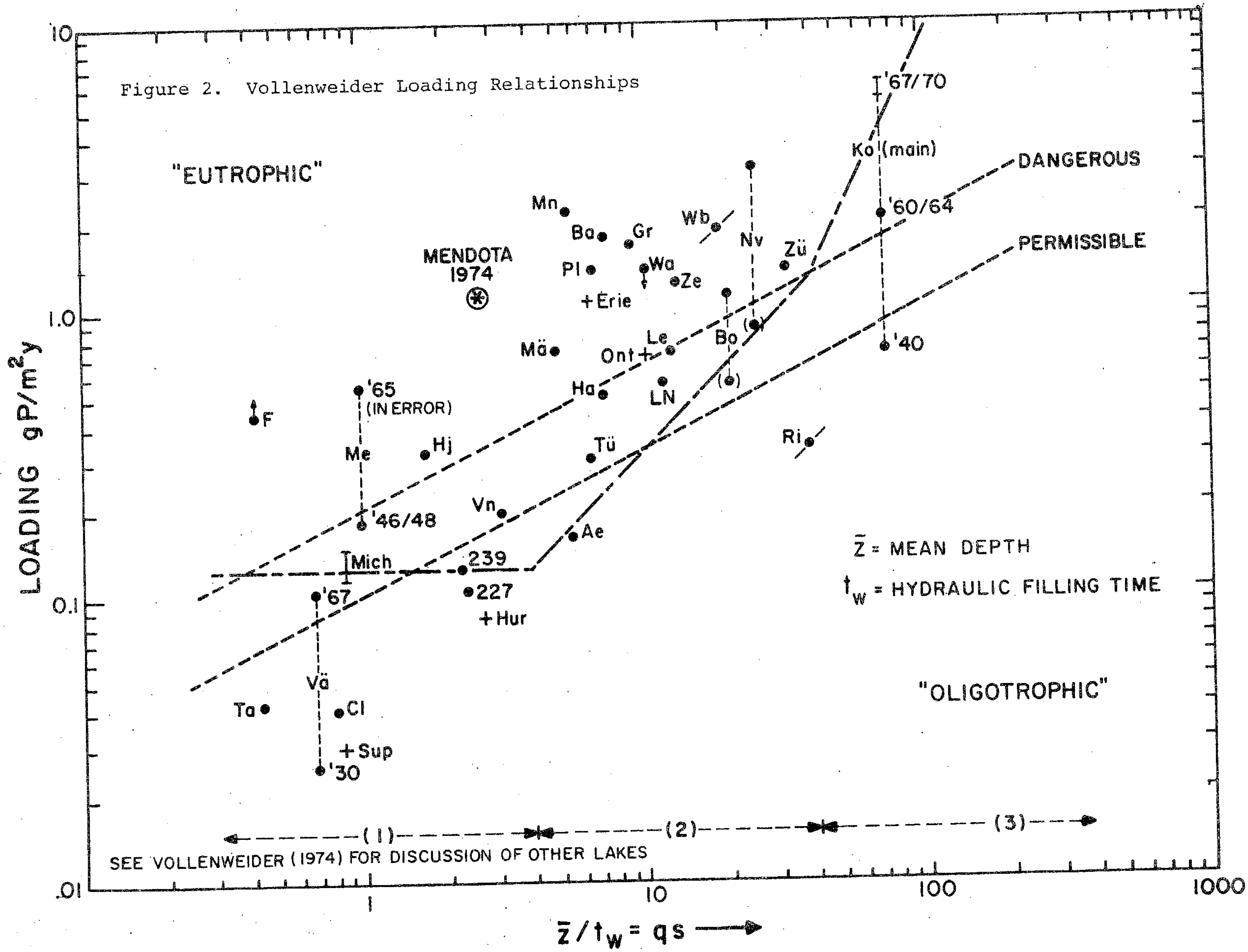

Figure 2. Vollenweider Loading Relationships

In order for the public to perceive a significant reduction in the frequency and severity of excessive blooms of planktonic blue-green algae, the phosphorus loadings would have to be decreased by at least a factor of five. Yet according to Vollenweider (1974), despite a reduction of this magnitude, the lake would still have a "dangerous" phosphorus loading level.

From an overall point of view, as a result of increased urbanization of the Lake Mendota watershed, it is highly likely that water quality in the lake will slowly deteriorate. Increased frequency of severe blue-green algal blooms and excessive growth of attached algae and macrophytes in littoral areas during the summer can be expected. Based on the current technology, it appears that efforts to control water quality deterioration from excessive fertilization of the lake should be directed towards maximal reduction of the phosphorus input from agricultural drainage and urban storm drainage. With respect to the former, particular emphasis should be given to controlling phosphorus input from animal manures associated with dairy farming. While it appears very unlikely that efforts to curb phosphorus input to Lake Mendota will create a measureable improvement in lake water quality, they probably will benefit area residents by slowing the deterioration.

As Lee noted in 1972, the use of chemicals such as alum to precipitate phosphorus could improve Lake Mendota water quality. This potential solution is both technically and economically feasible. The evidence available today clearly indicates that such a procedure would reduce significantly the frequency of the severe blue-green algal blooms that occur each summer. While alum should be used to treat the open waters of the lake, simultaneously a combination of mechanical weed harvesters and aquatic herbicides should be used to control excessive attached algae and macrophyte growth in selected areas of the lake.

It is technically and economically feasible to improve the water quality of Lake Mendota through judicious use of chemicals to control both excessive phosphorus and excessive plant growth. However, it is not politically feasible at this time because environmental activist groups wield sufficient political power to prevent use of such techniques although they have proven highly successful elsewhere.

ACKNOWLEDGMENTS

Information which serves as a basis for this report was collected through the support of numerous granting agencies. However, primary support for many of these studies was derived from US EPA and its predecessor organizations. In addition, substantial support to these studies was given by the University of Wisconsin-Madison, especially the Departments of Botany, Zoology, and Civil and Environmental Engineering. Special recognition is given to the assistance of J. Magnuson and A. D. Hasler of the University of Wisconsin-Madison laboratory of limnology for their help in compiling various reports used in this investigation.

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REPORT ON NUTRIENT LOAD - EUTROPHICATION RESPONSE OF LAKE WINGRA, WISCONSIN

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INTRODUCTION

Lake Wingra is the smallest of the lakes of the Yahara River chain at Madison, Wisconsin. It is a shallow, hardwater eutrophic lake. The drainage area of Lake Wingra is composed of the University of Wisconsin Arboretum and a portion of the urban region of southwest Madison.

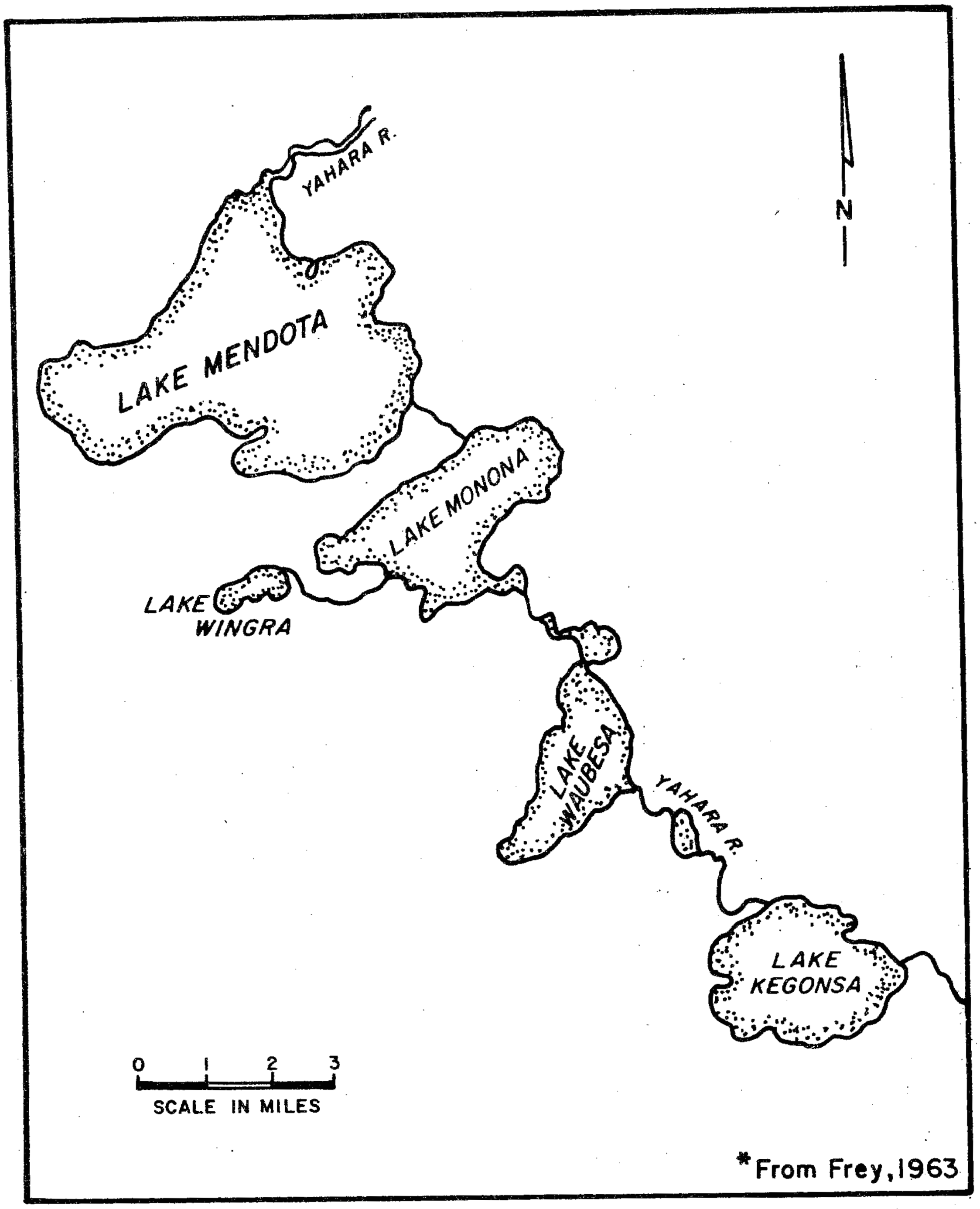
GEOGRAPHIC DESCRIPTION

Lake Wingra is located within the city of Madison, Wisconsin. The latitude and longitude of the centroid of the water area are 43°04.2' N and 89°23.6' W, respectively. The lake surface stands at an altitude of 848 feet above sea level (Huff et al., 1973). The size and location of Lake Wingra relative to the other lakes of the Yahara River chain are illustrated in Figure 1. The drainage basin is composed of a portion of the urban region of southwest Madison, and the University of Wisconsin Arboretum, approximately one-third of which drains directly to the lake. The urban area enclosed by the Lake Wingra drainage basin comprises residential middle- and upper middle-class homes (Kluesener and Lee, 1974). The lake has a cumulative drainage area of 8.1 mi^2 (2.1×10^7 m^2) including lake surface area, but 2.2 mi^2 (9×10^6 m^2) drain directly into Murphy Creek, bypassing Lake Wingra proper. Murphy Creek is the major outlet from Lake Wingra and flows directly into Lake Monona.

CLIMATE

The climate of the basin is typically continental. The summers are hot and the winters cold. The annual average temperature at Madison is 46.2°F and ranges from an average of 17.7°F in January to an average of 72.7°F in July. During

Figure 1
Lakes of the Yahara River chain at Madison, Wisconsin*



*From Frey, 1963

the winter months, December through March, the mean monthly temperature is below 32°F. The growing season generally extends from late April to mid-October and averages 175 days (Cline, 1965).

Although the record for Lake Wingra is incomplete, from 1877 to the present the lake froze over, on the average, on November 25 and thawed on March 29, for an average ice-bound period of 125 days (Noland, 1950, cited in Frey, 1963).

The precipitation varies widely during the year. The maximum and minimum monthly precipitation occur in June and February, respectively. During May through September, there are generally three to four inches of precipitation occurring per month. Between one and two inches of precipitation per month usually occurs during November through February. The total yearly precipitation at Madison averages 31.2 inches, which includes an average annual snowfall of 37.8 inches, or about seven inches of precipitation. The evapo-transpiration rate from that part of the Yahara River basin covered by lakes and marshes is about equal to the precipitation (Cline, 1965).

GEOLOGIC DESCRIPTION

Lake Wingra occupies a pre-glacial valley system evacuated by streams in sandstones and sandy dolomites of upper Cambrian Age. The lake was formed as a result of morainic damming during the most recent ice age (Twenhofel, 1933). Rocks of Cambrian Age, principally sandstone and dolomite, were deposited in shallow seas on a surface of igneous and metamorphic rocks of Precambrian Age; dolomite and sandstone of Ordovician Age were deposited on the Cambrian rocks. Glacial drift and loess overlie these formations (Cline, 1965).

Most of Lake Wingra's immediate shores are swamp and bog, and its shoreline material is mostly the lake's own organic deposits (Murray, 1956).

CHARACTERISTICS OF WATERSHED

Most of the vegetation surrounding Lake Wingra is that of the University of Wisconsin Arboretum. It is composed principally of coniferous and decidous forests, prairies, gardens and marshes. The Arboretum comprises approximately 800 acres, 20 percent of the drainage area of the lake. Of this 800 acres, approximately 470 acres consist of forests, while the remaining 330 acres consist of prairies, gardens and marshes (Kluesener, 1972).

Approximately 200,000 people reside in Madison, Wisconsin, on the northeast shore of Lake Wingra. Land usage in the Lake Wingra drainage basin is summarized in Table 1.

Table 1. LAND USAGE IN LAKE WINGRA DRAINAGE BASIN*

	Area	
	Acres	Hectares
Lake Wingra Basin	5200	2104
Area draining to Lake Wingra	3800	1538
Residential Area	2600	1052
Arboretum	775	314
Lake and Ponds	337	136
Area draining directly to Murphy Creek	1400	566

*After Kluesener, 1972

The residential drainage area consists mainly of storm sewer drain outlets, and also includes approximately 400 acres of golf courses and cemeteries (Kluesener, 1972).

Lake Wingra is used mainly for sports, fishing and recreation. There have never been any sewage or industrial discharges into Lake Wingra (Sonzogni, 1974) except for occasional sewer overflow due to failure of the sewage pumping station.

MORPHOMETRIC AND HYDROLOGIC DESCRIPTION

Lake Wingra has a surface area, including lagoons and ponds, of 137 hectares (1.37×10^6 m^2). It has a maximum length of 2.09 km, a maximum effective length of 2.16 km, a maximum width of 1.11 km and a mean width of 0.63 km. The shoreline is 5.91 km long. It has a shoreline development figure of 1.45 and a development of volume of 1.19. The water volume, including ponds and estimated lagoon volume, is 3.35×10^6-m^3. The maximum depth of the lake is 6.10 m and the mean depth (volume/surface area) is 2.42 m (Figure 2) (Huff et al., 1973). Because of its shallow mean depth (2.42 m), the lake does not permanently thermally stratify; the epilimnion generally extends to the bottom all year round (Murray, 1956).

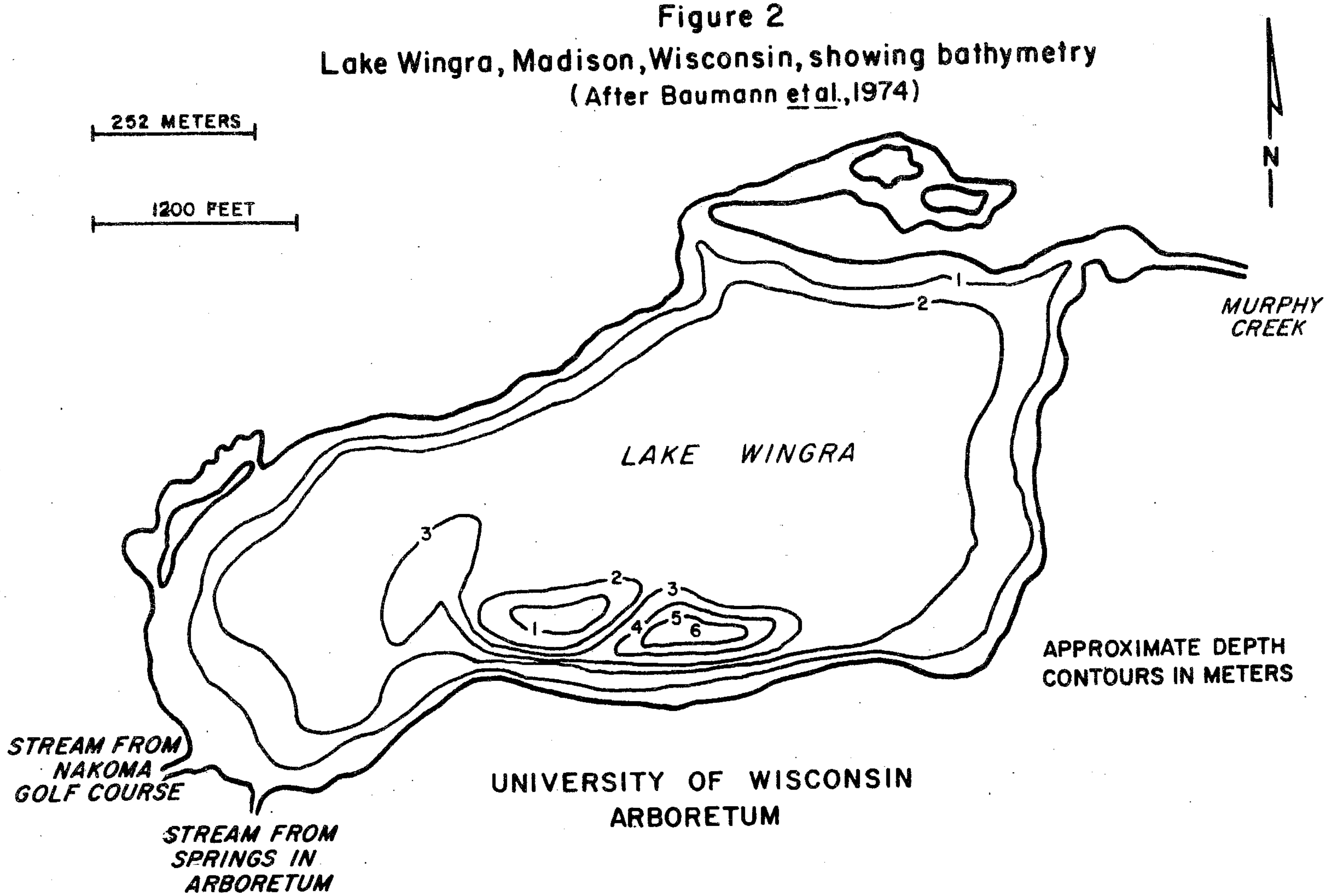
CHARACTERISTICS OF SEDIMENTS

The sediments of Lake Wingra have been studied extensively. The shore and bottom deposits are predominantly marl. In the sixties, Frey (1963) established that the recent bottom consists of gray marl, which becomes shell marl in shallow water. Murray (1956) established that the top six inches of recent sediment are a gray to dark gray marl. At least

Figure 2

Lake Wingra, Madison, Wisconsin, showing bathymetry

(After Baumann et al., 1974)



some locations in the lake contain abundant gastropod shells and clam shell fragments. The carbonate content is approximately 54 percent in the most recent sediments. The organic matter present in the sediment ranges from 11.7 to 13.5 percent (similar in organic content to the black sludge and buff marl of Lake Mendota). The clastics appear to be concentrated in the fine sizes. The surrounding bogs of Lake Wingra permit very little clastic deposition because little clastic material is available on the shore. The lack of black sediments similar to those found in Lake Mendota is thought to be evidence that Lake Wingra has a constant availability of oxygen throughout the epilimnion, which extends to the bottom. A sediment core analysis was conducted by Bortleson (1970), and the results of the upper five centimeters are summarized in Table 2.

Table 2. SEDIMENT ANALYSIS FOR THE UPPER 5 CM OF DRY SEDIMENTS*

Component	Lake Mendota (mg/g dry wt)	Lake Wingra (mg/g dry wt)
Fe	20-25	9
N	10	7-8
P	2	0.6
Ca	125	230-240

*After Bortleson, 1970

More recently, Bannerman (1973) examined Lake Wingra sediments in some detail for interstitial concentrations of inorganic phosphorus. He found that levels of total phosphorus, total inorganic phosphorus and total organic phosphorus present in core samples from both the open water areas and littoral zone were in good agreement with values for Lake Wingra sediments reported previously by others (Williams et al., 1971; Li et al., 1972). By contrast, sediments from lakes Mendota and Monona revealed levels of phosphorus approximately twice as large as those in Lake Wingra (Williams et al., 1970). It is believed this is due to the nature of the input waters into these lakes. Lake Wingra receives primarily urban runoff (Lee and Kluesner, 1971), while Lake Mendota and Lake Monona receive a combination of urban and agricultural runoff.

HYDROLOGY

The hydraulic residence time (water body volume/annual inflow volume) was calculated to be 0.44 years, based on data from Kluesner (1972) and Huff et al. (1973). The annual input for Lake Wingra is the sum of the precipitation, springflow, urban runoff and groundwater input. The USGS data as reported by Kluesner (1972) was used for the annual precipitation, springflow and urban runoff inputs (1×10^6 m^3/yr, 1.4×10^6 m^3/yr and 1.0×10^6 m^3/yr, respectively). Kluesner did not report the groundwater input to the lake. As the groundwater input is considered to be essentially constant, Huff's groundwater input for the period April 10 to September 15, 1970 (158 days) was extrapolated to a full year ($\approx 2.3 \times 10^6$ m^3/yr) and used in the calculation of the hydraulic residence time. Thus, the hydraulic residence time is calculated to be 0.44 years (i.e., lake volume (2.5×10^6 m^3)/total annual input (5.7×10^6 m^3/yr)).

A water balance summary for Lake Wingra for the period April 10, 1970 to September 15, 1970 is presented in Table 3.

Table 3. LAKE WINGRA WATER BALANCE SUMMARY*
(April 10, 1970 to September 15, 1970)

Period	Precipitation (m^3)	Evaporation (m^3)	Groundwater Inflow (m^3)	Groundwater Loss (m^3)	Surface Inflow (m^3)	Lake Outflow (m^3)	Lake Surface Change (m^3)
Apr 10-30	81,000	81,400	230,600	25,700	73,000	282,100	-4,600
May 1-31	211,000	142,000	242,800	38,000	218,600	434,200	58,200
Jun 1-30	78,300	177,000	205,800	36,800	28,300	184,000	-85,400
Jul 1-31	83,800	183,300	137,400	151,900	41,800	6,800	-79,000
Aug 1-31	33,600	158,300	117,100	227,800	14,900	0	-220,500
Sep 1-15	189,100	46,100	78,600	169,000	194,900	24	247,500
TOTAL	676,800	788,100	1,012,300	649,200	571,500	907,100	-83,800
INFLOW PERCENT	29.9	---	44.8	---	25.3	---	---
OUTFLOW PERCENT	---	33.6	---	27.7	---	38.7	---

PERIOD TOTAL INFLOW = 2,260,600 m^3

PERIOD TOTAL LOSSES (OUTFLOW) = 2,344,400 m^3

UNACCOUNTED FOR: 2,344,400 - 2,260,600 m^3 = 83,800 m^3 = 3.58 PERCENT OF OUTFLOW

*After Huff et al., 1973

PHYSICAL AND CHEMICAL DESCRIPTION

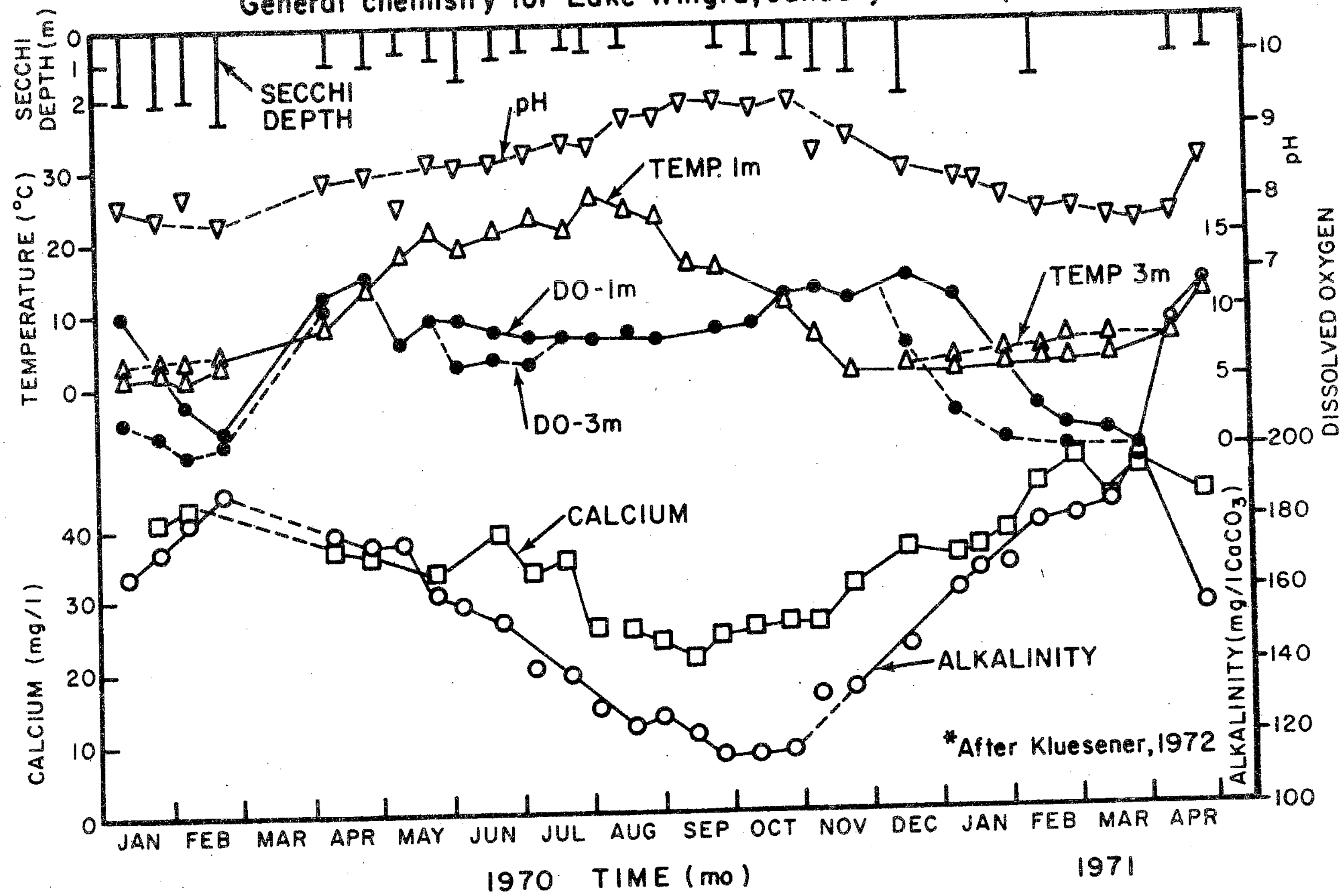
Dissolved oxygen studies (Kluesener, 1972) have shown that the lake is poorly mixed in the vertical plane when covered with ice. This is contrasted with relatively infrequent vertical stratification, with respect to dissolved oxygen, during the summer.

Lake Wingra was sampled for temperature, pH, dissolved oxygen (DO), alkalinity, calcium and total phosphorus and nitrogen, including N and P species, at four open water and four littoral zone stations. Sampling frequency was limited to collecting samples every two weeks. The sampling period ranged from twelve to eighteen months, depending on the parameter studied. Samples were collected at the one and two meter depths, with occasional sampling at three meters if the lake was sufficiently high. Kluesener (1972) provides the details on the characteristics of the sampling program, analytical methods and data obtained in the study. A summary of Kluesener's (1972) results are presented below.

Throughout most of the year the temperature was constant over the extent of the water column (Figure 3). Differences were seen at 1 m and 3 m depths during the winter as the bottom waters gained some heat from the lake sediments. After ice-out the lake warmed rapidly and was about 13°C by the end of April of both 1970 and 1971. The normal summer temperatures averaged approximately 23°C until the cooling trend began in September (Kluesener, 1972). In the winter when the water is clear, the Secchi disk reading was approximately 2 m (Figure 3). It was reduced to about 1 m after ice-out in both 1970 and 1971. From May to September in both 1970 and 1971, it was further reduced to 0.6-0.7 m.

Figure 3

General chemistry for Lake Wingra, January 1970-April 1971*



Light intensity was measured throughout the ice-free period in the air above the water, just below the water surface and at the depths of 0.5, 1.0 and 2.0 m. Between April and August, 1970, average values were 32 percent, 13 percent and 3.2 percent transmitted through the first 0.5 m, the upper 1 m and the upper 2 m, respectively (Table 4). The corresponding Secchi disk reading was about 0.85 m.

The pH (Figure 3) averaged 7.7 throughout the winter of 1970. It increased after ice-out to a maximum of 9.4 during September and October, and then decreased to 7.7 again by March, 1971.

The annual variations in the dissolved oxygen content for 1 and 3 m are also given in Figure 3. The dissolved oxygen (DO) content was approximately zero by late winter of both 1970 and 1971 and remained at that level until ice-out. In the upper meter of the lake water column, the DO decreased at a relatively constant rate in January and February of both 1970 and 1971. The average oxygen depletion rate was approximately 0.18 mg DO/l/day. In March, 1971, the oxygen depletion rate was much slower, and it took nearly the entire month to remove the remaining 1.5 mg/l DO at the 1 m depth. The DO then rose sharply as soon as ice-out occurred, and the water column remained essentially saturated at all depths throughout the open water period. The only exceptions were moderate DO stratifications during sampling in June and July, 1970. In the mornings, the DO in the littoral sampling stations showed a slightly higher concentration than in the main body of the lake. Also, the DO showed a definite increasing tendency in progressing from the first lake station to the final station. It is believed this higher DO concentration in the littoral zones was due to the presence of macrophytes, especially *Myriophyllum*, in

Table 4. PERCENT OF LIGHT MEASURED JUST ABOVE THE SURFACE OF LAKE WINGRA WHICH ACTUALLY PENETRATES TO A SPECIFIED DEPTH (0.5, 1.0, 2.0 m) IN THE LAKE*

Date	Depth 0.5 m	1.0 m	2.0 m	Secchi Depth m
Apr 11 1970	33	16	4	1.0
Apr 27	33	21	7	1.0
May 11	26	7.3	2	0.75
May 25	26	11		0.90
June 5	63	32	12	1.5
June 22	28	10	3	0.85
July 6	26	10	2	0.75
Aug 3	35	12	2	0.65
Aug 17	18	7		0.60
Aug 31	34	9	1	0.60
Sep 14	44	19	6	0.65
Sep 28	44	16	4	0.70
Oct 12	37	12		0.80
Oct 26	55	22	7	1.05
Nov 23	65	30	10	
Apr 13 1971	48	24		0.65
Apr 26	38	18		0.95

*After Kluesener, 1972

the littoral zone throughout the winter, as these organisms are able to undergo photosynthesis even under ice and at light intensities below those required for algae (Kluesener, 1972). A relatively sharp oxygen stratification between the 1 and 3 m depth in the winter indicates this lake is very poorly mixed in the vertical plane when covered with ice.

The data for the average phosphorus concentrations are summarized in Figure 4. After mid-January, the total phosphorus (t-P) concentration of the lake remained nearly constant at approximately 0.06 mg P/l. The annual average t-P concentration was 0.07 mg P/l; the annual average dissolved reactive phosphorus (DRP) concentration was 0.02 mg P/l. The average DRP concentration was approximately 0.08 mg P/l throughout the growing season, and nearly 0.04 mg P/l at the end of the winter season.

The average annual variation for the nitrogen species in Lake Wingra is summarized in Figure 5. The total nitrogen concentration varied from 1.0 to 1.8 mg N/l for the study. The inorganic nitrogen (i.e., NO_3^--N and NH_4^+-N) concentration averaged 1.51 mg N/l for the entire year and 1.01 mg N/l for the growing season (i.e., May through September).

Comparison of both the phosphorus and nitrogen data with earlier studies (Domogalla and Fred, 1926; Tressler and Domogalla, 1931; and Clesceri, 1961) suggests that there has been little change in the average levels of nitrogen and phosphorus in Lake Wingra in the last 45 years (Kluesener, 1972).

The inorganic nitrogen/dissolved reactive phosphorus and atomic ratios during the annual cycle and during the growing season, are greater than 30. As a critical N:P atomic ratio of 16 or greater in natural waters is indicative of phosphorus limitation, this suggests that Lake Wingra is phosphorus limited with respect to aquatic plant nutrients.

Figure 4

Average concentration of phosphorus in the open water of Lake Wingra from January 1970 to June 1971*

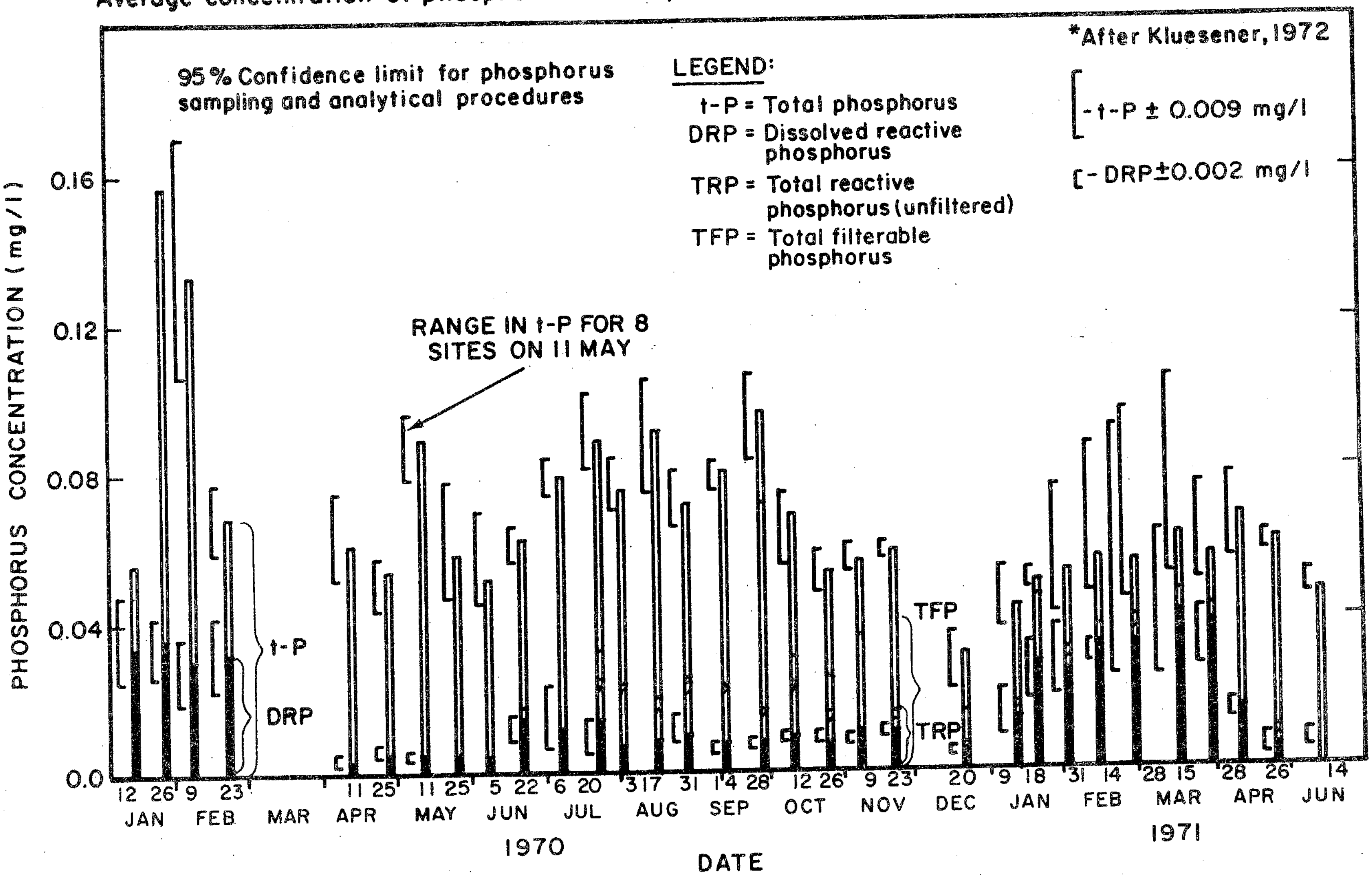
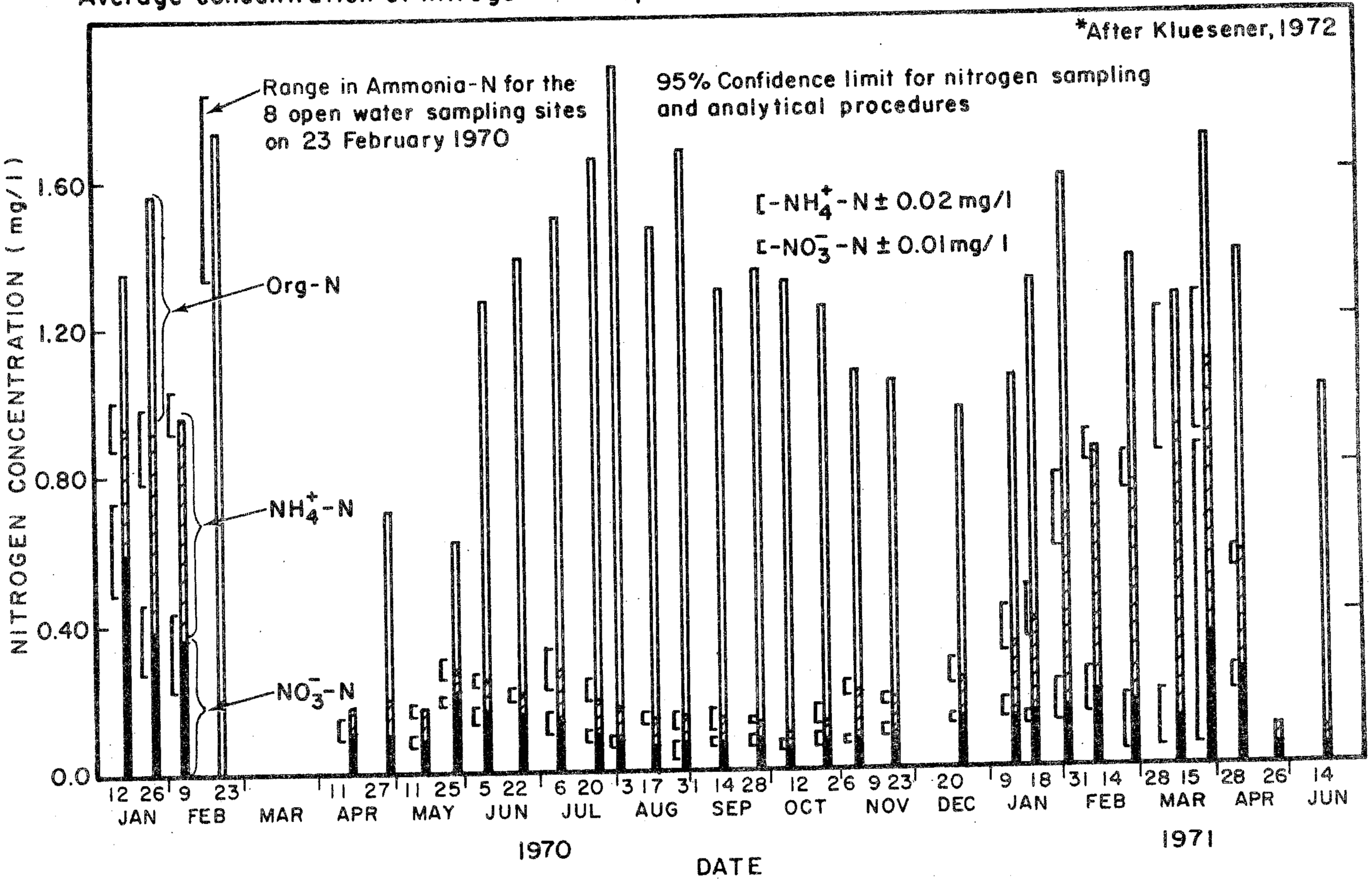


Figure 5

Average concentration of nitrogen in the open waters of Lake Wingra from January 1970 to June 1971*



Alkalinity was at a maximum during the winter months (see Figure 3), decreasing after ice-out as the temperature and biological activity increased. It decreased in a nearly linear fashion from the end of April to the end of September. It remained constant in October, and then increased to approximately 200 mg/l as $CaCO_3$ in March, 1971. The calcium concentration showed more variability during this same time period, but it followed the same general trend as the alkalinity (Kluesener, 1972).

BIOLOGICAL DESCRIPTION

ALGAE

The mean annual freshweight phytoplankton biomass in Lake Wingra was approximately 16 g/m^2 in 1970-71, with a growing season average of approximately 25 g/m^2 (Figure 6). The phytoplankton primary productivity was found to be about 2.4 g C/m^2/day (870 g C/m^2/yr), with a growing season average of about 4.6 g C/m^2/day. The phytoplankton class composition pattern for the same time period (Figure 7) shows the winter phytoplankton biomass to be dominated by the algae Cryptophyceae, while the spring season shows a dominance by the diatoms. They rapidly give way to the green algae. This is followed in the late summer and early fall by a dominance of the blue-green algae.

FISH

Early management practices (e.g., introduction of carp in the late 1880's, fish rescue and stocking operations during the 1930's, and carp removal programs during the 1930's, 1940's and 1950's) have had marked effects upon the fish populations in Lake Wingra. A total of 23 fish species have

Figure 6

Average daily productivity and freshweight biomass of phytoplankton in Lake Wingra.*

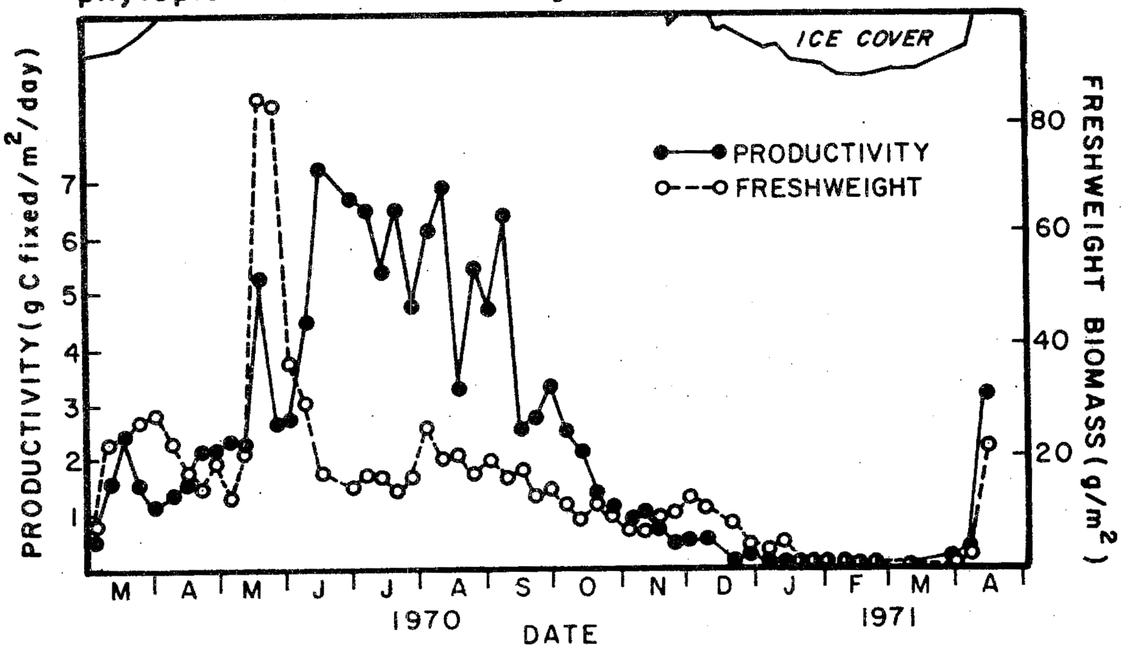
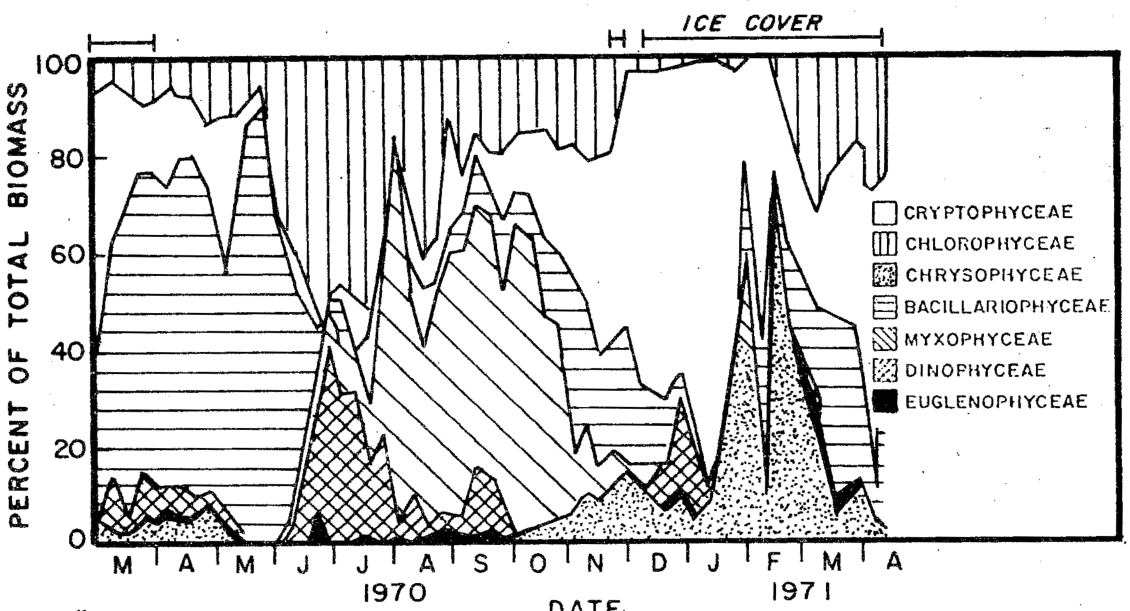


Figure 7

Class composition pattern for phytoplankton in Lake Wingra*



*After Huff et al., 1973

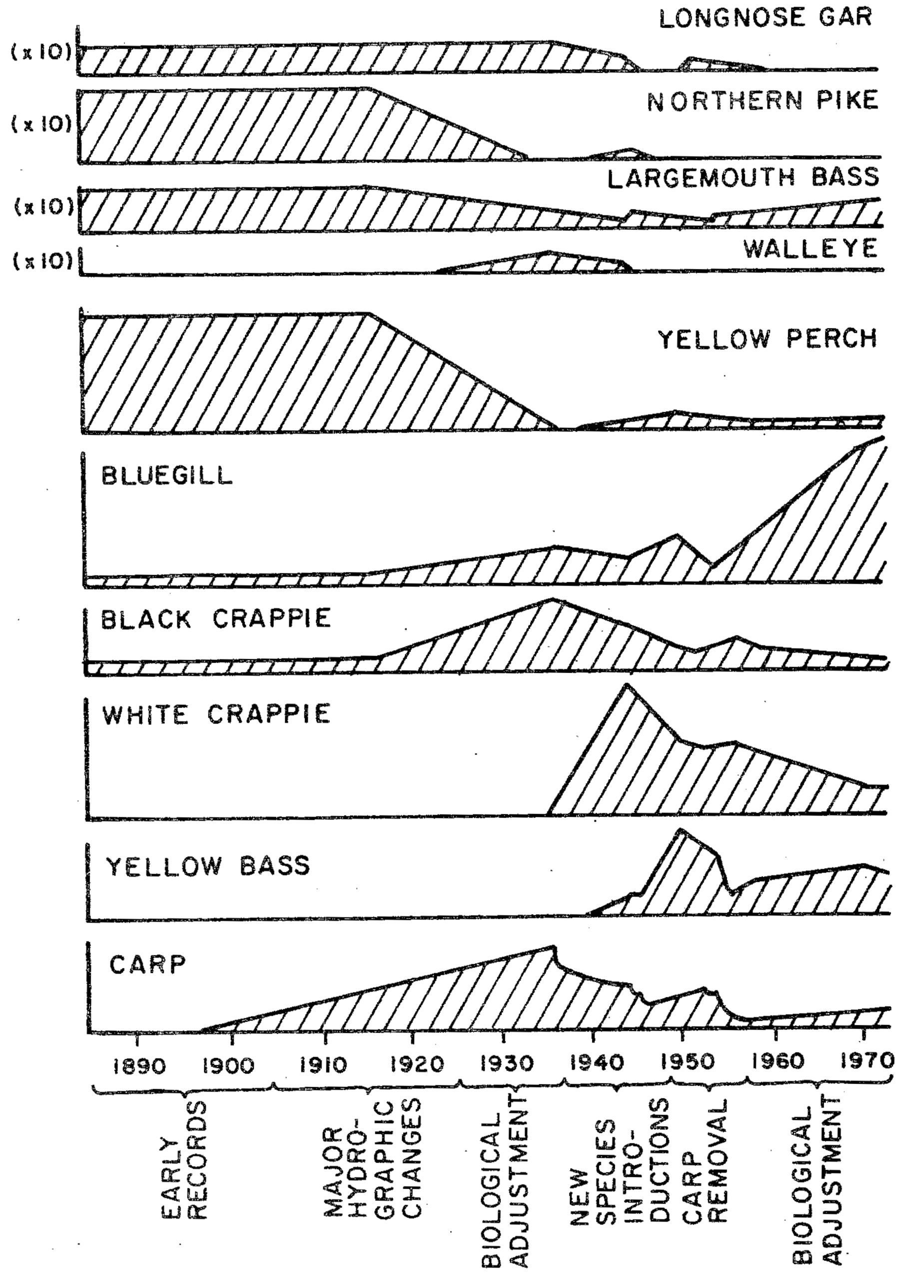
been introduced into Lake Wingra at one time or another in the past. Of these, two--the yellow bass and white crappie--have become abundant. The carp was introduced into Lake Wingra in the 1880's and became extremely abundant by the 1950's. Consequently, an intensive carp seining program was instituted by the Wisconsin Conservation Department between 1953 and 1955 because earlier efforts had not reduced the population to low enough levels (Neess *et al*,, 1957, cited in Hasler, 1963). Both largemouth bass and bluegill populations have recovered since the carp population became somewhat controlled (see Figure 8). The bluegill responded by becoming the dominant species in the lake. This increase, together with the establishment of white crappie and yellow bass, have produced the large, stunted panfish population which characterizes the present sport fishery (Baumann *et al*., 1974).

ZOOPLANKTON

Of the 48 cladoceran species present in Lake Wingra in 1891, only 23 are still present today (Table 5) (Baumann *et al*., 1974). Only four new species have been added to the list during this period. Sampling of the benthos during 1970-1972 has shown an invertebrate fauna dominated by small chironomids. No live mollusks or relatively large insects were found. The macroinvertebrate *Hyalella azteca* has virtually vanished. Intense fish predation on larger invertebrates may explain the rather recent decline of larger cladocerans and benthos in Lake Wingra.

Figure 8

Relative abundance of major fishes in Lake Wingra from 1890 to 1973, reconstructed from the literature*



*After Baumann et al, 1974

Table 5. CLADOCERAN SPECIES RECORDED FROM LAKE WINGRA BY BIRGE (1891) AND MORE RECENT STUDIES BY WHITE AND HASLER (1972) *

SPECIES	PRESENT 1891	PRESENT 1971
Acroperus harpae	X	X
Alona affinis	X	X
Alona costata	X	
Alona guttata	X	
Alona quadrangularis	X	X
Alona rectangula	X	
Alonella excisa	X	
Alonella exigua	X	
Bosimina longirostris	X	X
Camptocercus macrurus	X	X
Camptocercus rectirostris	X	X
Ceriodaphnia laticaudata	X	
Ceriodaphnia megalops	X	X
Ceriodaphnia pulchella	X	
Ceriodaphnia quadrangula		X
Ceriodaphnia reticulata	X	X
Chydorus globosus	X	
Chydorus ovalis		X
Chydorus sphaericus	X	X
Daphnia ambigua		X
Daphnia galeata	X	X
Daphnia pulex	X	
Daphnia retrocurva	X	X
Daphnia schodleri	X	
Diaphanosoma brachyurum	X	X
D. leuchtenbergianum	X	X
Drepanothrix dentata	X	
Dunhevia crassa	X	X
Eurycercus lamellatus	X	X
Graptoleberis testudinaria	X	
Holopedium gibberum	X	
Ilyocruptus sordidus	X	
Ilyocryptus spinifer	X	X

(continued)

Table 5 (continued)

SPECIES	PRESENT 1891	PRESENT 1971
Latona setifera	X	
Latonopis occidentalis	X	
Lathonuria rectrirostris	X	
Leptodora kindtii	X	X
Leydigia quadrangularis	X	X
Macrothrix laticornis	X	
Macrothrix rosea	X	X
Ophryoxus gracilis	X	
Oxyurella tenuicaudis	X	
Pleuroxus denticulatus	X	X
Pleuroxus procurvus		X
Pleuroxus striatus	X	
Pleuroxus trigonellus	X	
Polyphemus pediculus	X	
Sida crystallina	X	X
Scapholeberis aurita	X	
Scapholeberis kingi	X	X
Simocephalus serrulatus	X	X
Simocephalus vetulus	X	
TOTAL	48	27

*After Baumann *et al.*, 1974

MACROPHYTES

Around 1900, the shallow areas of Lake Wingra were dominated by cattails and bulrushes. Wild rice abounded in slightly deeper water, while submerged vegetation included water celery and pondweed. Dredging and filling in areas of the original lake (in 1900 Lake Wingra had a shallower maximum depth and covered about twice its present area) have produced water level fluctuations which have reduced the area available for littoral growth. In addition to hydrographic changes, from the 1920's to the mid-1950's the carp population essentially denuded Lake Wingra of macrophytes. After the carp removal program of the 1950's, vegetation of a different type returned to the lake. The Eurasian water milfoil, *Myriophyllum spicatum*, now dominates the macroflora of Lake Wingra.

Presently, five littoral communities occur in Lake Wingra. In shallow water, *Myriophyllum* constitutes 68 percent of the macroflora. In deeper waters, the communities are *Myriophyllum*, 13 percent; *Potamogeton-Myriophyllum*, 17 percent; *Nuphar*, 5 percent; and *Nymphaea*, 2 percent. Dominant emergents are *Typha latifolia*, *T. angustifolia* and *Scirpus validus*. The littoral zone covers approximately one-third of the lake's surface area and extends to 2.7 ± 0.4 meters. It is believed the littoral zone depth is dictated by light penetration, and that major reduction in turbidity would allow water milfoil stands to develop throughout the lake (Baumann *et al.*, 1974). Foraging by the carp population and resultant mixing of the muds are believed to have caused an increased turbidity and reduced light penetration. Their removal to less than 10 percent of their 1953 numbers by seining has been followed by an increased macrophyte population, indicating that control of the spread of macrophytes may have been one of their functions (Neess et al., 1955, cited in Kluesener, 1972).

NUTRIENT BUDGETS

Major potential nutrient sources for Lake Wingra have been shown to be precipitation, dry fallout, springflow, groundwater flow, urban runoff, surface runoff from the Arboretum and drainage from the marshes. The annual average nutrient loadings from precipitation, dry fallout, springflow and urban runoff are presented in Tables 6, 7, 8 and 9, respectively.

Groundwater loading into Lake Wingra could not be estimated by Kluesener because there was insufficient information available concerning the piezometric head and transmissibility of the soil in the lake vicinity. It is likely that any groundwater flow other than surface springs enters the lake through submerged springs, but these sources have not been identified. The marshes are believed to have input nutrient loads roughly equal to output nutrients.

The nutrient budget for Lake Wingra is summarized in Table 10. The most significant source of phosphorus to Lake Wingra is the urban runoff. More than 80 percent of the total phosphorus (980 kg/yr) and 90 percent of the dissolved reactive phosphorus (570 kg/yr) influent to Lake Wingra comes from this source (Kluesener, 1972). Very little dissolved reactive phosphorus enters the lake between storms (Huff et al., 1973). Precipitation, dry fallout and springflow contribute almost equally to the dissolved reactive phosphorus input (25, 21 and 30 kg/yr, respectively). Precipitation on the lake surface contributes less than 2 percent to the total phosphorus input. The groundwater phosphorus contributions to the lake are not known at present,

Table 6. AVERAGE ANNUAL LOADING OF NITROGEN AND PHOSPHORUS TO LAKE WINGRA FROM PRECIPITATION*

Loadings:	NH_4^+-N	NO_3^--N	Org-N	DRP	t-P
lbs/ac/in	0.089	0.10	0.059	0.0057	0.0073
kg/yr/lake	390	440	260	25	32

Volume of water to the lake/30.16 in of rain = 1 X $10^6 m^3$

*After Kluesener, 1972

Table 7. NUTRIENT LOADING OF LAKE WINGRA DUE TO DRY FALLOUT*

Period	NH_4^+-N	NO_3^--N	Org-N	DRP	t-P	Exposure Time Days
	kg/da	kg/da	kg/da	kg/da	kg/da	
Sep 26-Oct 4	0.76	0.32	1.40	0.027	0.10	7.8
Oct 15-Oct 24	1.85	1.25	1.41	0.050	0.18	9.9
Dec 10-Jan 1	0.62	1.35	1.22	0.045	0.060	29.0
Jan 1-Jan 31	2.08	2.08	0.93	0.042	0.23	22.0
Mar 6-Mar 14	1.31	1.44	---	0.018	0.060	8.3
Mar 28-Apr 4	1.06	0.36	5.4	0.09	0.59	8.5
Apr 26-May 4	2.30	1.47	2.02	0.10	0.16	7.5
May 12-May 18	2.50	0.59	9.5	0.12	0.93	5.6
Jun 6-Jun 14	1.50	0.76	1.8	0.03	0.35	8.5
Average load	1.60	1.30	3.0	0.06	0.30	-
Average Annual Lake Load (kg/yr)	565	475	1100	21	110	-

*After Kluesener, 1972

Table 8. AVERAGE ANNUAL LOADINGS FOR SPRINGS FLOWING INTO LAKE WINGRA*

Spring	Flow (cfs)	NH_4^+-N kg/yr	NO_3^--N kg/yr	DRP kg/yr	t-P kg/yr
Wingra	0.64	14.3	1690	8.5	14.0
Nakoma	0.30	44.0	700	5.5	19.7
Council Ring	0.06	0.4	46.5	0.3	0.4
East Spring	0.10	3.1	12.2	1.6	2.7
Duck Creek					
April-Dec 20	0.45	39.0	1100	8.4	18.6
Dec 20-Apr 1	--	70.0	480	6.2	22.0
Total Spring Input	1.55	170.8	4138.5	30.5	77.4

Total Lake Input = 1.37×10^6 m^3/yr

*After Kluesener, 1972

Table 9. SUMMARY OF NUTRIENT LOADING FROM URBAN RUNOFF AND PRECIPITATION (kg/storm)*

Date	Runoff Volume (ft^3)	Rain Amount (in)	NH_4^+-N	NO_3^--N	ORG-N	DRP	t-P	Suspended Solids	Volatile Solids
9/2/70	16,450		0.13	0.57	2.5	0.16	0.44	113	-
		0.36	0.15	0.10	0.25	0.002	0.012	-	-
9/3/70	8,380		0.10	0.24	0.61	0.17	0.22	28	-
		0.18	0.03	0.07	0.09	0.002	0.007	-	-
9/23/70	33,410		0.08	0.43	0.76	0.19	0.53	220	-
		0.74	--	--	--	--	--	-	-
9/23/70	175,600		0.89	1.57	3.57	0.63	2.55	2,215	-
		1.5	1.2	1.8	1.1	--	0.05	-	-
11/9/70	10,000		0.07	0.14	0.84	0.51	0.86	35	-
		0.30	--	--	--	--	--	-	-
3/14/71	25,000		0.24	0.30	4.0	0.08	0.91	890	247
		0.16	0.11	0.06	0.03	0.002	0.005	-	-
4/16/71	12,200		0.41	0.27	0.91	0.15	0.30	392	13
		0.33	0.42	0.25	0.96	0.006	0.01	-	-
5/4/71	8,320		0.14	0.17	0.07	0.07	0.15	34	10
		0.20	0.25	0.10	4.6	0.009	0.014	-	-
5/18/71	5,550		0.15	0.35	--	0.26	0.56	181	99
		0.11	--	--	4.8	--	--	-	-
5/23/71	23,100		0.23	0.35	--	0.23	0.82	153	58
		0.23	0.33	0.22	1.28	0.025	0.05	-	-
5/24/71	20,000		0.17	0.31	--	0.22	0.43	42	22
		0.30	0.28	0.18		0.011	0.021	2	1

*After Kluesener, 1972

Table 10. SUMMARY OF MEASURED NUTRIENT SOURCES FOR LAKE WINGRA*

Source	NH_4^+-N	NO_3^--N	Org-N (kg/yr)	DRP	t-P	Vol. of water(m^3)
Precipitation	390	440	260	25	32	1×10^6
Dry Fallout	560	480	1100	21	110	---
Springflow	170	4140	--	30	77	1.4×10^6
Urban Runoff	450	600	3500	570	980	1×10^6
Total =	1570	5660	4860	646	1199	

Lake Volume = 2.5×10^6 m^3

* After Kluesener, 1972

but it is not believed that this is a significant source (Kluesener, 1972).

The springflow contributes approximately 60 percent of the inorganic nitrogen (i.e., NH_4^+-N and NO_3^--N) influent to the lake. All other sources contribute approximately equal amounts of inorganic nitrogen. The urban runoff and springflow each contribute about 35 percent of the total nitrogen. Precipitation and dry fallout contribute about 10 percent and 20 percent respectively, to the total nitrogen loading. Thus, about 65 percent of the nitrogen budget of Lake Wingra comes from 'natural' sources. The resultant lake nitrogen concentration is thus more independent of storms than are the phosphorus concentrations, since springflow and groundwater provide significant nitrogen inputs to the lake between storms (Kluesener, 1972).

The volume of water contributed by rainfall, urban runoff and springflow is approximately 3.4 X 10^6 m^3/yr. The average nitrogen and phosphorus concentrations of this input are approximately 3.0 mg N/l and 0.3 mg P/l, respectively. This is about four times the average concentration of phosphorus and three times the average concentration of nitrogen (see Figures 4 and 5) normally found in Lake Wingra (0.07 mg total P/l and 1.1 mg total N/l, respectively). If it is assumed that Lake Wingra reacts like a large completely mixed body of water, as the data indicate , then nitrogen and phosphorus are being accumulated in the lake sediments or are being released to the atmosphere (Kluesner, 1972).

DISCUSSION

A summary of the available nutrient loading data and important physical features of Lake Wingra is presented in Table 11.

Huff et al.(1973) attempted to simulate urban runoff, nutrient loading and biotic response in Lake Wingra based on a Hydrologic Transport Model (HTM). The nutrients considered in their open lake model were dissolved inorganic phosphorus and dissolved inorganic nitrogen. They considered urban runoff, springflow, groundwater seepage, rainfall, dry fallout and internal sediment nutrient regeneration as nutrient sources.

Huff et al.(1973) assumed the available phosphorus form was dissolved inorganic phosphorus, and that this was a constant percentage of the total phosphorus entering the lake. However, Cowen (1973), studying the Madison area urban drainage, has shown that approximately 30 percent of the phosphorus in the particulate organic and inorganic forms will become available for algal growth in natural water systems. Therefore, the Huff et al. (1973) estimates of the

Table 11. SUMMARY OF AVAILABLE NUTRIENT LOADING DATA AND PHYSICAL CHARACTERISTICS*

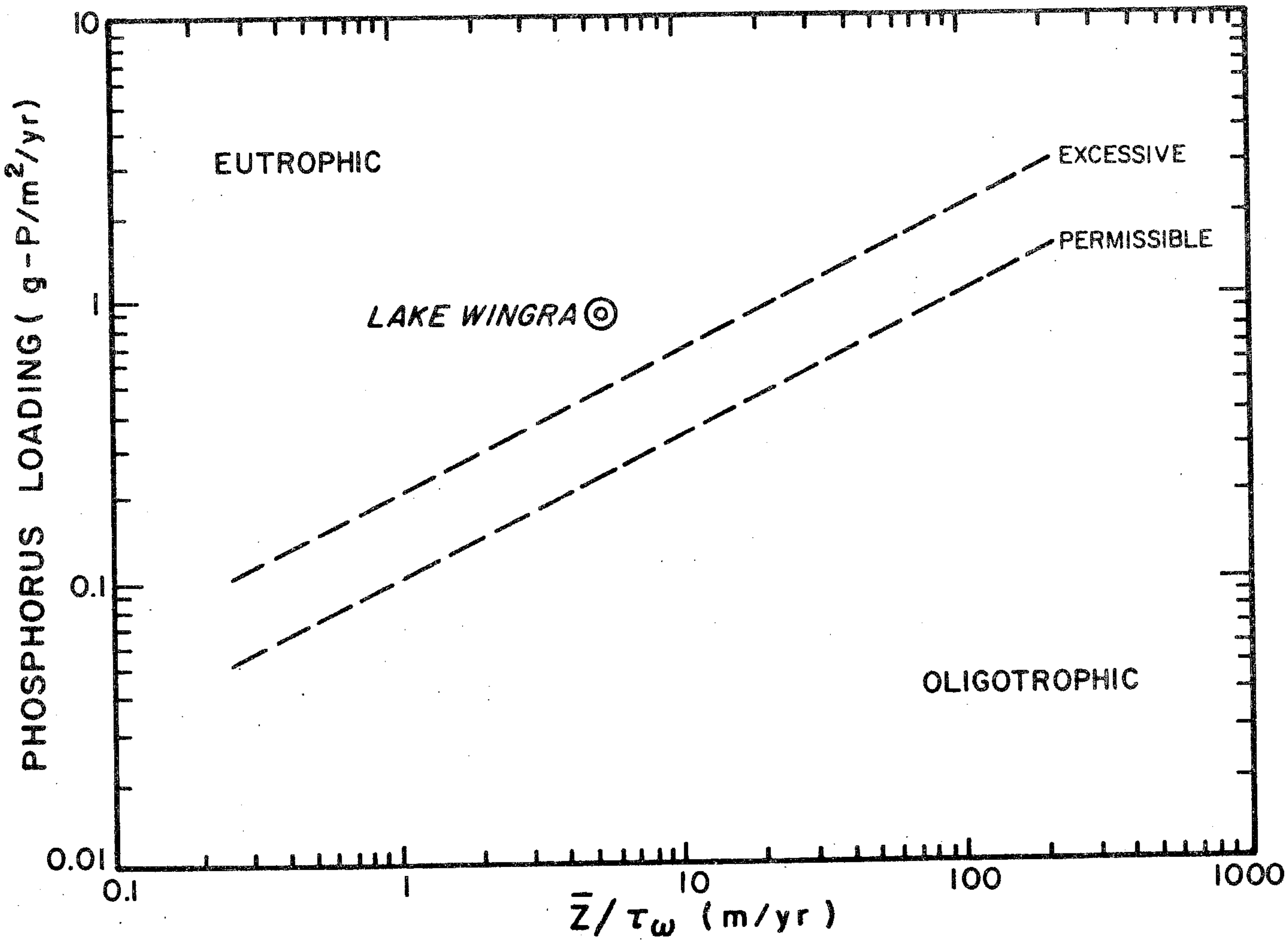
I. Nutrient Loadings		
Total Phosphorus (t-P)	1199 kg/yr	0.88 $g/m^2/yr$
Dissolved Reactive Phosphorus (DRP)	646 kg/yr	
Total Nitrogen (NH_4^+-N, NO_3^--N & Org-N)	12090 kg/yr	8.83 $g/m^2/yr$
Inorganic Nitrogen (NH_4^+-N & NO_3^--N)	7230 kg/yr	
II. Biomass & Productivity		
Phytoplankton Biomass, Annual Average		16 g/m^2
Phytoplankton Biomass, Growing Season Average		25 g/m^2
Phytoplankton Primary Productivity Annual Average (870 g $C/m^2/yr$)		2.4 g $C/m^2/day$
Phytoplankton Primary Productivity, Growing Season Average		4.6 g $C/m^2/day$
III. Physical Characteristics of Lake Wingra		
Maximum Depth		6.10 m
Mean Depth		2.42 m
Surface Area, excluding lagoons & ponds		$1.37 \times 10^6 m^2$
Total Volume, excluding lagoons & ponds		$2.50 \times 10^6 m^3$
Annual Input		$5.70 \times 10^6 m^3$
Hydraulic Residence Time		0.44 yr
Mean Depth/Hydraulic Residence Time		5.5 m/yr
Mean Secchi Depth		1.3 m
IV. Chemical Characteristics of Lake Wingra		
Mean Alkalinity		153 mg/l as $CaCo_3$
Mean Calcium Concentration		34 mg/l
Mean Conductivity		Not Determined
Mean Annual DO		1 m - 7.7 mg/l 3 m - 6.6 mg/l
pH		Min.-7.7 Max.-9.4

*After Huff et al., 1973 and Kluesener, 1972 available phosphorus input to Lake Wingra are expected to be low due to the fact that Kluesener (1972) found that a substantial part of the phosphorus entering the lake is not in the immediately available form. Further, it would be expected that the marshes through which much of the urban drainage enters the lake would significantly alter the transport rate of available phosphorus to the lake, making it essentially impossible, with the information available today, to develop meaningful models which relate nutrient transport in the urban areas of the Lake Wingra watershed to algae and macrophyte growth in the lake.

The Vollenweider loading curve (Vollenweider, 1975; Vollenweider and Dillon, 1974) values, based on data in Table 11, are 0.88 $g/m^2/yr$ for phosphorus loading and 5.5 m/yr for the mean depth/hydraulic residence time. When these values are plotted according to Vollenweider (1975), Lake Wingra falls in a category typical of lakes, with similar phosphorus loadings and morphometric and hydrological characteristics, which are considered eutrophic (Figure 9). The current phosphorus loading is about four times its "permissible" loading rate for its mean depth and hydraulic residence time characteristics.

The trophic status is in agreement with the physical, chemical and biological characteristics of the lake. Lake Wingra is a shallow lake with shallow sloping shoreline, in which the thermocline is absent. The hypolimnion volume is low or absent, and it has low transparency. The entire water column can be affected by wind-generated mixing. This situation tends to promote increased nutrient cycling and therefore a higher degree of eutrophication than for deeper lakes with similar nutrient loads. Blue-green algae are usually the dominant forms during the summer months. The fish present in Lake Wingra are abundant in number, but mostly trash species

Figure 9
Vollenweider loading relationship



such as carp (Neess et al., 1957, cited in Hasler, 1963; Huff et al., 1973).

IMPROVEMENT OF WATER QUALITY IN LAKE WINGRA

The overall water quality of Lake Wingra could be improved somewhat with the cooperation of the residents of the portion of the southwest corner of Madison, Wisconsin, which lies within the Lake Wingra drainage basin. The primary water quality problem in Lake Wingra and the other Madison lakes is an excessive amount of alga and waterweeds (particularly *Myrillophyllum* in Lake Wingra) caused by excessive inputs of aquatic plant nutrients such as nitrogen and phosphorus compounds. Studies by Walton and Lee (1972) and Fitzgerald and Lee (1971) have shown the key factor governing the excessive algal and waterweed growth in Lake Wingra is the concentration of phosphorus in the water.

As stated earlier, approximately one-third of the total drainage to Lake Wingra comes from the University of Wisconsin Arboretum, while the remaining two-thirds is urban runoff from southwest Madison. More significantly, the urban runoff delivers approximately 80 percent of the total phosphorus and 90 percent of the soluble orthophosphate (see Table 10) to Lake Wingra. Consequently, the amounts of nutrients entering Lake Wingra by way of urban runoff could be substantially reduced by reduction of the phosphorus content of the urban runoff. To achieve this will require that each individual living in the Lake Wingra drainage basin conduct his activities in such a manner as to reduce, and wherever possible, eliminate the transport of phosphorus to Lake Wingra. Particular attention should be given to improving the efficiency and frequency of street cleaning in the urban parts of the Lake Wingra watershed. Lee (1972) has discussed in detail various methods that can be used by the residents of Madison to improve the water quality of Lake Wingra and the other Madison lakes. Elimination of the phosphorus input from urban runoff would lower the annual phosphorus loading to 0.23 g/m^2/yr. This reduced loading would place Lake Wingra in the oligotrophic trophic category, according to the Vollenweider criteria. Realistically, of course, this will not be achieved. However, with even a moderate reduction of phosphorus input, Lake Wingra could likely become a less productive body of water and achieve a significant reduction in the frequency and severity of excessive algal blooms and macrophyte growth.

ACKNOWLEDGEMENTS

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REPORT ON NUTRIENT LOAD - EUTROPHICATION RESPONSE

OF SELECTED SOUTH-CENTRAL WISCONSIN IMPOUNDMENTS

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INTRODUCTION

To meet an increasing demand for lakeside property in Wisconsin, private developers constructed a number of recreational impoundments during the 1960's and early 1970's. In addition, state and local governmental agencies developed impoundments to provide water recreation for the public. This report assesses the relative water quality in ten impoundments in central and southern Wisconsin through the development of a trophic state index. The nutrient loadings to each of the impoundments are estimated and a comparison made between estimated nutrient load and trophic status for these impoundments. This is accomplished by applying a phosphorus loading relationship developed by Vollenweider (1973).

IMPOUNDMENTS STUDIED

Rickert and Spieker (1971) have defined real estate lakes as bodies of water created in an urban environment for the enhancement of real estate value. Wisconsin has experienced a large number of impoundments of this type. This study includes lakes created to facilitate high density lake front development. It also includes public recreation lakes which represent the other broad classification of impoundments investigated in this study. These are lakes created for various recreational uses of the public and are free of significant urban development around the lake. The impoundments investigated in this study can be divided into these two classifications as shown in Table 1.

Table 1. GENERAL CLASSIFICATION OF IMPOUNDMENTS IN THIS STUDY

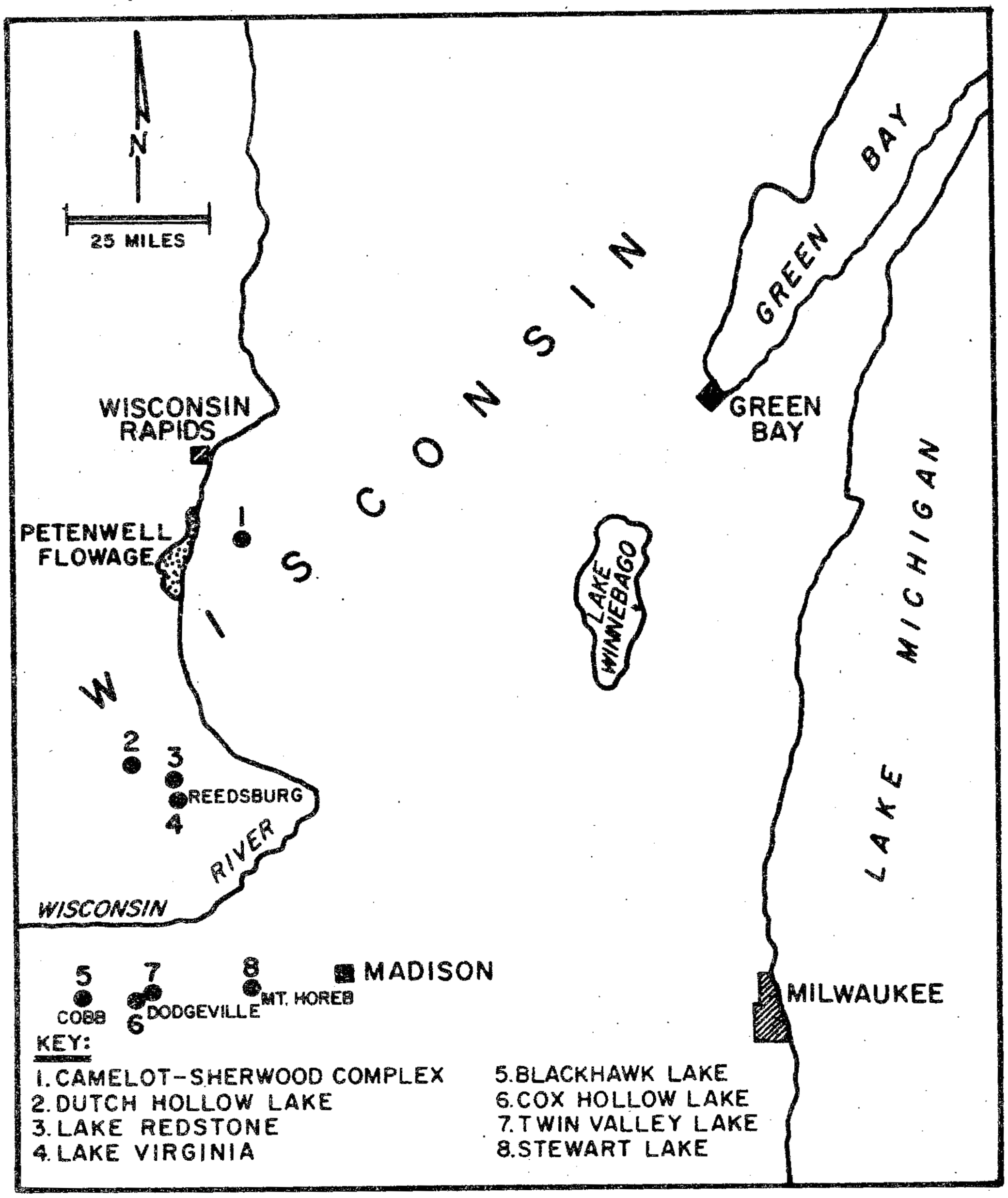
Real Estate Lakes	Public Recreation Lakes
Lake Redstone	Blackhawk Lake
Lake Virginia	Stewart Lake
Dutch Hollow Lake	Cox Hollow Lake
Lake Camelot North	Twin Valley Lake
Lake Camelot South	
Lake Sherwood	

GEOGRAPHIC AND HYDROLOGIC DESCRIPTION

The impoundments described in this report are all located in central and southern Wisconsin (see Figure 1). The Camelot-Sherwood complex is located in central Wisconsin about ten miles south of Wisconsin Rapids. These three impoundments are located on Spring Branch and Fourteen-Mile Creeks which drain marshy areas of the central sand plains. Lakes Redstone, Virginia and Dutch Hollow, also located in the central part of the state, are in Sauk County near Reedsburg. Lakes Redstone and Dutch Hollow were formed in dammed valleys of Big and Dutch Hollow Creeks, respectively. Lake Virginia is a seepage lake, relying predominantly on groundwater to maintain the water level.

Three of the impoundments are located in Iowa County in the southwestern part of the state. Blackhawk Lake is located north of the town of Cobb. Two dams on adjacent valleys resulted in a horseshoe-shaped lake. Two inlet streams (total average flow is about 4.4 cfs) provide water to the lake. Cox Hollow and Twin Valley Lakes, in Governor Dodge State Park north of Dodgeville, are interconnected by a stream

Figure 1
Impoundment Locations in Central and Southern Wisconsin



approximately one mile long. Effluent waters from Cox Hollow Lake flow into Twin Valley Lake. Cox Hollow Lake was formed by placing an earthen dike at the junction of two valleys, creating a horseshoe-shaped lake. The impoundment is equipped with several artificial circulation devices to maintain dissolved oxygen in the hypolimnion during periods of stratification. Twin Valley Lake receives about one-half of its normal water flow from Cox Hollow Lake. Three other small streams contribute to the 4 cfs average inflow. The dam structure of the impoundment is designed for withdrawal of bottom waters from the impoundment.

Stewart Lake is located in southwestern Dane County near Mt. Horeb. The lake is fed by a small inlet stream and several artesian springs.

The surface area, mean depth and hydraulic residence times of the impoundments are summarized in Table 2. All the impoundments are quite shallow, with surface areas ranging from 25,000 to 2.8 million square meters. Hydraulic residence times range from about one month to nearly three years.

CLIMATE

All of these impoundments are influenced by similar climatological conditions. Annual average temperatures range from 40 to 50°F. In January, average temperatures are 15-20°F, while in July the average temperature is about 70-75°F. An ice cover forms on the impoundments in early to mid-December and generally persists into early April.

Annual precipitation averages near 30 inches with much of the precipitation falling from April through September. Average annual snowfall ranges from 35-50 inches, with the larger accumulations occurring further north in the state. Based on studies of other lakes in this region of the state (Cline, 1965), the evapotranspiration is probably about equal to the precipitation.

Table 2. HYDROLOGICAL CHARACTERISTICS OF THE IMPOUNDMENTS*

Impoundment	Surface Area, $m^2 \times 10^5$	Mean Depth, m	Hydraulic Residence Times, Years
Camelot-Sherwood	28.3	2.9	0.09 - 0.14
Redstone	25.3	4.3	0.7 - 1
Dutch Hollow	8.5	6	1.8
Virginia	1.8	1.7	0.6 - 1.9
Blackhawk	8.9	4.9	0.5
Cox Hollow	3.9	3.8	0.5 - 0.7
Twin Valley	6.1	3.8	0.4 - 0.5
Stewart	0.25	1.9	0.08

* Information in this table was obtained from the lake developers or from Wisconsin DNR files (1972-73) and previously appeared in Piwoni and Lee (1974). Certain mean depth and hydraulic residence time values have been subsequently revised. All data are for projected normal pool elevations.

GEOLOGY

The Camelot-Sherwood basin is set into an area of unconsolidated morainal deposits composed of glacial till and gravel and sand outwash (Weeks and Strangland, 1971). These deposits are underlain by Cambrian sandstone over Precambrian crystalline rock. The unconsolidated deposits are the major source of water to the region.

Similar geology likely extends to the Reedsburg area (Lakes Virginia, Redstone and Dutch Hollow), although no specific information is available.

The Twin Valley-Cox Hollow basin contains a number of water-bearing geologic formations (Klingelhoets, 1962). These include Galena dolomite, Platteville limestone and St. Peter sandstone, all yielding small amounts of high mineral content water. Soils in the region are composed of silty loam and loamy alluvial materials with high organic content. Much of the area surrounding the impoundments is quite steep and rocky.

The geology of the Blackhawk Lake watershed is probably quite similar to Cox Hollow and Twin Valley watersheds. The watershed is located in the driftless region of the state and consists of narrow ridges with steep, narrow V-shaped valleys (Bredemus, 1970). The soil regime is composed of silty loam and stony undeveloped soils.

The Stewart Lake watershed consists of deposits of Trenton and Galena limestones and St. Peter sandstone (WDNR, 1972-73). The soils in the region are predominantly silty loam and sandy loam with some stony land.

WATERSHED CHARACTERISTICS

All of the impoundments are in rural, predominantly agricultural watersheds (except for Stewart Lake, which is located below a small city and receives much of the runoff from the city streets). The real estate lakes will likely undergo changes in nutrient loadings from the watershed as development of waterfront homes (with septic tank systems) proceeds.

Table 3 presents the areas of the watershed of each impoundment. The range of watershed size is from 510 acres for Stewart Lake to over 22,000 acres for the Camelot-Sherwood Complex.

Table 3. AREA OF IMPOUNDMENT WATERSHEDS*

Impoundment	Acres	Hectares
Camelot-Sherwood	22,400	9,060
Redstone	18,940	7,660
Dutch Hollow	3,100	1,250
Virginia	1,600	650
Blackhawk	8,960	3,630
Cox Hollow	3,970	1,610
Twin Valley	7,680	3,110
Stewart	510	210

* Information in this table was compiled from lake developer data, data from the Wisconsin DNR files (1972-73) and from USGS topographic maps. This information is based on Piwoni and Lee (1974).

BIOLOGICAL DESCRIPTION

FISHERIES

Limited information was available on the fisheries of several of the impoundments. Attempts have been made to manage the fisheries in nearly all of the impoundments. Consequently, stocking of various game fishes has taken place over the years. However, the fisheries in nearly all of the impoundments are predominantly composed of panfish such as the bluegill (Lepomis macrochirus) and sunfish (Lepomis spp.),plus a number of rough bottom feeding fish. Fishing time,and consequently fish yields, has generally dropped in recent years, presumably because of an overabundance of small panfish varieties instead of more favored gamefish.

Lake Redstone has been stocked several times with walleye (Stizostedion vitreum) to supplement existing fish populations (Smith, 1973). The lake also contains apparently declining populations of largemouth bass (Micropterus salmoides) and northern pike (Esox lucius). Walleye and panfish appeared to be stunted in growth probably due to reduced living space indirectly caused by reduced DO levels in the hypolimnion during winter and summer.

Reports by Dunst (1969) and Wirth et al. (1970) indicate that similar fish species inhabit Cox Hollow and Twin Valley Lakes. Largemouth bass and northern pike were stocked in Cox Hollow Lake in 1958 (Dunst, 1969), but populations generally decreased after 1962. Records on these two fishes in Twin Valley Lake (Wirth et al., 1970) indicated both species suffered from a 60-70 percent mortality rate. Smaller sized bluegills were beginning to dominate the fish populations.

Rainbow trout have been stocked in Stewart Lake and seem to grow well although reproduction information was not available. Annual opening day trout fishing is quite heavy on this impoundment (with reportedly good results)

AQUATIC PLANTS

Very little information was available on specific macrophyte populations in these impoundments. All of the impoundments suffered from excessive macrophyte growth in littoral areas. Impoundments with steep banks, such as Redstone, Blackhawk, and Twin Valley, did not have the problems prevalent in impoundments with large littoral regions such as the Camelot-Sherwood complex. The latter was treated annually with herbicide to control aquatic weeds.

Dunst (1969) reported that Ceratophyllum demersum had become the dominant macrophyte in Cox Hollow Lake. That plant, as well as Myriophyllum, Potamogeton and Limna spp., was observed by the authors in the Camelot-Sherwood complex. Stewart Lake supported a variety of aquatic vegetation in the shallow waters near the point of inflow of the creek, including cattails, water lilies, wild rice and sedges. Macrophyte problems were not critical in any of the other impoundments sampled during a two year period from June, 1971 to April, 1973 (Piwoni and Lee, 1974).

Blue-green algae obtained dominance in all of the impoundments at least on two occasions during the summers of 1971 and 1972 (Piwoni and Lee, 1974). *Anacystis* spp. attained dominance at least once in all the impoundments except Lake Virginia. *Aphanizomenon*, *Coelastrum* and *Anabaena* spp. were other predominant blue-greens. In Lake Virginia, *Scenedesmus* was the dominant algal genus on several sampling dates. During spring and fall, algal populations in the impoundments were dominated by diatoms, such as *Fragilaria* and *Asterionella*, and flagellates, such as *Trachelamonas* and *Chlamydomonas*. The amount of algae is reflected in chlorophyll *a* concentrations presented later in this paper.

TROPHIC INDEX PARAMETERS AND ANALYSIS METHODS

To assess the water quality of the impoundments, a trophic state index (TSI) was developed. The approach employed is similar to that used by Lueschow *et al*. (1970) in their evaluation of Wisconsin lakes. The index parameters were chosen because it was felt that they would present a relative indication of water quality in the impoundments. While the overall approach is approximately the same, there are important differences between the formulation and use of the different parameters in this evaluation and in the trophic state index used by Lueschow *et al*. (1970) in their studies.

The trophic state index parameters used in this study are presented in Table 4. Secchi depth measurements were used as a measure of turbidity and light penetration (Ruttner, 1965). Chlorophyll *a*, an estimate of the phytoplankton biomass, was determined using the method described by Strickland and Parsons (1965). The percentage of the lake volume containing less than

Table 4. TROPHIC STATE INDEX PARAMETERS

1. Secchi Depth	- Mean of all values obtained.
2. Chlorophyll *a*	- Average concentration in first 2 meters of water column during study period.
3. DO Depletion	- Percent of lake volume with less than 0.5 mg-DO/l; May to October, inclusive.
4. Orthophosphate - winter	- Average in-lake concentration during winter under ice.
5. Orthophosphate - summer	- Average epilimnion concentration; May to October, inclusive.
6. Total phosphorus - winter	- Average in-lake concentration during winter under ice.
7. Total phosphorus - summer	- Average epilimnion concentration; May to October, inclusive.
8. Organic Nitrogen	- Average concentration in first 2 meters of water column during study period.

0.5 mg/l of dissolved oxygen gave an indication of the aquatic plant material that accumulated in the hypolimnion and exerted an oxygen demand. DO was determined using a YSI Model 54 Dissolved Oxygen Meter. All phosphorus determinations were made using the ascorbic acid method described in Standard Methods, 13th edition (APHA *et al.*, 1971). Total phosphorus determinations were made on unfiltered, autoclaved samples which were treated with persulfate. Ammonium and Kjeldahl-nitrogen analyses were automated using a Technicon AutoAnalyzer and the phenate method as described in Standard Methods (APHA *et al.*, 1971). Organic-N was calculated as Kjeldahl-N minus NH_4^+-N. All nitrogen and phosphorus values are reported as mg-N/l and mg-P/l. The atomic ratio of inorganic-N to soluble ortho-P was in excess of 16 to 1 in all the impoundments except Lake Virginia and, therefore, nitrogen apparently was not limiting algal growth. Consequently, inorganic-N was not included as a trophic state index parameter since the algal growth in these impoundments was probably not dependent on the inorganic-N concentration.

Samples used for analysis were collected over a two-year period at approximately six week intervals. Samples were collected at either one or two meter intervals in the deepest part of the impoundments. Volumn-weighted mean lake concentrations of the parameters in Table 4, excluding Secchi depth and DO depletion, were calculated for each sampling date.

The mean values for the index parameters for each of the impoundments were then determined. These values are presented in Table 5, along with the trophic ranking received by the impoundments for each parameter. This ranking was based on a relative scale in which each impoundment was assigned an integer value from 1 to 10 dependent on the relative magnitude of each of its water quality-TSI parameters. For example, Blackhawk Lake had the highest average Secchi depth. It was ranked number 1 for this parameter. Dutch Hollow Lake had the lowest average Secchi depth and was ranked number 10. The sum of these individual parameter rankings yielded an overall trophic state index value for each lake. Inorganic-N was not used to compute this sum. These values provided the basis for the water quality ranking of the impoundments given in Table 6. The complete data obtained in this investigation have been reported by Piwoni and Lee (1974).

Lakes Camelot North and South and Sherwood received the highest water quality rankings (see Table 6). These lakes were arbitrarily designated as moderately eutrophic based on the TSI value and general water quality characteristics. The reasons for the relatively high water quality in these lakes is probably because of the low in-lake phosphorus levels and the highly-colored nature of the water. The latter can limit algal growth by limiting light penetration (Lee, 1972).

Table 5. DETERMINATION OF TROPHIC STATE INDEX PARAMETERS

Lake	Secchi Depth (meters)	Chloro-phyll a (μg/l)	DO Deple-tion (%)	Winter Ortho-P (mg-P/l)	Summer Ortho-P (mg-P/l)	Winter TIN (mg-N/l)	Summer TIN (mg-N/l)	Organic Nitrogen (mg-N/l)	Winter Total-P (mg-P/l)	Summer Total-P (mg-P/l)
Redstone	1.6(4)	12.8(5)	23.2(10)	0.008(4)	0.008(2)	0.80(5)	0.31(4)	0.82(4)	0.03(4)	0.11(8)
Virginia	1.2(9)	29.0(9)	5.6(5)	0.004(1)	0.025(10)	0.22(1)	0.18(1)	1.31(10)	0.02(1)	0.15(10)
Dutch Hollow	0.8(10)	33.9(10)	8.0(7)	0.021(8)	0.013(7)	0.61(4)	0.22(2)	0.97(6)	0.40(10)	0.12(9)
Camelot North	2.6(2)	5.8(1)	0.5(1)	0.007(2)	0.005(1)	0.49(2)	0.48(6)	0.66(2)	0.02(1)	0.04(1)
Camelot South	1.8(3)	5.8(1)	4.8(4)	0.010(5)	0.010(6)	1.41(9)	0.80(9)	0.65(1)	0.04(5)	0.04(1)
Sherwood	1.6(4)	7.2(3)	3.2(3)	0.007(2)	0.008(2)	1.32(8)	0.48(6)	0.66(2)	0.02(1)	0.04(1)
Blackhawk	3.6(1)	14.6(6)	7.3(6)	0.044(10)	0.015(8)	1.02(7)	0.54(8)	1.26(9)	0.12(9)	0.05(4)
Stewart	1.4(8)	12.3(4)	8.6(8)	0.011(6)	0.008(2)	2.26(10)	0.86(10)	0.96(5)	0.04(5)	0.08(7)
Cox Hollow	1.5(6)	26.5(8)	2.0(2)	0.036(9)	0.015(8)	0.83(6)	0.36(5)	1.13(8)	0.10(8)	0.06(5)
Twin Valley	1.5(6)	19.0(7)	9.2(9)	0.019(7)	0.009(5)	0.51(3)	0.23(3)	1.06(7)	0.07(7)	0.06(5)

* Values in parentheses after parameter values indicate trophic ranking. The highest water quality received the lowest number.

Table 6. WATER QUALITY RANKING OF IMPOUNDMENTS BASED ON TROPHIC STATE INDEX

Relative Degree of Eutrophication	Rank	Lake	TSI*
Moderate	1	Camelot North	11
	2	Sherwood	18
	3	Camelot South	26
	4	Redstone	41
	5	Stewart	45
	6(tie)	Blackhawk	53
	6	Twin Valley	53
	8	Cox Hollow	54
	9	Virginia	55
High	10	Dutch Hollow	67

* Trophic State Index Value. This value is the sum of the individual parameter relative rankings in Table 5.

Lakes Redstone through Virginia have similar water quality problems and TSI values. Dutch Hollow Lake, which had severe algal and turbidity problems throughout most of the study, was designated as highly eutrophic.

Algal and macrophyte growth produces some aesthetic problems in all these impoundments and can hamper establishment of game fisheries. Most of the impoundments, which have occasional excessive algal blooms and macrophyte growth provide water-related recreational activities for large numbers of people and, therefore, are important recreational assets to the area.

Water quality in most of these impoundments should be quite stable, except perhaps in the newest lakes, Blackhawk and Dutch Hollow, where water quality is likely to improve with time (Frey, 1963). In the real estate lakes, which will experience continued development over the next 20-50 years (Carlson, 1971), it is possible that some deterioration of water quality could result as nutrients from septic tank effluents enter the lake. The significance of this potential source of nutrients would have to be evaluated in light of the total nutrient loadings to the lakes and the amount of phosphorus that enters the lakes from this source.

NUTRIENT LOADINGS

Nutrient loadings for all the impoundments were estimated from land use in the watershed using the values Sonzogni and Lee (1974) developed for the Lake Mendota (Wisconsin) watershed. These values are presented in Table 7. The only variation in the table values applied in the study was for total-P from rural runoff. The range of 0.34 to 0.45 kg/ha/yr was used because these values correlated most closely with a nutrient input study on Cox Hollow Lake performed by Dunst et al. (1972). Information on lake and watershed characteristics was obtained from the Wisconsin Department of Natural Resources and lake development personnel. The loadings, calculated as kg/year and g/m^2 of surface area, are presented for each impoundment in Table 8. Ranges reflect the range used for contribution of P from rural lands. Available information on the watershed required grouping Lake Camelot North and South and Sherwood. Groundwater was not included in the estimates of nutrient loadings because of the difficulties of evaluating the relative importance of this potential nutrient source. Groundwater would be expected to supply considerable amounts of nitrogen, and possibly some phosphorus, particularly in sandy soil regions. The omission of the groundwater component is thought to be of minor importance based on studies conducted by Lee (1972).

Vollenweider (1973) had developed a logarithmic plot relating phosphorus loading to mean depth/hydraulic residence time. This graph also contains straight-line definitions of "permissible" and "excessive" phosphorus loading limits relative

Table 7. AMOUNTS OF NITROGEN AND PHOSPHORUS DERIVED FROM VARIOUS TYPES OF LAND USE DANE COUNTY, WISCONSIN*

	Amounts Contributed (kg/hectare/yr)			
Activity	Inorganic-N	Organic-N	Soluble O-P	Total-P
Base Flow	1.2	--	0.11	0.11
Woodland	0	0	0	0
Rural Runoff	3.1	2.0	0.34	0.67** 0.22***
Urban Runoff	1.1	3.9	0.67	1.1
Manured lands 100 cows/sq.mi. 15 tons manure/year	--	3.4	--	1.1
Precipitation	6.0	1.9	0.18	0.22
Dry Fallout	7.5	8.1	0.11	0.78
		(kg/capita/yr)		
Domestic Waste-waters	2.7	0.9	1.4	2.0
Septic Tanks	var.	var.	var.	var.
Groundwater	var.	var.	var.	var.
Drained marshes	101 kg/hectare		45 kg/hectare	

* After Sonzogni and Lee (1974)

** Wisconsin

*** Other Areas

Table 8. NUTRIENT LOADINGS TO THE IMPOUNDMENTS*

Impoundment	Nitrogen Loading kg/year	Nitrogen Loading g/m^2	Phosphorus Loading kg/year	Phosphorus Loading g/m^2
Redstone	45,400	18.1	3630-4230	1.44-1.68
Dutch Hollow	8,800	10.4	810- 870	0.95-1.02
Virginia	3,300	18.3	210- 270	1.15-1.48
Camelot-Sherwood	97,600	34.6	6660-7580	2.35-2.68
Blackhawk	20,900	23.4	1900-2070	2.11-2.32
Stewart	1,850	73.6	120- 200	4.82-8.05
Cox Hollow	7,410	19.1	630- 810	1.62-2.08
Twin Valley	10,500	17.4	1090-1250	1.74-2.05

* Date are revised from earlier values presented by Piwoni and Lee (1974). Loadings are presented as total kg/year and as grams per square meter of lake surface area.

to depth-flushing characteristics of the lake. The necessary calculations were made to determine values of mean depth/hydraulic residence times for each of the study impoundments. Annual total phosphorus loadings, in g/m^2, were then plotted against the mean depth/hydraulic residence time values, in m/yr, on a reproduction of the Vollenweider plot (Vollenweider, 1973) (Figure 2).

All of the impoundments fell into the region of the graph defined by Vollenweider (1973) to be eutrophic. The degree of eutrophication was interpreted as the "distance" a specific lake was above the "permissible" loading level for a lake with the same mean depth/hydraulic residence time value. Table 9 presents the estimated phosphorus loading and the "permissible" loading level, as defined by Vollenweider (1973), for each of

Figure 2
Wisconsin Impoundments on Vollenweider Phosphorus Loading Plot.

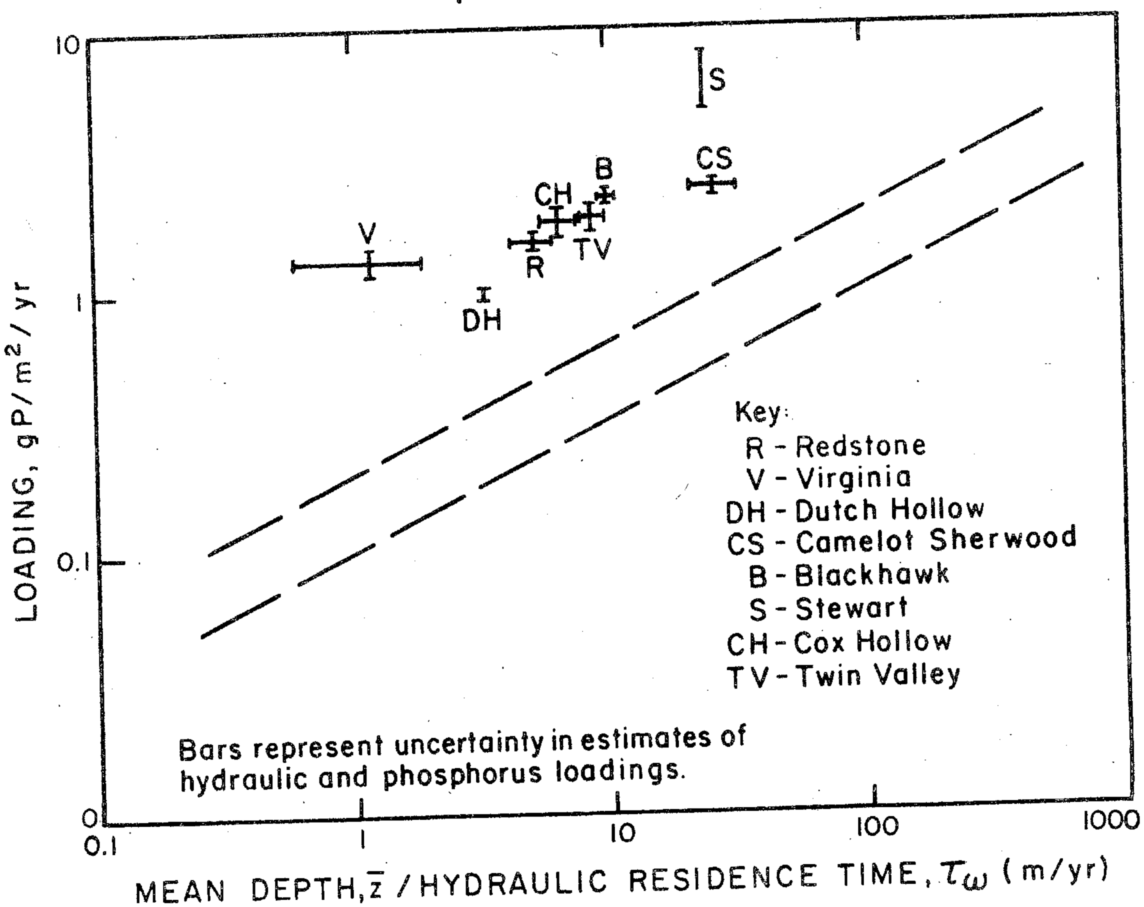


Table 9. PHOSPHORUS LOADING RELATIONSHIPS

Rank[a]	Lake	TSI Ranking[b]	Estimated Total Loading $g/m^2/yr$[c]	"Permissible" Loading $g/m^2/yr$[d]	Estimated Loading / Permissible Loading Ratio
1-3	Camelot-Sherwood complex	1-3	2.35 - 2.68	0.47 - 0.58	4.0 - 5.7
4	Dutch Hollow (at pool elevation)	10[e]	0.95 - 1.02	0.18	5.1 - 5.5
5	Twin Valley	6	1.74 - 2.05	0.28 - 0.32	5.5 - 7.2
6	Redstone	4	1.44 - 1.68	0.21 - 0.26	5.6 - 8.0
7	Blackhawk	6	2.13 - 2.32	0.32	6.7 - 7.2
8	Cox Hollow	8	1.62 - 2.08	0.24 - 0.28	5.7 - 8.7
9	Stewart	5	4.82 - 8.05	0.50	9.6 - 16.1
10	Virginia	9	1.15 - 1.48	0.08 - 0.14	8.2 - 18.7

[a]Rank based on the midpoint of the phosphorus loading ratio range.

[b]TSI ranking from Table 6.

[c]Estimated from land use in the watershed.

[d]Loading values according to Vollenweider (1973) for lakes with that hydraulic residence time.

[e]TSI value for Dutch Hollow was based on data collected while the lake was filling.

of the impoundments. It also gives the ratio of estimated loading to "permissible" phosphorus loading. This ratio should be an indication of the degree of eutrophication of each impoundment. The impoundments were arranged in Table 9 in order of ascending mid-range ratio values, i.e., in decreasing lake water quality based on this loading ratio.

Dutch Hollow Lake was filling throughout the study period; this is reflected in the ranking in Tables 6 and 9 and indicates the lake water quality would improve when the lake was filled. Blackhawk was also filling throughout the first eight months of the study; however, normal pool elevation was used in all calculations presented here.

Comparison of the water quality ranking in Table 6 with that in Table 9 shows reasonably good correlation particularly at the ends of the ranking scale. Several of the impoundments have exchanged positions in the order in Table 9, but these lakes generally have overlapping phosphorus loading ratio ranges. Part of the problem for a lack of correlation between estimated total phosphorus loads and the overall water quality characteristics may be due to a number of factors. One of these is that in some instances a substantial part of the total phosphorus, such as the particulate forms, entering the impoundment may not become available to aquatic plants in the impoundment. Studies by Cowen (1973) show that only about 30 percent of the particulate phosphorus in both urban and rural drainage will likely become available for algal growth in lakes.

It appears that either approach to assessing relative water quality in lakes and impoundments is viable, and together they may provide an approach to lake water quality assessment and management.

A number of hydrologic and water quality parameters for each lake are summarized in Tables A-1 to A-8 in the Appendix.

ACKNOWLEDGEMENTS

Most of the information in this report was taken from Piwoni and Lee (1974) Report to the Wisconsin Department of Natural Resources. That report was assembled as part of a Master's thesis study at the University of Wisconsin-Madison. Copies of the report may be obtained by writing the authors. Presentation of much of the background information on these impoundments would not have been possible without the assistance of the Wisconsin Department of Natural Resources personnel, especially T. Wirth and R. Dunst, and several of the lake developers. Special thanks also go to J. Stroud for his assistance throughout the project.

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TABLE A-1 DATA SUMMARY FOR NORTH AMERICAN PROJECT
LAKE REDSTONE (WISCONSIN)

Trophic State	- Eutrophic
Lake Type	- Impoundment
Drainage Area	- 7.67×10^7 square meters
Lake Surface Area	- 2.52×10^6 square meters
Mean Depth	- 4.3 meters
Retention Time	- 0.7 - 1 years
Mean Alkalinity	- 125 mg/l as $CaCO_3$
Mean Conductivity	- 260 μmhos/cm @ 25°C
Mean Secchi Disk	- 1.6 meters
Mean Dissolved Phosphorus	- 0.008[a,b] mg/l as P
Mean Total Phosphorus	- 0.03[a] 0.11[b] mg/l as P
Mean Inorganic Nitrogen	- 0.80[a] 0.31[b] mg/l as N
Mean Chlorophyll a	- 12.8[c] μg/l
Annual Productivity	- --
Phosphorus Loading	
Point source	- 0 kg/year
Non-point source	- 3630 - 4230 kg/year
Surface area loading	- 1.44 - 1.68 gr/meter²/year
Nitrogen Loading	
Point source	- 0 kg/year
Non-point source	- 45,400 kg/year
Surface area loading	- 18.1 gr/meter²/yr

[a]Average winter
[b]Average summer epilimnion
[c]In first two meters of water column
--Not determined

TABLE A-2 DATA SUMMARY FOR NORTH AMERICAN PROJECT
DUTCH HOLLOW LAKE (WISCONSIN)

Trophic State	- Eutrophic
Lake Type	- Impoundment
Drainage Area	- 1.25×10^7 square meters
Lake Surface Area	- 8.50×10^5 square meters
Mean Depth	- 3 m(study level)- 6 m(when filled)
Retention Time	- 1.8 years
Mean Alkalinity	- 133 mg/l as $CaCO_3$
Mean Conductivity	- 252 μm hos/cm @ 25°C
Mean Secchi Disk	- 0.8 meters
Mean Dissolved Phosphorus	- 0.021[a] 0.013[b] mg/l as P
Mean Total Phosphorus	- 0.40[a] 0.12[b] mg/l as P
Mean Inorganic Nitrogen	- 0.61[a] 0.22[b] mg/l as N
Mean Chlorophyll *a*	- 33.9[c] μg/l
Annual Productivity	- --
Phosphorus Loading	
Point source	- 0 kg/year
Non-point source	- 810 - 870 kg/year
Surface area loading	- 0.95 - 1.01 gr/meter^2/year
Nitrogen Loading	
Point source	- 0 kg/year
Non-point source	- 8,800 kg/year
Surface area loading	- 10.4 gr/meter^2/year

[a]Average winter

[b]Average summer epilimnion

[c]In first two meters of water column

--Not determined

TABLE A-3 DATA SUMMARY FOR NORTH AMERICAN PROJECT
LAKE VIRGINIA (WISCONSIN)

Trophic State	- Eutrophic
Lake Type	- Impoundment
Drainage Area	- 6.48×10^6 square meters
Lake Surface Area	- 1.82×10^5 square meters
Mean Depth	- 1.7 meters
Retention Time	- 0.9 - 2.8 years
Mean Alkalinity	- 64 mg/l as $CaCO_3$
Mean Conductivity	- 230 μmhos/cm at 25°C
Mean Secchi Disk	- 1.2 meters
Mean Dissolved Phosphorus	- 0.004[a] 0.025[b] mg/l as P
Mean Total Phosphorus	- 0.02[a] 0.15[b] mg/l as P
Mean Inorganic Nitrogen	- 0.22[a] 0.18[b] mg/l as N
Mean Chlorophyll <u>a</u>	- 29.0[c] μg/l
Annual Productivity	- --
Phosphorus Loading	
Point source	- 0 kg/year
Non-point source	- 210 - 270 kg/year
Surface area loading	- 1.15 - 1.48 gr/meter^2/year
Nitrogen Loading	
Point source	- 0 kg/year
Non-point source	- 3,300 kg/year
Surface area loading	- 18.3 gr/meter^2/year

[a]Average winter

[b]Average summer epilimnion

[c]In first two meters of water column

--Not determined

TABLE A-4 DATA SUMMARY FOR NORTH AMERICAN PROJECT
CAMELOT-SHERWOOD COMPLEX (WISCONSIN)

Trophic State	- Mesotrophic - Eutrophic
Lake Type	- Impoundment
Drainage Area	- 9.06×10^7 square meters
Lake Surface Area	- 2.83×10^6 square meters
Mean Depth	- 2.9 meters
Retention Time	- 0.09 - 0.14 years
Mean Alkalinity	- 125 mg/l as $CaCO_3$
Mean Conductivity	- 311 µmhos/cm at 25°C
Mean Secchi Disk	- 2.0 meters
Mean Dissolved Phosphorus	- 0.008[a] 0.008[b] mg/l as P
Mean Total Phosphorus	- 0.03[a] 0.04[b] mg/l as P
Mean Inorganic Nitrogen	- 1.07[a] 0.59[b] mg/l as N
Mean Chlorophyll a	- 6.3[c] µg/l
Annual Productivity	- --
Phosphorus Loading	
Point source	- 0 kg/year
Non-point source	- 6660 - 7580 kg/year
Surface area loading	- 2.35 - 2.68 gr/meter^2/year
Nitrogen Loading	
Point source	- 0 kg/year
Non-point source	- 97,600 kg/year
Surface area loading	- 34.6 gr/meter^2/year

[a]Average winter

[b]Average summer epilimnion

[c]In first two meters of water column

--Not determined

TABLE A-5 DATA SUMMARY FOR NORTH AMERICAN PROJECT LAKE BLACKHAWK (WISCONSIN)

Trophic State	- Eutrophic
Lake Type	- Impoundment
Drainage Area	- 3.63×10^7 square meters
Lake Surface Area	- 8.90×10^5 square meters
Mean Depth	- 4.9 meters
Retention Time	- 0.5 years
Mean Alkalinity	- 227 mg/l as $CaCO_3$
Mean Conductivity	- 471 μmhos/cm at 25°C
Mean Secchi Disk	- 3.6 meters
Mean Dissolved Phosphorus	- 0.044[a] 0.015[b] mg/l as P
Mean Total Phosphorus	- 0.12[a] 0.05[b] mg/l as P
Mean Inorganic Nitrogen	- 1.02[a] 0.54[b] mg/l as N
Mean Chlorophyll a	- 14.6[c] μg/l
Annual Productivity	- --
Phosphorus Loading	
Point source	- 0 kg/year
Non-point source	- 1900 - 2070 kg/year
Surface area loading	- 2.13 - 2.32 gr/meter^2/year
Nitrogen Loading	
Point source	- 0 kg/year
Non-point source	- 20,900 kg/year
Surface area loading	- 23.4 gr/meter^2/year

[a]Average winter

[b]Average summer epilimnion

[c]In first two meters of water column

--Not determined

TABLE A-6 DATA SUMMARY FOR NORTH AMERICAN PROJECT
LAKE STEWART (WISCONSIN)

Trophic State	- Eutrophic
Lake Type	- Impoundment
Drainage Area	- 2.07×10^6 square meters
Lake Surface Area	- 2.51×10^4 square meters
Mean Depth	- 1.9 meters
Retention Time	- 0.08 years
Mean Alkalinity	- 213 mg/l as $CaCO_3$
Mean Conductivity	- 540 μmhos/cm @ 25°C
Mean Secchi Disk	- 1.4 meters
Mean Dissolved Phosphorus	- 0.011[a] 0.008[b] mg/l as P
Mean Total Phosphorus	- 0.04[a] 0.08[b] mg/l as P
Mean Inorganic Nitrogen	- 2.26[a] 0.86[b] mg/l as N
Mean Chlorophyll a	- 12.3[c] ug/l
Annual Productivity	- --
Phosphorus Loading	
Point source	- 0 kg/year
Non-point source	- 121 - 202 kg/year
Surface area loading	- 4.82 - 8.05 gr/meter^2/year
Nitrogen Loading	
Point source	- 0 kg/year
Non-point source	- 1,850 kg/year
Surface area loading	- 73.6 gr/meter^2/year

[a]Average winter

[b]Average summer epilimnion

[c]In first two meters of water column

--Not determined

TABLE A-7 DATA SUMMARY FOR NORTH AMERICAN PROJECT
COX HOLLOW LAKE (WISCONSIN)

Trophic State	- Eutrophic
Lake Type	- Impoundment
Drainage Area	- 1.61×10^7 square meters
Lake Surface Area	- 3.88×10^5 square meters
Mean Depth	- 3.8 meters
Retention Time	- 0.5 - 0.7 years
Mean Alkalinity	- 205 mg/l as $CaCO_3$
Mean Conductivity	- 440 μmhos/cm @ 25°C
Mean Secchi Disk	- 1.5 meters
Mean Dissolved Phosphorus	- 0.036[a] 0.015[b] mg/l as P
Mean Total Phosphorus	- 0.10[a] 0.06[b] mg/l as P
Mean Inorganic Nitrogen	- 0.83[a] 0.36[b] mg/l as N
Mean Chlorophyll a	- 26.5[c] μg/l
Annual Productivity	- --
Phosphorus Loading	
Point source	- 0 kg/year
Non-point source	- 630 - 810 kg/year
Surface area loading	- 1.62 - 2.08 gr/meter^2/year
Nitrogen Loading	
Point source	- 0 kg/year
Non-point source	- 7,410 kg/year
Surface area loading	- 19.1 gr/meter^2/year

[a]Average winter

[b]Average summer epilimnion

[c]In first two meters of water column

--Not determined

TABLE A-8 DATA SUMMARY FOR NORTH AMERICAN PROJECT
TWIN VALLEY LAKE (WISCONSIN)

Trophic State	- Eutrophic
Lake Type	- Impoundment
Drainage Area	- 3.11×10^7 square meters*
Lake Surface Area	- 6.07×10^5 square meters
Mean Depth	- 3.8 meters
Retention Time	- 0.4 - 0.5 years
Mean Alkalinity	- 175 mg/l as $CaCO_3$
Mean Conductivity	- 370 μmhos/cm @ 25°C
Mean Secchi Disk	- 1.5 meters
Mean Dissolved Phosphorus	- 0.019[a] 0.009[b] mg/l as P
Mean Total Phosphorus	- 0.07[a] 0.06[b] mg/l as P
Mean Inorganic Nitrogen	- 0.51[a] 0.23[b] mg/l as N
Mean Chlorophyll a	- 19.0[c] μg/l
Annual Productivity	- --
Phosphorus Loading	
Point source	- 0 kg/year
Non-point source	- 1090 - 1250 kg/year
Surface area loading	- 1.74 - 2.05 gr/meter^2/year
Nitrogen Loading	
Point source	- 0 kg/year
Non-point source	- 10,500 kg/year
Surface area loading	- 17.4 gr/meter^2/year

* About 1/2 of drainage area is controlled by an upstream impoundment.

[a]Average winter

[b]Average summer epilimnion

[c]In first two meters of water column

--Not determined

SECTION VIII - MULTIPLE-STATE LAKES AND SPECIAL TOPICS

LIMNOLOGICAL CHARACTERISTICS OF THE POTOMAC ESTUARY

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INTRODUCTION

Increasingly, over the past few centuries, the water of the Potomac Estuary has been degraded, primarily by the domestic wastewater discharged from the Washington, D.C. metropolitan area. High coliform counts, low dissolved oxygen levels, and nuisance algal growths typify water quality management problems in the Potomac Estuary.

Since the early 1900's, numerous water quality studies have been conducted on the Potomac River Basin including the Estuary. Initial studies primarily emphasized bacterial quality and dissolved oxygen problems. Beginning in the late 1960's, studies were expanded to include the problem of eutrophication. A report on the "Water Resources/Water Supply" by Jaworski, et al. (1971) culminated over six years of intensive investigation of the Upper Potomac Estuary.

The Potomac Estuary was included in the North American Project, a eutrophication study by the Organization of Economic Cooperation and Development (OECD). This report summarizes history and recent data relative to the goals of the North American Project of OECD.

DESCRIPTION OF THE POTOMAC RIVER BASIN

The Potomac River Basin, including the Estuary, comprises the second largest watershed in the Middle Atlantic States, with a drainage area of approximately 38,000 square kilometers (km^2). From its headwaters on the eastern slope of the Appalachian Mountains, the Potomac flows first northeasterly and then generally southeasterly some 644 km, flowing past the nation's capital. The Potomac is tidal from Washington, D. C. to its confluence with the Chesapeake Bay, a distance of 183 km (Figure 1).

Of the 3.3 million people living in the basin, about 2.8 million live in the Washington, D. C. metropolitan area. The upper basin is largely rural with scattered small cities populated by 10,000 to 20,000. Land use in the entire Potomac Basin is estimated to be 5 percent urban, 55 percent forest, and 40 percent agriculture, including pasture lands.

Climate Study Area

The Potomac River Tidal System lies in a sort of climatic crossroads. Cold air masses invade from Canada and the Arctic, while the Appalachian Mountains provide some protection from the cold.

Hurricanes moving north along the Atlantic Seaboard generally pass over the lower tidal system about once every five years. Coastal "northeastern" storms often bring strong winds accompanied by heavy rain or snow from that direction, most frequently in winter and early spring.

Annual precipitation ranges from 89 to 114 centimeters (cm), including about 61 cm of snow. Table 1 shows that precipitation is fairly well distributed throughout the year.

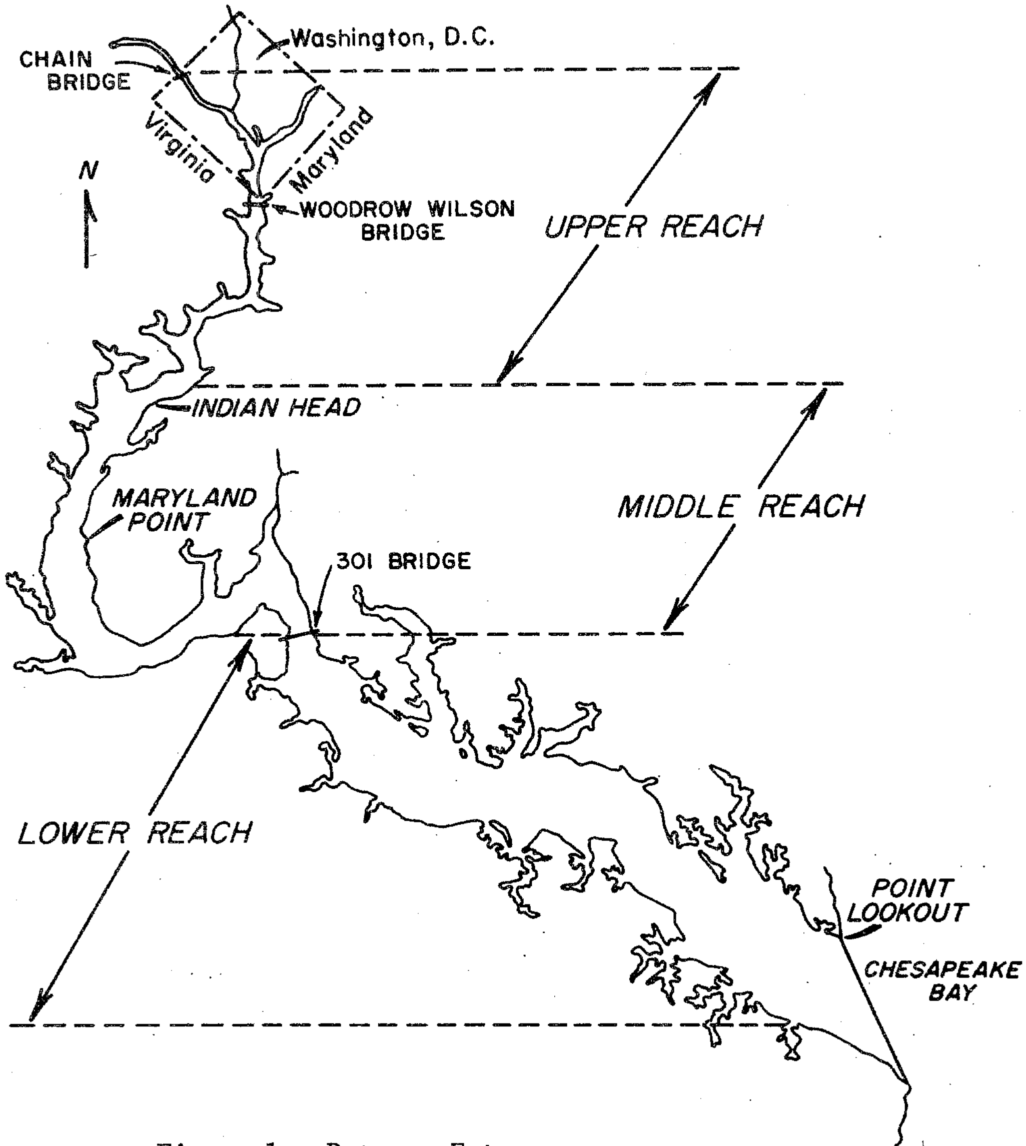

Figure 1. Potomac Estuary.

Winters in the Potomac River Basin are moderately cold and the summers warm, indicated by the mean monthly temperatures also in Table 1. Daytime temperatures of more than 35 C. are not unusual in summer. The frost-free season averages about 150 days.

Description of Potomac Estuary

For discussion and investigative purposes, the tidal portion of the Potomac River was divided into three reaches shown in Figure 1 and described below:

Reach	Description	River Kilometer	Volume ($m^3 \times 10^7$)
Upper	From Chain Bridge to Indian Head	183 to 135	26.4
Middle	From Indian Head to Rt. 301 Bridge	135 to 75.0	102.5
Lower	From Rt. 301 Bridge to Chesapeake Bay	75.0 to 00.0	496.5

The upper reach, although tidal, contains fresh water. The middle reach normally is the transition zone from fresh to brackish water. In the lower reach, chloride concentrations near the Chesapeake Bay range from about 9,000 to 15,000 mg/l.

The tidal portion, about 60 meters in width at its head and at Washington, broadens to nearly 10 km at its mouth. A shipping channel with a minimum depth of 7.5 meters is maintained upstream to Washington. Except for this channel and a few short reaches where depths reach up to 30 meters, the tidal portion is relatively shallow, averaging about 5.5 meters in depth.

The mean tidal range is about 0.9 meter in the upper portion near Washington and about 0.5 meter near Chesapeake Bay. The tidal lag time between Washington and Chesapeake Bay is about 6.5 hours. The latitude and longitude of the centroid of the Potomac Estuary are 38° 22' 150" and 77° 00' 300", respectively. The altitude is at the level of the Atlantic Ocean.

TABLE 1

MEAN MONTHLY TEMPERATURE AND PRECIPITATION
FOR
WASHINGTON, D. C., NATIONAL AIRPORT
1933 - 1972

Month	Monthly Temperature (°C)	Monthly Precipitation (cm)
January	2.1	6.55
February	3.1	6.83
March	7.4	8.36
April	13.4	7.34
May	18.8	9.88
June	23.4	9.22
July	25.7	10.52
August	24.8	12.12
September	21.2	7.80
October	15.2	7.24
November	8.9	7.90
December	3.3	7.72

WATER RESOURCE USES OF THE POTOMAC ESTUARY

Municipal Water Supply Use

The municipal water supply of the Washington metropolitan area comes from five major sources, primarily the Potomac River above Washington, D. C. During 1969-1970, the five sources provided 1.4×10^6 m^3/day. Currently, no municipal water is drawn from the freshwater portion of the Potomac Estuary; however, an emergency estuary intake was considered during the drought in the summer of 1969.

Industrial Use

In the Washington metropolitan area, an insignificant amount of water is used for manufacturing, primarily as cooling water in stream electric plants.

Currently six major consumers in the Potomac River tidal system use 10.4×10^6 m^3/day of cooling water. A seventh user has been proposed.

Recreation and Boating

Aside from enhancing the suburban environment, the water and land resources of the Potomac Estuary and its tributaries improve the aesthetics of the capital. From Washington with its many tourists to the remote park at Point Lookout near Chesapeake Bay, the Potomac's resources are widely used. These include freshwater and tidal sport fishing, boating, hunting, swimming, camping, and picnicking.

Commercial Fisheries

The Potomac Estuary supports a substantial commercial fishery. Approximately 160 fish species live in the Potomac Estuary ecosystem. The most significant economically are the anadromous and the semi-anadromous species such as striped bass, shad, white and yellow perch, winter flounder, and herring.

Another group of commercially important fish species spawn and winter outside of Chesapeake Bay in the Atlantic Ocean, using the Potomac for a nursery area and feeding ground. This group includes the menhaden, croaker, silver perch, sea trout, and drum.

The lower reaches of the Potomac Estuary are considered prime shellfish waters. There oysters and soft clams are indigenous, occurring in the same general areas. Only in recent years, however, have they been harvested commercially, and the demand far exceeds the resource.

The lower Potomac affords a favorable habitat for blue crabs. As juveniles, the young crabs feed and grow in the Estuary before completing their life cycle at the mouth of Chesapeake Bay.

WASTEWATER LOADINGS AND TRENDS

Approximately 1.4 million m^3/day of municipal wastewater are discharged into the upper reach of the Potomac River tidal system. Currently, 18 waste treatment facilities serve approximately 2.8 million people in the Washington metropolitan area.

That current discharge is a nine-fold increase over the 0.16 million m^3/day in 1913. Similarly, total nitrogen and phosphorus loads have increased about 10-fold and 22-fold, respectively (see Table 2).

TABLE 2

WASTEWATER LOADING TRENDS
(AFTER TREATMENT)

Year	Waste Flow (m^3/day)	5-day BOD (kg/day)	Total Nitrogen (kg/day)	Total Phosphorus (kg/day)
1913	160,000	26,300	2,900	500
1932	283,000	46,700	5,200	1,000
1944	632,000	63,900	10,400	2,000
1954	738,000	90,600	14,400	2,500
1960	840,000	49,800	16,800	4,500
1970	1,400,000	63,900	27,200	10,900

MORPHOMETRY AND HYDROLOGY

Morphometry

The basic morphometric data for the three reaches of the Potomac Estuary are tabulated:

	Upper	Middle	Lower
Length (km)	48	60	75
Avg. depth (m)	4.8	5.1	7.2
Avg. width (m)	1100	3625	9740
Surface area ($m^2 \times 10^6$)	57.4	211.6	695.2
Volume ($m^3 \times 10^7$)	26.4	102.5	496.5

Because of tidal action and low salinity, the upper reach is unstratified. Stratification begins in the middle reach during summer conditions. In the lower reach, stratification occurs mostly during summer conditions.

Hydrology

The upper Potomac River Basin is the major source of freshwater inflow into the Estuary. From 1930-1968, the average flow at Great Falls was 305 m^3/sec before diversions for municipal water supply.

The mean monthly flows of the Potomac at Great Falls are tabulated below for the reference period of 1931-1960.

	Mean Monthly Flow (m^3/sec)		Mean Monthly Flow (m^3/sec)
January	215	July	100
February	245	August	75
March	395	September	55
April	360	October	55
May	245	November	85
June	170	December	110

Each year the Potomac River delivers about 2,300 million kilograms of sand, silt, clay, and organic debris to the tidal system. Most of this usually occurs during February or March with maximum monthly loads ranging from 50 to 90 percent of the total annual load.

Tides dominate the currents in the Estuary. Typical maximum tidal velocities for the three reaches are:

Reach	Velocity (cm/sec)
Upper	25
Middle	28
Lower	18

The hydraulic detention time for any given reach of the tidal system depends on the rate of fresh water inflow. The water renewal time for the three reaches are given for the 5, 50, and 95% flow conditions:

Flow (m^3/sec)	Percent of time flow exceeded	Hydraulic Detention Time (years) Upper	Middle	Lower	Total
40	95%	0.21	0.81	3.95	4.07
185	50%	0.045	0.175	0.854	1.07
1150	5%	0.0073	0.028	0.137	0.17

The above tabulation indicates that the upper reach has relatively short detention times, while the lower reach has times similar to lakes.

LIMNOLOGICAL CHARACTERIZATION

Physical

Even though it is 189 kilometers long, the Potomac Estuary maintains a rather homogeneous temperature. While some stratification occurs in the lower reach, tidal action appears to keep the system fairly well mixed.

The mean monthly water temperatures in the upper Estuary recorded for 22 years are:

Month	°C	Month	°C
January	2.5	July	28.1
February	3.3	August	27.8
March	7.8	September	24.7
April	14.0	October	18.4
May	20.4	November	11.5
June	25.9	December	4.8

The light penetration measured by the Secchi disk varies considerably in the Potomac Estuary:

Reach	Ranges of Secchi Disk (meters)
Upper	0.4 to 0.8
Middle	0.5 to 1.3
Lower	1.0 to 2.3

The turbid upper reach has a rather low transparency due to particulate matter originating in the upper river basin. Suspended solids in wastewater discharges also hinder light penetration. During the summer months, pronounced algal growths in the lower part of the upper reach sometimes limit Secchi disk depths to less than 0.15 meter.

The conductivity in the Potomac Estuary is related to the salinity concentration. The average ranges of conductivity and salinity for the three reaches of the Estuary are:

Reach	Conductivity µmhos at 25°C	Salinity 0/00
Upper	200 to 500	0.06 to 0.16
Middle	600 to 17000	0.22 to 9.00
Lower	17000 to 26000	9.00 to 15.00

In the lower reach, significant stratification is due to a two-layer flow of water---that is, an upper layer with a net seaward movement and a lower layer with a net upstream movement. The seaward flow of fresh water results in less salinity in the lower layer than in the upper layer of water.

Chemical

The Potomac River tidal system appears well buffered chemically. Average ranges for pH and alkalinity are:

Reach	pH (Units)	Total Alkalinity (mg/1)
Upper	7.0 to 8.0	70 to 110
Middle	7.5 to 8.2	60 to 85
Lower	7.5 to 8.0	65 to 85

The well buffered inflows from the upper Potomac River Basin and wastewater discharges maintain the Estuary in narrow ranges of pH and alkalinity. Tidal action keeps the system fairly well mixed.

Dissolved oxygen concentrations in the upper Potomac Estuary have been routinely monitored since 1935. In the Upper Estuary near the wastewater outfalls, those concentrations have followed a continuous downward trend since 1938, slightly enhanced in 1960. Measurements for the point of least concentration in the Upper Estuary illustrate this downward trend:

Year	Min. Consecutive 28 Day Average (mg/1)	Min. Single Value During Period (mg/1)
1940	4.0	3.0
1948	3.5	2.5
1950	3.0	2.3
1955	2.5	1.0
1960	3.5	2.5
1965	3.0	2.0
1970	2.5	2.0

These low concentrations occurred mainly during the warm temperature months in a zone extending about 10 kilometers from the wastewater outfalls.

In the middle reach of the Estuary, dissolved oxygen decreased significantly only during periods of massive algal blooms. Tidal action kept this region well mixed.

In the lower Estuary, low dissolved oxygen levels are common in the summer months. Concentrations less than 2.0 mg/l occur in the deeper reaches because high biological turnover with thermal and salinity stratification restricts reaeration.

To date, trace elements in the water of the Potomac Estuary have not been comprehensively analyzed. A recent study by Jaworski, et al. (1971) on heavy metals in the Estuary sediments has caused concern about the accumulation of metals and resulting water quality problems. The study included analyses for lead, cobalt, chromium, cadmium, copper, nickel, zinc, silver, barium, aluminum, iron, and lithium.

The concentration of nutrients along the Estuary varies as a function of wastewater loading, temperature, freshwater inflow from the upper basin, biological activity, and salinity. Jaworski, et al. (1971, 1972) have reported the annual distribution of the various nutrient concentrations. Table 3 summarizes the summer levels for six key stations along the Estuary.

In the vicinity of the Woodrow Wilson Bridge, the increase in total and inorganic phosphorus, $NO_2 + NO_3$, ammonia, and total Kjeldahl nitrogen can be attributed to the 1.40 million m^3/day of wastewater discharged from the Washington metropolitan area. Between Woodrow Wilson Bridge and Indian Head, ammonia nitrogen rapidly disappears as nitrifying

TABLE 3

AVERAGE RANGE OF CONCENTRATION
SUMMER CONDITIONS
Upper Potomac Estuary

Station and Kilometers from Chain Bridge	Total Phosphorus (mg/1)	Inorganic Phosphorus (mg/1)	$NO_2 + NO_3$ Nitrogen (mg/1)	NH_3 Nitrogen (mg/1)	TKN Nitrogen (mg/1)
Chain Bridge (0.0)	0.08 - 0.20	0.02 - 0.10	0.3 - 1.0	0.10 - 0.50	0.5 - 0.9
W. Wilson Bridge (19.5)	0.30 - 1.20	0.20 - 0.80	0.8 - 1.2	1.00 - 2.00	1.5 - 3.0
Indian Head (49.3)	0.20 - 0.40	0.10 - 0.30	0.5 - 1.5	0.10 - 0.50	0.80 -1.5
Maryland Point (84.3)	0.10 - 0.25	0.08 - 0.15	0.1 - 0.3	0.05 - 0.30	0.3 - 0.6
301 Bridge (104.7)	0.05 - 0.20	0.03 - 0.10	0.1 - 0.2	0.05 - 0.20	0.2 - 0.6
Point Lookout (185.0)	0.03 - 0.060	0.01 - 0.04	0.0 - 0.1	0.05 - 0.10	0.2 - 0.4

bacteria oxidize NH^3 to $NO_2 + NO_3$. $NO_2 + NO_3$ nitrogen drops sharply between Indian Head and Maryland Point, taken up by the pronounced algal growths in this area.

Biological

The previously described differences in salinity, as well as nutrient enrichment by wastewater discharges, markedly affect the ecology of the Estuary. Under summer and fall conditions, large populations of blue-green algae, mainly Anacystis sp., prevail in the freshwater portion of the Estuary. Large standing crops of this alga occur, especially during periods of low flow, forming green mats of cells.

In the saline portion of the Potomac Estuary, the algal populations are not as dense as in the freshwater portion. At times large populations of marine phytoplankton occur, primarily Gymnodinium sp. and Amphidinium sp., producing massive growths known as "red tides."

Increased nutrient loadings from wastewater since 1913 have impressively affected the dominant plant forms in the upper Estuary, as documented by Jaworski et al. (1972) and shown in Figure 2. Of several nutrients and other growth factors implicated as stimulating this, nitrogen and phosphorus probably are the most manageable.

Plant succession in the upper Potomac Estuary can be inferred from several studies. Cumming (1916) surveyed the Estuary in 1913-1914 and noted the absence of plant life near the major wastewater outfalls. He observed normal amounts of rooted aquatic plants on the flats or shoal areas below the urban area, but no nuisance levels of rooted aquatic plants or phytoplankton.

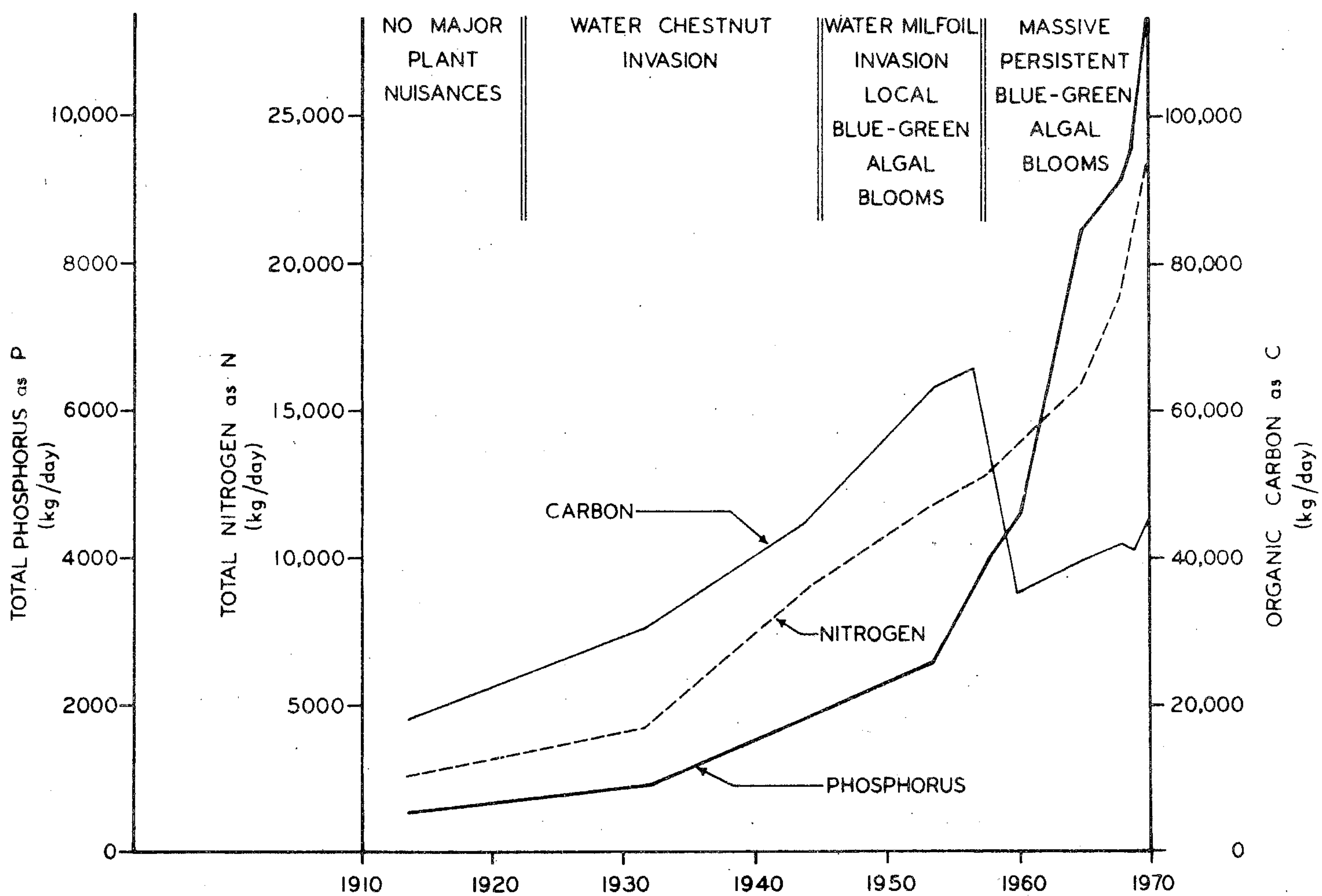

Figure 2. Wastewater nutrient enrichment trends and ecological effects.

In the 1920's, water chestnut (Trapa natans) infested the waters of Chesapeake Bay and the Potomac Estuary. This weed was controlled by mechanical removal.

In September and October, 1952, Bartsch (1954) surveyed the reaches near the metropolitan area and found that vegetation was virtually nonexistent in the area. He reported no dense phytoplankton blooms although the study did not include the downstream areas where the blooms subsequently occurred.

In 1958, a rooted aquatic plant, water milfoil (Myriophyllum spicatum), developed in the Potomac Estuary and created nuisance conditions. These increased to major proportions by 1963, especially in the embayments downstream from Indian Head (Elser, 1965). These dense strands of rooted aquatic plants disappeared rapidly in 1965 and 1966, presumably due to a natural virus (Bailey, et al., 1968).

In August and September, 1959, Scotts and Longwell (1962) surveyed the upper Estuary. They observed high levels of the nuisance blue-green alga, Anacystis sp., in the Potomac Estuary near Washington. Subsequent and continuing observations have confirmed persistent summer blooms of this alga in nuisance concentrations greater than 50 ug/l, occurring from the metropolitan area downstream at least as far as Maryland Point. Chlorophyll a determinations in the upper, middle, and lower reaches of the Potomac Estuary are presented for six key stations:

Station and Kilometers from Chain Bridge	Average Yearly Range of chlorophyll (ug/l)
Chain Bridge (0.0)	20 - 50
W. Wilson Bridge (19.5)	30 - 60
Indian Head (49.3)	30 - 150
Maryland Point (84.3)	30 - 100
301 Bridge (104.7)	10 - 30
Point Lookout (185.0)	10 - 20

Diatom blooms have been observed in the late winter and spring. The occurrence and persistence of these blooms appear greatly influenced by the spring runoff in the Potomac River Basin.

Nutrient Budgets

Runoff from the upper basin greatly influences the nutrient budgets of the Estuary reaches. Table 4 shows that the loading for carbon, nitrogen and phosphorus is a function of the discharge flow from the upper basin. Considering only upper basin runoff and wastewater discharges to the Estuary leads to the conclusion that the nutrients to be controlled by wastewater treatment are (1) phosphorus, nitrogen, and (3) carbon.

While the percentages of controllable phosphorus and nitrogen decrease at higher flows, these conditions usually occur during the months of February, March, and April when temperatures and algal crops are lowest. Since nuisance algal conditions occur primarily in the upper, freshwater portion of the Estuary, the higher flow effects are considerably less

TABLE 4

SUMMARY OF MAJOR NUTRIENT SOURCES

Upper Reach of the Potomac Estuary

Low-flow Conditions
(95 % of time exceeded)

(Potomac River Discharge at Washington, D. C. = 40 meters^3/sec)

	Upper Basin Runoff* (kg/day)	Percent of Total	Estuarine Wastewater Discharges (kg/day)	Percent of Total	Total (kg/day)
Carbon	77,100	52	72,600	48	148,700
Nitrogen	3,000	10	27,200	90	30,200
Phosphorus	450	4	10,900	96	11,350

Median-flow Conditions
(50 % of time exceeded)

(Potomac River Discharge at Washington, D. C. = 185 meters^3/sec)

	Upper Basin Runoff* (kg/day)	Percent of Total	Estuarine Wastewater Discharges (kg/day)	Percent of Total	Total (kg/day)
Carbon	159,000	68	72,600	32	231,600
Nitrogen	18,100	40	27,200	60	45,300
Phosphorus	2,400	18	10,900	82	13,300

High-flow Conditions
(5 % of time exceeded)

(Potomac River Discharge at Washington, D. C. = 1150 meters^3/sec)

	Upper Basin Runoff* (kg/day)	Percent of Total	Estuarine Wastewater Discharges (kg/day)	Percent of Total	Total (kg/day)
Carbon	680,000	90	72,600	10	752,600
Nitrogen	185,000	87	27,200	13	212,200
Phosphorus	10,000	47	10,900	53	20,900

*Upper basin runoff includes both land runoff and wastewater discharges in upper basin. The contribution of ground water and direct precipitation were estimated to be insignificant.

during July, August, and September when the blooms are most prolific.

Current nutrient loading rates for the upper Estuary, the upper and middle Estuary combined, and the upper, middle and lower Estuary combined are:

Nutrient Loadings (grams/meter^2 surface area/year) at median flows

Nutrient	Upper	Upper & Middle	Upper, Middle, & Lower
Phosphorus	89.6	17.4	5.0
Nitrogen	288.0	55.6	16.9

Using the revised Vollenweider (in press) loading approach for lakes, Figure 3 shows the current rate for the three groupings of the Estuary. Figure 3 also shows the loading rate resulting from a protected degree of phosphorus control and for the year 1913 as developed in the study by Jaworski, et al. (1971).

Figure 3 demonstrates that providing a high degree of phosphorus removal will cause the loading levels for the three combined segments to approach the conditions of 1913. Moreover, another ready conclusion is that the permissible and excessive loadings to the Estuary would be considerably larger than for lakes. Nevertheless, the general overall relationship appears to hold true; that is, the critical phosphorus loading is a function of mean depth/mean hydraulic residence time.

When comparing the chlorophyll data of the Estuary to those of the OECD lakes, the Estuary appears less affected by high concentrations of chlorophyll. In part this may be due to the greater mixing of the Estuary, compared to lakes.

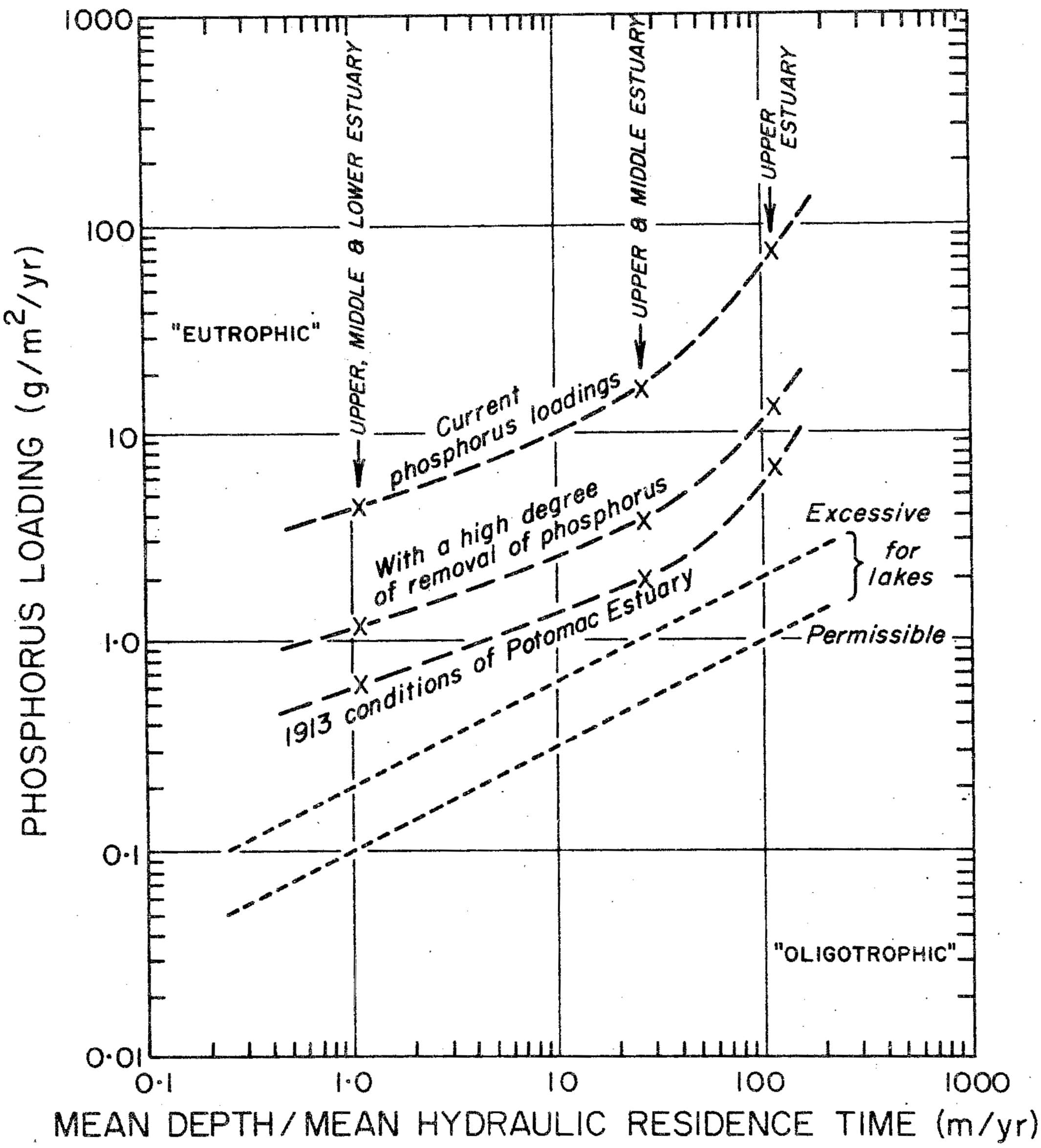

Figure 3. Phosphorus loading vs hydraulic residence time.

DISCUSSION

The eutrophication problems in the Potomac Estuary are more pronounced than those of other North Atlantic Coast estuaries. While the James and Delaware Estuaries are experiencing some eutrophication problems, the more severe conditions in the Potomac can be classified as hyper-eutrophic.

Analysis of the Potomac Estuary is complicated by two variables: (1) salinity, and (2) light-limiting conditions. Frequently high sediment loads from the upper drainage basin and suspended matter in wastewater discharges make the upper portion turbid. This light-limiting condition restricts algal growth in the upper portion of the upper reach. In the middle and lower reaches, light penetration increases; however, salinity also increases, resulting in a transition from fresh-water to marine-water organisms.

Determining appropriate alternatives for water quality management, including the eutrophication problem, of the Potomac Estuary requires the ability to predict the effect of removing essential nutrients. Numerous investigations, most dealing with lake eutrophication, have attempted various approaches to relate trophic state and nutrient input.

An approach to defining a relationship between the ecology of the Estuary and nutrient input can be delineated from the historical data in Figure 2 and Tables 2 and 4. The Estuary responded dramatically to the large increase of nutrients mainly from the wastewater discharges in the Washington, D. C. area. The nutrient increase initially resulted in rooted aquatic weeds with nuisance blue-green algal growths overtaking the weeds when nutrients increased more. Figure 2 shows that phosphorus and nitrogen loadings should be about equal to the 1913-1920 conditions---about 600 kg/day of phosphorus and 3000 kg/day of nitrogen from wastewater discharges---that resulted in no major plant nuisances.

Mathematical modeling has been another approach to relate trophic state to nutrient input. Studies summarized by Jaworski (1975) have shown that the upper and middle reach of the Estuary become nitrogen limited in the summer months. Recent model studies by Clark (personal communication) project that instituting a high degree of phosphorus removal at the wastewater treatment facilities in the Washington, D. C. area will make the Estuary either phosphorus or nitrogen limited. The degree to which either nutrient becomes limiting depends on factors such as runoff and distance along the Estuary. Jaworski et al. (1970) used mathematical models with a 25 μg/l chlorophyll target to estimate wastewater nutrient loadings of 1000 kg/day of phosphorus and 3100 kg/day of nitrogen into upper zones of the Estuary.

A third appraoch, the loading concept developed by Vollenweider, relates nutrient loading to mean depth/mean hydraulic residence time. This has been developed mainly for phosphorus and lakes. As previously indicated, this method appears applicable to theEstuary but with higher excessive and permissible loadings. Using an 18 $g/m^2/yr$ loading rate from Figure 3, the loading for the Upper Estuary would be about 500 kg/day from wastewater effluents after subtracting the non-point source contribution.

The three approaches yield about the same values for phosphorus and nitrogen loadings. However, the Vollenweider loading concept needs further verification with other estuaries before definitive relationships can be formulated.

SUMMARY

High oxygen-consuming and nutrient loadings, mainly from domestic wastewater discharges, have degraded the water quality of the Potomac Estuary. This high nutrient input has resulted in severe eutrophication problems in the Estuary.

In this report the concept of critical phosphorus loading as a function of mean depth/mean hydraulic residence time is applied to the Potomac Estuary. When compared to lakes, the Potomac Estuary apparently has a much higher capacity for assimilating nutrients. Furthermore, the Estuary apparently can tolerate high trophic states because it is a well-mixed system.

The critical phosphorus loadings compare favorably to estimates derived from historical data and mathematical modeling efforts. However, more research is needed to determine the validity of using loading concepts on estuaries.

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THE JOHN H. KERR RESERVOIR

VIRGINIA - NORTH CAROLINA

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INTRODUCTION

The 2,785 foot long concrete dam that impounds John H. Kerr Reservoir is located in Mecklenburg County, Virginia, on the Roanoke River, about 178.7 river miles above the mouth in the Albermarle Sound, 20.3 miles downstream from Clarksville, Virginia; 18 miles upstream from the Virginia-North Carolina State Line and 80 air miles from Richmond, Virginia. Formed by closure of the dam in 1952, the impoundment is a multipurpose project and was built for reduction of flood damage in Lower Roanoke River, for generation of hydroelectric power and for low water control for pollution abatement and conservation of fish and wildlife.

GEOGRAPHIC DESCRIPTION

John H. Kerr Reservoir

Latitude - 36° 35' 56"; Longitude - 78° 18' 06"

Altitude - 300 feet MSL (maximum power tool)

Catchment area - Total of sub-basins and lake 7,800 sq. miles

General Climatic Data

The Climate in the Roanoke River Basin is temperate characterized by warm summers and rigorous but generally not severe winters. Light snow and subzero temperatures occur annually in the western portion of the basin and occasionally over the entire basin. The average annual temperature for the basin is about 14.4°C (58°F) and average monthly temperatures vary between 3.3°C (38°F) and 25°C (77°F), (See Table 1 on following page for detailed monthly temperatures).

The average annual precipitation over the entire basin is about 43 inches with annual extremes of 27 and 56 inches and is well distributed throughout the year. Precipitation varies from 50 inches near the mouth of the Roanoke River, decreases with distance inland to 42 inches at about the center of the basin, and then increases with elevation to approximately 54 inches at the headwaters of the Dan River. In the vicinity of the John H. Kerr Reservoir the average annual precipitation is about 43 inches. In the area at the headwaters of the Roanoke River which lies between two mountain ranges (Allegheny and Blue Ridge Mountains), the average annual precipitation is 38 inches. The average annual snowfall is about 13 inches and does not accumulate sufficiently to have a noticeable effect on flood flows.

Prevailing winds over the basin blow from the west to northwest in the mountains and westerly elsewhere. The average annual wind velocity is 7 to 11 miles per hour. Wind velocities reach and exceed 80 miles per hour during various types of storms. Most of the annual wind damage occurs during intense thunderstorms.

The evaporation rate in the basin averages 37 inches from April to September, which is 80 to 85 percent of the annual evaporation rate based on records for the years 1954 to 1958 at Philpott and Kerr Reservoirs.

Table 1

Average Maximum, Average Minimum and Normal Monthly Air Temperatures*
John H. Kerr Dam

Month	Average Maximum		Average Minimum		Normal	
	°C	°F	°C	°F	°C	°F
January	9.6	49.3	-2.6	27.3	3.5	38.3
February	11.3	52.3	-1.8	28.7	4.7	40.5
March	15.7	60.3	1.9	35.5	8.8	47.9
April	21.2	70.2	6.8	44.3	14.1	57.3
May	26.3	79.3	12.4	54.3	19.3	66.8
June	30.1	86.2	17.1	62.7	23.6	74.5
July	31.6	88.9	19.3	66.7	25.4	77.8
August	30.7	87.2	18.6	65.4	24.6	76.3
September	27.7	81.9	14.9	58.9	21.3	70.3
October	22.2	72.0	7.7	45.8	15.0	59.0
November	16.8	60.8	1.8	35.2	8.9	48.1
December	9.5	49.1	-2.0	28.3	39.4	39.1
Annual	21.0	69.8	7.8	46.1	14.4	58.0

*Reservoir Regulation Manual, Roanoke River Basin, North Carolina-Virginia. U.S. Army Engineer District, Wilmington, Corps of Engineers, Wilmington, N.C. October 1965.

General Geological Characteristics

In general the Piedmont Province, in which Kerr Reservoir is located, is a maturely dissected upland underlain by a vast complex of igneous, metamorphic and sedimentary rocks which are exposed in broad, northeast trending belts. Deformation and intrusion of igneous material have altered preexisting igneous and sedimentary rocks into metamorphic rocks which include gneisses, schists and quartzites. The older rocks have been intensely folded, displaced by faults, and intruded by igneous rocks, predominantly granites. The complexity of the structure and the obscuring soil mantle make interpretation difficult and the age relationships of many of the older formations uncertain. One large body and three smaller outliers of Triassic sedimentary rocks occur within the Piedmont portion of the Roanoke River Basin. These rocks consist of younger, unaltered sandstones and shales which were preserved from erosion in down-faulted basins. Diabase and gabbro dikes have been injected into the Triassic rocks as well as into some of the older igneous and metamorphic rocks of the Piedmont Province. Minerals abundant enough to be of commercial value include tungsten, granite, gneiss, stone, sand and grave.

Actual sediment accumulation (due to erosion) measurements from a 9-year survey period 1950-1959 showed about one ton of sediment per acre per year (or 639 tons per square mile per year). This rate of sedimentation if extended to the whole Roanoke River Basin in Virginia would give a total of about 4 million tons of sediment per year.

Vegetation

Over 60% of the drainage area is forested predominantly by Virginia, loblolly, and shortleaf pine, and mixed-pine hardwood stands; small areas of pure hardwoods are scattered throughout the basin. Vegetation on the lake margins and in the lake is severely limited due to the fluctuating water level and wave action on the shoreline.

Other Basin Characteristics

The population of Roanoke River Basin in 1970 was 772,000. Land use was predominately rural-agricultural, approximately 60% wooded, 30% cropland and pasture, less than 10% urban and industrial. Water use of the impoundment includes flood control, hydroelectric power, low water control for pollution abatement and for conservation of fish and wildlife, recreation (fishing, swimming, boating, etc.). The reservoir waters are also under development as a regional water supply for several North Carolina towns.

Sewage and Effluent Discharges

Communities upstream of Kerr Reservoir contribute waste water effluents to the rivers and streams that flow into the basin. In nearly all instances these are treated sewages. However, in some instances plant breakdowns will release untreated wastes to the inflowing streams. A recent compilation of industrial and domestic point source discharges in the drainage of the Roanoke basin is summarized in Table 2. Monitoring of the Dan, Banister and Roanoke River and Nutbush Creek illustrates the nitrogen and phosphorus concentrations and load (kg/d) currently entering the Kerr Reservoir, Tables 3 and 4. The configuration of John H. Kerr Reservoir characterized by two major arms each with substantially different morphometric and hydraulic dimensions (see map and Tables 5 and 6) requires that nutrient loading rates and characteristic productivity responses be examined independently for each. In turn, since each arm receives its major nutrient input at the head end, each arm has been subdivided into five linear compartments, the discharge from each becoming the inflow to the next downstream segment.

Table 2

John H. Kerr Reservoir
Point Source Discharges, Industrial and Municipal
In Reservoir Drainage Area*

County	Industrial			Domestic		
Virginia	No.	MGD	BOD_5 lb/day	No.	MGD	BOD_5 lb/day
Montgomery	3	.410	75	5	.181	54
Roanoke	15	2.757	75	6	29.029	6,077
Bedford	9	.305	-	18	.968	975
Franklin	3	.023	8	4	.606	154
Patrick	5	.528	303	5	.091	164
Henry	11	54.799	3,279	6	2.939	2,637
Pittsylvania	10	16.834	20,334	11	10.173	935
Campbell	10	6.364	1,995	12	.830	454
Charlotte	5	.385	34	11	.187	105
Halifax	12	4.492	11,325	18	1.770	1,965
Appomattox	-	-	-	1	.100	30
Mecklenberg	-	-	-	15	.837	346
Other minor discharges	-	-	-	92	1.088	561
North Carolina						
Vance	-	-	-	1	1.500	626
Granville	-	-	-	1	3.800	1,809
Rockingham	-	-	-	4	5.450	2,823
Total	83	86.897	37,428	199	48.799	19,715

*Data for Virginia assembled from tabulations prepared by Hayes, Seay, Mattern and Mattern for the Roanoke River Basin Study and provided by the Wilmington District, U.S. Army Corps of Engineers. North Carolina data from The Division of Environmental Management, Department of Economic and Natural Resources.

Table 3

John H. Kerr Reservoir
Concentration (mg/l) Nitrogen and Phosphorus Fractions in Feeder Rivers and Streams
Mean Values of Monthly Samples

	Average Flow (cfs) Period of Sampling***	Kjel-N	NH_3-N	NO_2+NO_3-N	Total N	PO_4-P	Total Dissolved P	Total P	TN/TP
Roanoke River*	3306	.252	.049	.298	.552	.011	.020	.065	8.5
Banister River	574	.322	.078	.222	.550	.008	.016	.041	13.4
Dan River*	2835	.375	.097	.459	.828	.072	.089	.150	5.5
North Drainage									
Bluestone Creek	47	.247	.060	.113	.360	.010	.018	.036	10.0
Little Bluestone Creek	19	.294	.065	.241	.535	.062	.051	.137	3.9
Butcher Creek	25	.248	.074	.242	.488	.020	.036	.070	7.0
South Drainage									
Hyco River	309	.310	.061	.335	.649	.021	.033	.075	8.6
Aarons Creek	48	.241	.074	.270	.512	.011	.016	.028	18.3
Grassy Creek	64	.248	.067	.191	.439	.010	.019	.036	12.2
Island Creek	33	.210	.060	.150	.370	.009	.017	.035	10.6
Little Island Creek	21	.202	.069	.213	.415	.008	.017	.038	10.9
Nutbush Arm									
Flat Creek - 1326	14	.236	.105	.143	.379	.008	.016	.034	11.1
Flat Creek - 39	-	.312	.056	.140	.453	.006	.014	.039	11.6
Nutbush Creek	9	12.74	6.53	.613	13.01	2.62	3.57	4.45	2.9
Kerr Dam Discharge**	8322	.366	.135	.210	.548	.014	.021	.035	15.7

*28 to 31 monthly samples, July 1972-March 1975;
No star, 11-12 monthly samples April 1974-March 1975

**16-18 samples for the several N and P forms, September 1973-April 1975

***Average of flow on dates of sampling

Table 4

John H. Kerr Reservoir
River and Stream Flows and Nutrient Loads Based on Monthly Samples

	No. of Samples	Average c.f.s.	Total-N Kg/d	Total-P Kg/d	N/P
Roanoke River*					
July 1972-March 1973	8	3,488	4,380	608	7.2
April 1973-March 1974	11	3,707	5,495	524	10.5
April 1974-March 1975	12	2,725	3,248	574	5.6
Banister River*					
July 1972-March 1973	8	562	659	51	12.9
April 1973-March 1974	11	546	833	64	13.0
April 1974-March 1975	12	615	880	73	12.1
Dan River*					
July 1972-March 1973	8	2,769	3,977	765	5.2
April 1973-March 1974	11	2,934	6,306	1,174	5.4
April 1974-March 1975	12	2,803	5,770	970	5.9
North Drainage - April 1974-March 1975					
Bluestone Creek	12	47	41	4.1	10.0
Little Bluestone Creek	12	19	24	6.4	3.8
Butcher Creek	12	25	31	4.3	7.2
South Drainage - April 1974-March 1975					
Hyco River	12	309	422	49	8.6
Aarons Creek	12	48	68	3.4	20.0
Grassy Creek	12	64	75	5.7	13.2
Island Creek	12	33	32	2.9	11.0
Little Island Creek	12	21	25	2.2	11.4
Nutbush Arm					
Flat Creek - April 1974-March 1975	12	14	14	1.3	10.8
Nutbush Creek					
July 1972-March 1973	8	10.9	266	81	3.3
April 1973-March 1974	11	8.3	282	87	3.2
April 1974-March 1975	12	6.4	150	61	2.5
Kerr Dam*					
Sept. 1973-March 1974	6	7,425	11,509	553	20.8
April 1974-March 1975	12	9,219	11,032	1,033	10.6
April 1974-March 1975	365	8,859	-	-	-

*Gaged flows, others calculated from weighted drainage area.

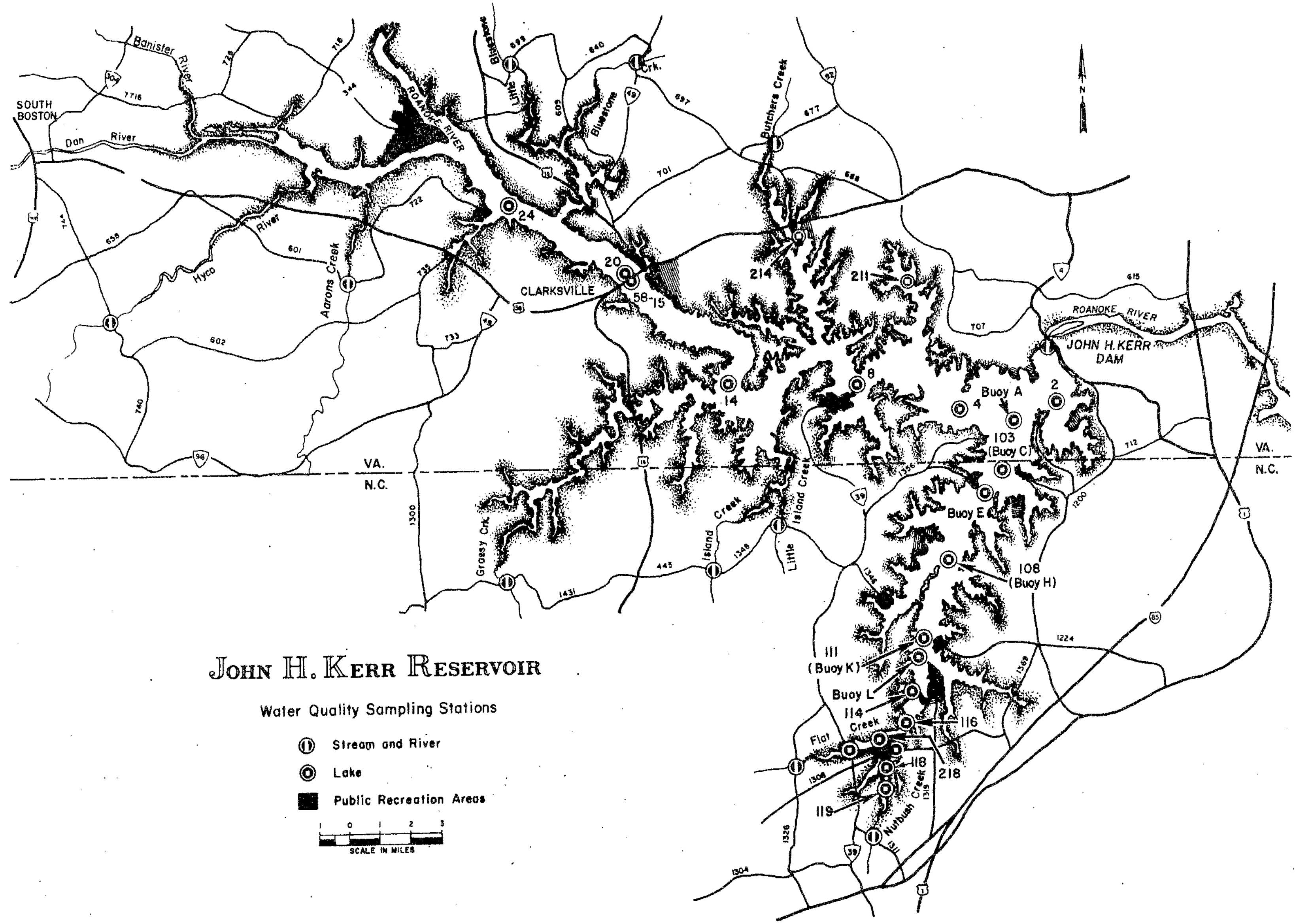
JOHN H. KERR RESERVOIR
Water Quality Sampling Stations
Stream and River
Lake
Public Recreation Areas
SCALE IN MILES
SOUTH BOSTON
Dan River
Banister River
ROANOKE RIVER
Hyco River
Aarons Creek
Little Bluestone
Bluestone Crk.
Butchers Creek
CLARKSVILLE
Grassy Crk.
Island Creek
Little Island Creek
Flat Creek
Nutbush Creek
JOHN H. KERR DAM
VA.
N.C.
24
20
58-15
214
211
14
8
4
2
Buoy A
103 (Buoy C)
Buoy E
108 (Buoy H)
111 (Buoy K)
Buoy L
114
116
118
218
119

Table 5

John H. Kerr Reservoir
Distance of Sampling Stations or Reference Buoys From
John H. Kerr Dam

Roanoke Arm	Miles	Km
Dam	0	0
2	1.5	2.4
4	4.5	7.3
8	8.5	13.7
14	13.2	21.3
58-15*	19.5	31.3
20*	20	32.2
24*	24	38.6
Nutbush Arm		
Buoy A	2.7	4.3
C (103)	4.7	7.5
E	5.4	8.7
H (108)	7.9	12.7
K (111)	10.7	17.3
L	11.5	18.5
N (114)	12.6	20.3
P (116)	13.4	21.5
218 (From Buoy P)	0.6	0.9
1308	14.0	22.4
118	14.6	23.3
119	15.0	24.0

*Distance scaled from 1,250,000 USGS quadrangle "Greensboro, N. C." Other distances scaled from USGS 1/24,000 quadrangles, "John H. Kerr Dam" and "Middleburg, N. C."

Table 6

John H. Kerr Reservoir
Morphometric and Hydrologic Characteristics

Reservoir Surface	Acres	Hectares
At maximum flood-control pool (elev. 320)	83,200	33.670
At maximum power pool (elev. 300)	48,900	19,789
At minimum power pool (elev. 268)	19,700	7,972
Original river area (below elev. 320)	4,280	1,732
River elevation at dam elev. 200	-	-
Length at elevation 320	Miles	Kilometers
Roanoke River	56	90
Dan River above junction	34	55
Nutbush Arm above Buoy A	14	22
Length of Shoreline at elevation 300	800	1,287
Maximum width at elevation 300	2	3.2
Volume	Acre-Ft.	Meter^3 X 10^3
Flood storage elev. 320 to elev. 300	1,278,000	1,576,400
Volume at elevation 300	1,472,300	1,816,067
Power draw down (elev. 300 to elev. 268)	1,029,100	1,269,384
Mean Depth	Feet	Meters
Roanoke Arm, Dam to Buoy 24	33.7	10.3
Nutbush Arm, Buoy A to 1308 bridge	29.7	9.1
Ratio of Epilimnion to Hypolimnion - Transition depth	50	15.2

Acres - 48,900/11,000 hectares - 20,231/4,452
Acre-Ft. - 1,472,300/186,800 meters^3 X 10^3 - 1,816,067/230,416

Stratification

Seasonal heating generally produces a thermal gradient of more than 2°C, in depths of 70-80 ft. (21.3-24.4 m) as early as mid-March. The upper 15 feet (4.6 m) may still be well mixed at this time. By mid-May the temperature differential between the surface and the deeper portions of the reservoir has increased to 5°C with the transition depth between 20 and 25 ft. (6.1-7.6 m). In the upper arms of the reservoir the transition depth shallows to a depth of 10 to 15 ft. (3.0-4.6 m). In spite of hydro-power withdrawals stratification persists with a 10°C differential, top to bottom, evident in August and the transition depth persisting between 40 to 50 ft. (12.2-15.2 m). Seasonal cooling produces the fall overturn late in November and the reservoir is generally well mixed by early December. Water temperatures lower than 4°C are seldom found during the short winter period of December through February.

Lake Sediments

Bottom sediment samples from locations along the axes of the Roanoke and Nutbush Arms of John H. Kerr Reservoir as well as in several of the lateral arms that were also sampled for benthic organisms, were characterized according to particle size dimension. These defined the sand, silt and clay content. In addition the carbon content of these sediments was also determined by dichromate oxidation, Tables 7 and 8. As might be expected the sand content of the bottom sediments was much higher at the upper end of the Roanoke Arm changing to a higher proportion of clay in the deeper portion of the impoundment. Along the Nutbush Arm the silt content was generally greater than sand, which was primarily limited to sublittoral locations. The carbon content of the Nutbush Arm was also greater at its upper end where a substantial pollution load enters. Even at the farthest downstream station, 103, the carbon content of the sediment was still slightly higher than the average carbon content of the main impoundment.

Seasonal Variation of Precipitation

The rainfall pattern of this area is characterized by regional precipitation originating in air masses flowing from the Gulf of Mexico. This is generally true of the winter and spring rains which give way to localized thunderstorms from May to September. Except for the random intrusion of subtropical hurricanes, the fall months, particularly October and November, are the driest although midsummer droughts are quite common. The monthly

Table 7

John H. Kerr Reservoir
Particle Size Characteristics of Bottom Sediments

Particle Size Range, mm.	Percent, By Weight, Total Sample		
	Sand 1-0.625	Silt 0.0039-0.0625	Clay 0.0039
Stations			
Roanoke Arm			
24	17.8	31.3	51.2
20	21.7	26.9	53.0
14	0.0	8.3	91.7
8	0.0	24.0	76.0
2	0.0	11.8	88.2
Butcher Creek			
214	0.0	34.4	65.6
Eastland Creek			
211	0.0	30.2	69.8
Nutbush Arm			
119	0.0	84.5	15.5
118	0.0	74.6	25.4
118E	62.2	21.5	17.3
1308	20.3	13.2	65.9
1308E	35.0	40.3	24.7
116	7.7	35.6	56.7
114	6.0	27.0	67.0
114E	38.1	41.9	20.0
111	0.0	24.4	75.6
111W	73.6	17.8	8.6
108	0.0	22.0	78.0
103	0.0	16.0	84.0
103W	62.2	25.7	12.1
Flat Creek			
219	13.9	28.9	57.2
219S	34.9	34.7	30.5
218	0	38.4	61.6
218S	37.2	40.3	22.5

Letter designated stations are sub-littoral locations, others center channel locations.

Table 8

John H. Kerr Reservoir
Percent Carbon Content Bottom Sediments

Date Sampled	Aug. 1973	Nov.	Feb. 1974	April	Average
Station					
Roanoke Arm					
24	-	-	2.48	1.75	2.12
20	-	-	2.03	1.88	1.94
14	-	-	1.78	2.17	1.98
8	-	-	2.22	2.32	2.27
2	-	-	2.23	2.13	2.10
Butcher Creek					
214	2.03	2.10	2.23	2.13	2.12
Eastland Creek					
211	1.71	1.66	1.89	2.00	1.82
Nutbush Arm					
119	4.03	3.97	3.83	3.95	3.95
118	3.33	3.38	3.38	3.35	3.36
118E	0.94	1.04	0.71	0.33	0.76
1308	2.46	2.35	2.23	1.90	2.24
1308E	-	1.15	1.14	0.95	1.08
116	-	2.67	2.90	3.00	2.85
114	2.30	2.31	2.56	2.60	2.36
114E	0.79	0.60	0.71	0.70	0.70
111	-	2.27	2.27	2.30	2.28
111W	0.48	0.20	0.65	0.90	0.56
108	-	-	2.30	2.50	2.40
103	-	-	2.20	2.51	2.36
103W	0.52	0.30	0.67	0.53	0.51
Flat Creek					
219	1.57	1.63	1.20	1.90	1.58
219S	0.51	-	0.46	0.73	0.57
218	2.31	2.13	1.97	2.40	2.20
218S	0.60	-	0.59	0.65	0.61

Percentages carbon determined by dichromate oxidation.
For conversion to "organic matter" multiply by a factor of 1.33.
Letter designated stations are sub-littoral locations, others center channel locations.

precipitation record for Henderson, N. C., at the head of the Nutbush Arm of John H. Kerr Reservoir, is presented in Table 9.

Water Renewal Time

The water renewal or retention time of an impoundment operated both for hydropower and flood control needs to be considered over a range of discharges. For both arms of John H. Kerr Reservoir water retention has been calculated over a range of discharges based on annual averages, Table 10. The difference in retention time of the two arms is generally by a factor of 30.

LIMNOLOGICAL CHARACTERISTICS

Physical and Chemical

Year round collection of limnological data from John H. Kerr Reservoir early established a lengthy stratification period, April-November inclusive, and a limited period in which the body of water was in a mixed condition, December-March inclusive. Water temperatures in the reservoir during the current period of observation never fell to 4°C and thus stratification in the spring generally proceeded rapidly. In the following data tables, the presentations when feasible are organized into the two yearly periods of April-November and December-March. Samples collected after November 25, in some instances, were considered as part of the winter period. Data from vertical profiles are averaged as epilimnetic or hypolimnetic with the transition depth indicated for each station. Physical characteristics are presented in Tables 11 and 12 and chemical characteristics in Tables 13, 14 and 15.

Biological-Phytoplankton

As with the physical and chemical parameters the phytoplankton have been grouped into April-November and December-March data sets. For this report the quantitative phytoplankton presentation is limited to cell no. per milliliter.

Table 9

John H. Kerr Reservoir
Seasonal Variation of Monthly Precipitation*

Virginia Stations	Total Annual Rainfall - Inches 1972	1973	1974	Long Term Average
Halifax	49.07	38.30	49.24	41.83
Clarksville	51.53	45.41	43.62	41.91
John H. Kerr Dam	52.25	45.61	43.43	-
North Carolina Station				
Henderson	50.30	49.91	47.96	40.04

Henderson	Monthly Precip. - inches	Departure From Average
December 1973	6.37	3.22
January 1974	4.38	1.11
February	3.70	.34
March	4.03	.31
April	1.34	-1.75
May	5.89	2.27
June	3.37	-.79
July	.99	-4.57
August	10.03	5.37
September	7.76	4.16
October	1.26	-1.36
November	1.19	-1.94
December	4.02	.77
January 1975	6.22	2.95
February	2.97	-.39

*Data from National Climate Center, Asheville, N. C.

Table 10

John H. Kerr Reservoir
Water Renewal Time (Retention Time)-Days
Regulated Discharge - Hydropower and Flood Control

Average Flow Rates - c.f.s.	Roanoke Arm 1,054,719 Acre-Ft. (elev. 300)
7,500	71
8,000	67
8,500	63
9,000	59
9,500	56
10,000	53
	Nutbush Arm 363,400 Acre-Ft. (elev. 300)
90	2,036
95	1,929
100	1,832
105	1,745
110	1,666
115	1,593
120	1,527

Table 11

John H. Kerr Reservoir
Physical Characteristics
Temperature, Conductivity, Light Penetration, Secchi Depth, Turbidity, Color, Solar Radiation (Day Light Hours)
April - November Inclusive
Average Values of All Samples From Designated Segment of Vertical Profile

Roanoke Arm	Average Transition Depth-M	Temperature °C		Cond. µmho		Light 1% Depth-M	Secchi Depth-M	Turbidity-JTU		Color Surface	Day Light-Hrs.	
		Epi.	Hypo.	Epi.	Hypo.			Epi.	Hypo.		Min.	Max.
Station												
24	3.65	23.5	21.6	106	112	2.74	.82	13.5	21.0	13	10.0	14.5
20	3.81	23.9	21.8	101	108	3.90	1.18	10.7	18.2	12		
58-15	*	22.2	-	95	-	-	1.00	14.8	-	16		
14	4.91	23.2	21.0	97	103	4.45	1.28	9.3	18.7	13		
8	10.24	22.6	18.6	92	106	5.42	1.71	7.6	19.7	12		
2	10.57	22.5	18.3	91	104	6.71	2.04	7.3	11.5	10		
Nutbush Arm												
119	.88	22.1	20.9	325	334	1.49	.58	21.1	20.9	16		
118	1.71	23.4	20.6	216	242	2.01	.67	16.0	18.0	17		
218	3.05	20.8	19.9	116	117	3.05	.94	13.4	21.4	14		
1308	2.65	22.4	20.2	168	207	2.38	.85	14.0	21.2	13		
116	3.54	21.3	20.2	133	155	3.78	1.19	12.5	15.5	10		
114	4.39	21.6	20.2	112	123	3.17	1.19	10.6	15.7	10		
111	7.77	22.4	20.3	107	117	4.11	1.40	10.3	13.6	9		
108	8.44	22.7	19.9	99	107	4.51	1.58	8.2	10.7	10		
103	10.49	22.5	19.0	95	109	5.91	1.77	8.7	11.1	10		

*Samples from Secchi Depth - monthly

Epi - Epilimnetic samples; Hypo - Hypolimnetic samples.

Table 12

John H. Kerr Reservoir
Physical Characteristics
Temperature, Conductivity, Light Penetration, Secchi Depth, Turbidity, Color, Solar Radiation (Day Light Hours)
December - March Inclusive
Average Values of All Samples From Designated Segment of Vertical Profile

Roanoke Arm	Average Transition Depth-M	Temperature °C Epi.	Temperature °C Hypo.	Cond. µmho Epi.	Cond. µmho Hypo.	Light 1% Depth-M	Secchi Depth-M	Turbidity-JTU Epi.	Turbidity-JTU Hypo.	Color Surface	Day Light-Hrs. Min.	Day Light-Hrs. Max.
Station												
24	4.57	9.4	8.6	80	80	1.06	0.58	39.5	43.3	35	9.8	12.5
20	4.57	9.3	9.2	86	86	2.74	0.55	38.6	43.9	21		
58-15	*	8.4	-	83	-	-	0.43	42.7	-	22		
14	mixed	8.8	-	81	-	2.53	0.70	31.9	-	28		
8	11.67	9.2	8.2	76	82	2.20	0.76	31.8	45.7	30		
2	5.33	9.5	8.8	80	81	2.65	0.88	18.6	17.8	19		
Nutbush Arm												
119	mixed	8.3	8.3	323	323	.91	.40	29.5	29.5	83		
118	mixed	9.9	9.9	294	294	1.01	.40	27.4	27.4	34		
218	mixed	7.2	7.2	110	110	.90	.30	31.6	31.6	18		
1308	mixed	8.3	8.3	220	220	1.31	.49	22.4	22.4	22		
116	3.05	7.9	7.7	163	166	1.52	.61	22.4	24.5	13		
114	mixed	8.9	8.9	130	130	2.65	.88	15.5	15.5	11		
111	6.10	9.2	9.1	106	109	3.51	1.07	11.4	11.9	11		
108	4.57	9.2	9.2	93	94	2.83	1.25	10.7	10.9	12		
103	11.43	9.1	8.8	83	87	3.44	1.04	13.2	15.6	26		

*Samples from Secchi Depth - monthly.

Epi - Epilimnetic samples; Hypo - Hypolimnetic samples.

Table 13

John H. Kerr Reservoir
Chemical Characteristics - pH, DO, N and P Fractions
April to November Inclusive
Average Values of All Samples from Designated Segment of Vertical Profile

Roanoke Arm	Average Transition Depth-M	pH Epi.	pH Hypo.	DO Epi.	DO Hypo.	Kjel-N Epi.	Kjel-N Hypo.	NH_3-N Epi.	NH_3-N Hypo.	NO_2+NO_3-N Epi.	NO_2+NO_3-N Hypo.	Total N Epi.	Total N Hypo.	PO_4-P Epi.	PO_4-P Hypo.	Total Sol-P Epi.	Total Sol-P Hypo.	Total-P Epi.	Total-P Hypo.
Station																			
24	3.65	8.1	7.4	9.1	6.3	.342	.266	.030	.168	.118	.183	.460	.476	.007	.008	.017	.019	.047	.068
20	3.81	8.1	7.6	9.3	5.8	.328	.323	.042	.100	.087	.157	.416	.480	.006	.007	.011	.015	.032	.036
58-15	*	7.9	-	9.7	-	.554	-	.072	-	.222	-	.792	-	.009	-	.017	-	.045	-
14	4.91	7.7	7.3	8.1	4.1	.294	.298	.048	.135	.128	.221	.441	.520	.006	.007	.010	.012	.023	.036
8	10.24	7.4	6.8	7.4	2.1	.258	.448	.053	.255	.075	.186	.320	.631	.006	.009	.010	.016	.025	.049
2	10.57	7.6	6.9	8.1	2.8	.252	.299	.048	.233	.051	.127	.304	.426	.006	.008	.011	.015	.016	.024
Nutbush Arm																			
119	.88	8.8	7.9	11.2	6.7	.925	.984	.162	.336	.045	.078	.970	1.062	.079	.110	.156	.115	.285	.252
118	1.71	8.9	7.9	11.0	5.8	.598	.649	.063	.181	.046	.031	.644	.680	.028	.037	.037	.047	.124	.135
218	3.05	7.9	7.5	9.3	7.1	.289	.304	.034	.090	.045	.048	.334	.351	.011	.008	.013	.011	.036	.044
1308	2.65	8.2	7.4	9.4	4.3	.477	.604	.077	.273	.032	.050	.509	.654	.009	.023	.017	.034	.061	.105
116	3.54	7.7	7.5	8.9	6.1	.293	.375	.051	.130	.055	.048	.349	.424	.007	.009	.008	.012	.036	.052
114	4.39	7.7	7.2	8.7	5.4	.317	.410	.047	.124	.047	.042	.364	.414	.006	.007	.008	.014	.027	.035
111	7.77	7.7	7.1	7.7	3.0	.287	.364	.056	.168	.030	.034	.317	.399	.007	.006	.011	.013	.020	.023
108	8.44	7.7	7.1	7.9	3.1	.284	.354	.044	.157	.034	.061	.317	.416	.006	.006	.009	.009	.016	.020
103	10.49	7.5	6.9	7.8	3.7	.239	.336	.047	.178	.046	.098	.280	.370	.006	.007	.008	.010	.015	.018

DO, N and P fractions mg/1

Epi - Epilimnetic samples; Hypo - Hypolimnetic samples.

*Samples from Secchi Depth - monthly.

Table 14

John H. Kerr Reservoir
Chemical Characteristics - pH, DO, N and P Fractions
December to March Inclusive
Average Values of All Samples from Designated Segment of Vertical Profile

Roanoke Arm	Average Transition Depth-M	pH Epi.	pH Hypo.	DO Epi.	DO Hypo.	Kjel-N Epi.	Kjel-N Hypo.	NH_3-N Epi.	NH_3-N Hypo.	NO_2+NO_3-N Epi.	NO_2+NO_3-N Hypo.	Total N Epi.	Total N Hypo.	PO_4-P Epi.	PO_4-P Hypo.	Total Sol-P Epi.	Total Sol-P Hypo.	Total-P Epi.	Total-P Hypo.
Station																			
24	4.57	6.7	6.6	9.9	9.6	.218	.260	.090	.128	.225	.228	.443	.448	.015	.014	.024	.024	.117	.122
20	4.57	6.7	6.7	9.3	10.2	.245	.245	.068	.065	.245	.248	.490	.492	.014	.014	.024	.025	.098	.101
58-15	*	6.9	-	10.3	-	.339	-	.113	-	.318	-	.669	-	.018	-	.028	-	.079	-
14	mixed	6.9	6.9	10.5	10.5	.245	.245	.068	.068	.223	.223	.468	.468	.007	.007	.021	.021	.069	.069
8	11.67	6.9	6.8	10.3	9.4	.236	.250	.066	-	.216	.223	.468	.493	.007	.007	.017	.019	.025	.054
2	5.33	6.8	6.8	10.4	10.3	.216	.206	.060	.066	.266	.287	.493	.498	.007	.007	.016	.017	.036	.036
Nutbush Arm																			
119	mixed	8.1	8.1	9.1	9.1	1.107	1.107	.333	.333	.350	.350	1.457	1.457	.063	.063	.139	.139	.325	.325
118	mixed	7.9	7.9	10.3	10.3	1.195	.195	.505	.505	.093	.093	1.288	1.288	.067	.067	.126	.126	.268	.268
218	mixed	7.1	7.1	11.4	11.4	.335	.335	.095	.095	.115	.115	.450	.450	.006	.006	.029	.029	.076	.076
1308	mixed	6.9	6.9	10.7	10.7	.886	.886	.254	.254	.152	.152	1.04	1.04	.032	.032	.079	.079	.156	.156
116	3.05	7.2	7.1	11.1	11.2	.535	.520	.070	.070	.110	.110	.645	.630	.007	.007	.041	.040	.087	.091
114	mixed	7.1	7.1	10.8	10.8	.394	.394	.070	.070	.130	.130	.524	.524	.006	.006	.012	.012	.042	.042
111	6.10	7.2	7.2	10.6	10.7	.325	.333	.060	.060	.163	.158	.488	.490	.006	.006	.009	.009	.024	.024
108	4.57	7.2	7.2	10.5	10.6	.263	.268	.058	.058	.193	.190	.455	.458	.006	.006	.010	.010	.024	.024
103	4.43	7.1	6.9	10.3	9.9	.220	.240	.050	.063	.240	.270	460	.510	.006	.006	.014	.014	.029	.031

DO, N and P fractions mg/l

Epi - Epilimnetic samples; Hypo - Hypolimnetic samples.

*Samples from Secchi Depth - monthly.

Table 15

John H. Kerr Reservoir
Alkalinity, Ca, Mg, Na, K, $SO_4^{=}$, Cl^{-}, Fe
September 1972 - March 1975
Average Values of All Samples From Designated Segment of Vertical Profile

Roanoke Arm	Average Transition Depth-M	Alkalinity From Secchi-Depth	Ca (mg/1)		Mg		Na		K		$SO_4^{=}$		Cl^{-}		Total Fe		Diss. Fe	
			Epi.	Hypo.	Epi.	Hypo.	Epi.	Hypo.	Epi.	Hypo.	Epi.	Hypo.	Epi.	Hypo.	Epi.	Hypo.	Epi.	Hypo.
Station																		
24	3.65	29.5	2.7	2.6	2.8	2.9	7.1	6.9	2.4	2.5	6.5	6.5	5.8	5.7	.62	1.22	.12	.14
20	3.81	28.7	2.8	2.9	2.8	2.8	6.9	6.9	2.3	2.4	6.5	6.8	6.6	6.2	.45	.79	.08	.09
58-15	*	23.2	2.6	-	2.7	-	6.3	-	2.4	-	7.2	-	5.6	-	1.23	-	.27	-
14	4.91	29.0	2.8	2.8	2.9	2.9	6.4	6.3	2.1	2.3	6.5	6.8	5.4	5.3	.43	.76	.07	.11
8	10.24	26.1	2.8	2.6	2.8	2.9	6.0	6.2	2.0	2.2	6.9	6.6	5.3	5.4	.36	1.51	.13	.27
2	10.57	25.7	2.9	3.1	2.9	3.1	6.4	6.5	1.8	2.0	6.6	6.8	5.8	5.8	.45	.87	.06	.30
Nutbush Arm																		
119	.88	25.7	4.0	3.8	2.8	2.8	63.1	56.4	3.2	3.1	8.9	8.6	62.6	68.3	.53	.60	.05	.10
118	1.71	27.6	3.5	3.2	2.8	2.8	37.9	36.9	2.7	2.6	9.4	9.3	44.6	46.4	.49	.78	.21	.27
218	3.05	25.6	2.8	2.7	2.6	2.7	12.3	12.4	2.2	2.1	8.0	8.1	14.3	14.7	.40	.51	.15	.14
1308	2.65	26.5	3.3	2.9	2.8	2.8	23.8	26.4	2.3	2.3	8.6	8.7	33.0	36.3	.53	.92	.17	.22
116	3.54	18.7	3.0	3.0	2.7	2.8	13.4	16.3	2.1	2.2	7.6	7.8	18.1	20.6	.31	.40	.08	.08
114	4.39	24.7	2.9	2.9	2.8	2.8	12.6	12.8	2.0	2.1	7.1	7.0	15.1	15.9	.28	.52	.05	.06
111	7.77	24.8	3.0	2.9	2.8	2.8	10.2	10.3	2.0	2.0	6.6	6.6	11.7	12.2	.20	.44	.05	.09
108	8.44	23.8	2.9	2.9	2.8	2.9	8.0	8.5	1.9	1.9	6.6	6.3	9.0	9.2	.18	.48	.03	.09
103	10.49	25.1	2.9	3.0	2.8	3.0	6.6	6.8	1.9	1.9	7.9	6.8	6.6	6.5	.14	.87	.05	.33

*Samples from Secchi Depth - monthly

Epi - Epilimnetic samples; Hypo - Hypolimnetic samples.

Associated parameters, productivity, both P_S and P_{max}, chlorophyll a and Secchi depth, (Phytoplankton sample depth) are presented in Table 16. Characterization of the total phytoplankton community by class and percent of each class of the total is presented in Table 17.

Algal Assay

Algal assays for limiting nutrients in each of the two arms of John H. Kerr Reservoir over the total period of observation from March 1972 through May 1974 show a characteristic "downstream" decrease in growth potential as indicated by the quantity of biomass grown in the reseeded control. This was evident in both filtered and autoclaved samples. Of particular interest is the clearly indicated shift from a higher frequency of nitrogen limited assays, at the head of each arm of the reservoir, changing to more frequent phosphorus-limited assays at the downstream end, Table 18. This would be related to the observed decrease in concentration of PO_4-P and Total-P at these same stations.

Biological-Zooplankton

The total zooplankton populations of the sampling points along both arms have been defined by vertical net tows in the euphotic zone. Monthly totals and the April-November averages are presented in Table 19. A genera list is presented in Table 20.

Bottom Fauna

Dredge samples from the stations along the two major arms of the reservoir as well as several side embayments were collected in four seasonal periods to define the bottom fauna. The density of five major groups as found in the four collections is presented in Table 21.

Table 16

John H. Kerr Reservoir
Productivity and Related Parameters - Mean Values of Monthly Samples

	P_s		P_{max}	Chlor. a mg/m^3		Algal Cell Density No./ml				Secchi Depth - Meters	
Roanoke Arm	Apr.-Nov.	Dec.-Mar.	Apr.-Nov.	Apr.-Nov.	Dec.-Mar.	Apr.-Nov.	Log No.	Dec.-Mar.	Log No.	Apr.-Nov.	Dec.-Mar.
Station											
24	753	102	-	24.6	8.8	9,387	3.97	3,441	3.54	0.84	0.57
20	870	214	81.3[a]	14.0	14.0	7,887	3.89	1,861	3.27	1.19	0.55
58-15	588	122	-	16.7	5.3	9,026	3.96	3,492	3.54	0.96	0.39
14	805	142	-	16.6	8.3	9,005	3.95	1,449	3.16	1.27	0.53
8	646	295	41.3[a]	11.6	9.3	3,670	3.56	2,556	3.41	1.70	0.75
2	724	299	33.6[a]	12.3	10.1	3,240	3.51	1,716	3.23	2.04	0.87
Nutbush Arm											
119	1,405	-	-	69.2	94.2(1)	40,706	4.61	17,404	4.24	0.59	0.27
118	1,086	916	-	47.0	82.6	31,674	4.50	31,474	4.50	0.66	0.30
218	590	416	-	20.3	49.4	10,478	4.02	10,074	4.00	0.91	0.30
1308	854	908	120.6[a]	28.9	59.3	16,827	4.22	13,471	4.13	0.86	0.46
116	676	1,047	-	18.9	50.4	10,687	4.03	7,197	3.86	1.22	0.61
114	622	804	59.5[b]	16.4	41.1	9,497	3.98	11,304	4.05	1.25	0.87
111	592	735	-	14.8	25.4	6,156	3.79	4,762	3.68	1.41	1.07
108	643	619	40.3[a]	13.6	17.8	4,362	3.64	3,829	3.58	1.59	1.23
103	650	406	41.5[b]	11.8	12.8	2,889	3.46	3,225	3.51	1.76	1.03

P_s - Oxygen method, exposed under standard laboratory conditions, 24°C, 400 f.c., converted to $mgC/m^2/day$ by $P_{opt}/hr \cdot \lambda/\varepsilon \cdot hr[(1.26\text{-}1.54)I_o/IK]$
Mean values of 2 to 16 determinations from each station.

P_{max} - *In situ* light-dark bottles oxygen method. $mgC/m^3/hr$. [a] 9/73, 5/74, 8/74; [b] 5/74, 8/74.

Table 17

John H. Kerr Reservoir
Algal Cell Density by Class
Average No/ml - Growing Season April-November
( ) Percent of Total Sample

Station	Cyano.*	Crypto.	Dino.	Hapto.	Chryso.	Bacillario.	Xantho.	Raphido.	Eugleno.	Prasino.	Chloro.	Unidentified Chromophyta	Unidentified Chlorophyta
Roanoke Arm													
24[a]	3085 (30.4)	60 (0.6)	37 (0.4)	0 -	377 (3.7)	2561 (25.2)	33 (0.3)	0 -	228 (2.2)	2 (0.02)	3496 (34.5)	18 (0.2)	250 (2.5)
20[a]	6008 (58.3)	49 (0.5)	94 (0.9)	0 -	12 (0.1)	863 (8.4)	65 (0.6)	19 (0.2)	310 (3.0)	93 (0.9)	2575 (25.0)	12 (0.1)	204 (2.0)
5815[b]	1994 (32.7)	26 (0.4)	31 (0.5)	3 (0.05)	28 (0.5)	857 (14.1)	74 (1.2)	0 -	153 (2.5)	33 (0.5)	2704 (44.4)	4 (0.7)	184 (3.0)
8[a]	650 (27.0)	0 -	34 (1.4)	2 (0.1)	10 (0.4)	454 (18.8)	33 (1.4)	0 -	76 (3.1)	7 (0.3)	923 (38.3)	7 (0.3)	213 (8.8)
2[a]	546 (26.8)	18 (0.8)	27 (1.3)	16 (0.7)	37 (1.8)	578 (28.4)	41 (2.0)	0.25 (0.01)	20 (1.0)	0 -	684 (33.5)	4.5 (0.2)	65 (3.2)
Nutbush Arm													
118[c]	31567 (64.3)	268 (0.5)	149 (0.3)	0 -	0 -	1861 (3.8)	74 (0.2)	0 -	492 (1.0)	121 (0.2)	14107 (28.7)	0 -	472 (1.0)
1308[b]	3490 (26.4)	90 (0.7)	45 (0.3)	21 (0.2)	41 (0.3)	4614 (34.9)	265 (2.0)	0 -	576 (4.4)	29 (0.2)	3669 (27.8)	48 (0.4)	333 (2.5)
114[d]	9050 (62.6)	3 (0.02)	26 (0.2)	129 (0.9)	12 (0.1)	1571 (10.9)	279 (1.9)	0 -	199 (1.4)	92 (0.6)	2991 (20.7)	25 (0.2)	85 (0.6)
111[d]	213 (8.4)	4.5 (0.2)	0.5 (0.02)	81 (3.2)	51 (2.0)	796 (31.5)	102 (4.0)	0 -	75 (2.9)	14 (0.5)	1059 (41.9)	0 -	130 (5.1)
108[c]	1978 (43.8)	0 -	51 (1.1)	34 (0.8)	6 (0.1)	619 (13.7)	142 (3.1)	0.3 (0.01)	78 (1.7)	3 (0.1)	1362 (30.1)	0 -	241 (5.3)
103[d]	820 (29.8)	9 (0.3)	65 (2.4)	12 (0.4)	41 (1.5)	544 (19.8)	65 (2.4)	0.3 (0.01)	64 (2.3)	0 -	959 (34.9)	0 -	170 (6.2)

*Add ending "phyceae" to abbreviated names.
[a]June, Sept. 1972, May, Sept. 1973; [b]April-Oct. 1973; [c]May, Sept. 1973; [d]Sept. 1972, May and Sept. 1973

Table 18

John H. Kerr Reservoir
P and/or N Limiting Conditions as Shown by Algal Assay
March 1972 - May 1974

		Filtered					Autoclaved				
Roanoke Arm	No. of Assays On Samples From Station	Average Biomass of Control, mg/1	Number P Limited	Number N Limited	Number P & N Limited	Number No Response	Average Biomass of Control mg/1	Number P Limited	Number N Limited	Number P & N Limited	Number No Response
Station											
24	6	2.0	1	2	3	0	7.9	0	4	2	0
20	7	3.1	1	1	5	0	5.6	2	3	1	1
8	8	1.0	2	1	2	3	5.4	2	1	5	0
2	8	1.0	4	1	3	0	4.4	4	1	3	0
Nutbush Arm											
118	5	9.0	0	5	0	0	14.4	1	4	0	0
1308	6	1.0	0	1	3	2	10.7	1	4	0	1
114	7	0.6	0	1	5	1	2.9	1	2	4	0
111	2	0.7	1	0	0	1	3.7(1)	1	0	0	0
108	7	0.8	3	0	4	0	1.4(6)	1	0	5	0
103	8	2.0	3	1	4	0	2.6(7)	3	2	4	0

() Number of samples assayed if other than indicated

Table 19

John H. Kerr Reservoir
Total Zooplankton/m^3 - Vertical Net Tows, 1974

Station	January	February	March	April	May	June	July	August	September	October	November	December	Average of April-November Samples	Log of Average
Roanoke Arm														
24	-	-	18,067	-	96,409	-	16,520	23,476	-	-	38,324	-	38,559	4.59
20	-	-	123,664	-	93,194	-	157.353	24,618	-	-	53,480	-	90,462	4.96
14	-	-	109,499	-	297,764	-	138,059	46,320	-	-	45,745	-	127,477	5.11
8	-	-	113,358	-	253,383	-	145,450	43,320	-	-	17,352	-	114,573	5.06
2	-	-	64,676	-	144,052	-	165,990	18,161	-	-	23,408	-	83,257	4.92
Nutbush Arm														
119	185,302	222,482	41,805	277,716	450,562	97,190	+	-	78,392	256,626	-	-	232,097	5.37
118	65,852	238,432	29,991	301,009	90,501	185,017	620,807	47,983	146,799	185,744	59,122	-	204,622	5.31
218	-	-	-	92,802	177,022	38,138	216,226	18,949	203,383	214,659	40,206	-	125,173	5.10
1308	195,120	-	195,213	181,758	245,378	113,907	-	33,339	122,022	126,718	37,620	+	122,963	5.09
116	-	-	-	232,386	103,125	25,310	99,094	30,173	132,280	162,619	25,323	+	101,289	5.01
114	-	-	+	124,054	167,456	43,394	146,146	17,153	91,015	86,785	33,912	+	88,739	4.95
111	-	-	+	-	221,418	-	141,988	41,376	-	-	23,126	-	106,977	5.03
108	-	-	+	-	182,089	-	123,337	40,167	-	-	38,191	-	95,946	4.98
103	-	-	956	-	133,122	-	141,766	3,643	-	-	23,781	-	60,654	4.78

- No sample
+ Qualitative data only

Table 20

John H. Kerr Reservoir
Genera of Zooplankton
Net Tows - 1973-1974

COPEPODS

- Argulus sp.
- Diaptomus spp.
- Misc. Calanoid Adults + copepodids
- Cyclopoid Adults + copepodids
- Harpacticoid Adults
- Nauplii

CLADOCERA

- Alona sp.
- Bosmina longirostris
- Ceriodaphnia sp.
- Daphnia sp.
- Diaphanosoma sp.
- Leptodora kindtii
- Pleuroxus sp.
- Unknown Cladocera, adults and immatures

ROTIFERA

- Ascomorpha sp.
- Asplanchna sp.
- Brachionus sp.
- Collotheca sp.
- Conochiloides sp.
- Conochilus sp.
- Epiphanes sp. (?)
- Filinia sp.
- Gastropus sp.
- Hexarthra sp.
- Kellicottia bostoniensis
- Keratella sp.
- Lecane sp.
- Monostyla sp.
- Ploesoma sp.
- Polyarthra sp
- Proales sp.
- Rotaria sp.
- Synchaeta sp.
- Testudinella sp. (?)
- Trichocerca sp.
- Unknown Flosculariidae
- Unknown Rotifera

PROTOZOA

- Actinosphaerium sp.
- Arcella sp.
- Difflugia sp.
- Epistylis sp.
- Paramecium sp.
- Stentor sp.
- Vorticella sp.

OTHERS

- Chironomidae

Table 21. Major Groups of Benthic Organisms, Density-no./m^2

Date Sampled	August 27-30, 1973					November 21-23, 1973				
Station	Olig.	Chir.	Chao.	Palpo.	Ephe.	Olig.	Chir.	Chao.	Palpo.	Ephe.
Roanoke Arm										
24	194	915	5713	0	0	1356	656	43	0	0
20	742	150	2109	0	0	581	1764	13052	0	0
14	108	11	3045	0	0	129	334	6004	0	0
8	1119	0	140	0	0	1904	140	2744	0	0
2	990	0	11	0	0	1000	452	1323	0	0
Butcher Creek										
214	65	65	3099	43	0	86	1205	2023	65	0
Eastland Creek										
211	65	86	1410	0	0	32	2529	549	11	0
Nutbush Arm										
119*	75	366	3605	43	0	108	452	1851	0	0
118*	0	0	3777	0	0	118	75	2862	11	0
118E	2044	441	990	0	0	8769	10846	118	75	0
1308*	0	0	2561	54	0	32	5918	6693	54	0
1308E	5434	1549	882	22	0	2432	3465	764	430	0
116	75	398	3809	0	0	247	4207	3540	0	11
114	151	247	3519	0	0	215	4369	2787	0	0
114E	1055	915	1840	11	0	3314	2324	22	22	0
111	86	0	1689	0	0	22	1302	4390	0	0
111W	774	968	43	22	0	3809	1528	54	0	0
108	43	21	538	11	0	560	1517	2711	0	0
103	21	0	97	0	0	290	1237	2830	0	0
103W	118	538	151	0	0	2141	1840	22	0	22
Flat Creek										
219	2884	1173	667	22	0	1011	11040	2701	441	0
219S	872	1732	1065	43	0	1302	5326	312	75	0
218	667	861	1356	129	0	1194	6736	3034	151	0
218S	1334	2841	839	54	0	2130	3163	86	43	0

Olig.-Oligochaeta; Chir.-Chironomidae; Chao.-Chaoborus; Palpo.-Palpomyia; Ephe.-Ephemeroptera
*High salinity in center channel; Stations designated E,W, or S are sublittoral locations.

Table 21 (continued)

Date Sampled	February 12-15, 1974					April 27-30, 1974				
Station	Olig.	Chir.	Chao.	Palpo.	Ephe.	Olig.	Chir.	Chao.	Palpo.	Ephe.
Roanoke Arm										
24	2550	1033	441	22	0	1797	2065	258	0	32
20	1625	495	1614	0	0	1325	796	549	0	0
14	194	247	5843	0	0	473	172	3185	0	0
8	1356	54	1797	0	0	1775	43	420	0	22
2	2572	387	968	0	0	775	215	43	0	0
Butcher Creek										
214	0	925	473	0	22	43	387	172	0	11
Eastland Creek										
211	11	1872	355	0	0	118	624	65	43	54
Nutbush Arm										
119*	2292	818	430	22	0	75	22	237	0	0
118*	2518	344	2055	0	0	129	43	516	0	0
118E	13396	1829	32	11	0	6112	2001	118	11	0
1308*	7059	1829	108	11	0	1065	710	2001	0	0
1308E	4455	3637	366	570	0	4164	2841	86	140	22
116	1151	2087	3045	11	0	990	721	1173	65	0
114	592	1614	1592	0	0	215	538	742	11	0
114E	6456	3519	32	54	0	10168	1679	97	129	0
111	54	1399	2475	0	0	107	355	301	0	0
111W	6929	2324	11	11	0	3852	732	11	54	75
108	560	1937	1549	0	0	796	646	366	0	11
103	204	775	1140	0	0	323	355	86	0	0
103W	828	1183	97	54	0	1119	667	32	75	65
Flat Creek										
219	1485	10835	1484	97	0	3111	2281	2141	140	0
219S	2432	4390	258	129	0	2873	3927	54	75	0
218	1904	2636	1894	43	0	2658	1905	409	43	0
218S	5348	6026	54	43	0	4153	1485	32	151	0

Olig.-Oligochaeta; Chir.-Chironomidae; Chao.-Chaoborus; Palpo.-Palpomyia; Ephe.-Ephemeroptera
*High salinity in center channel; Stations designated E,W, or S are sublittoral locations.

Fish

Although the John H. Kerr Reservoir is one of the more popular fishing locations in the Virginia-North Carolina region, quantitative data on the productivity of the fishery is somewhat limited or unavailable at this time. The data of two limited creel censuses from North Carolina waters, primarily Nutbush Arm are as follows:

1964-1965 Census; 385 contacts yielded a gross catch rate of 0.86 fish/hr. -- which 54% were "sunfishes" (other than crappy, bluegill and redbreast sunfish), 30% catfish, 10% crappy, 3% carp, 2% large mouth bass, 1% pickerel and 1% other species.

1970-1971 Census; 413 contacts yielded a gross catch rate of 1.79 fish/hr. -- of which 68.6% were crappy, 15.4% bluegill, 11.3% catfish, 1.5% large mouth bass, 1.3% "sunfishes," 0.6% pickerel, 0.5% carp, 0.4% rough fishes, 0.4% striped bass.*

NUTRIENT BUDGETS

Nitrogen and Phosphorus

The nutrient budget presented in Table 22 is described for the period April 1974-March 1975. Only data collected within this period has been used in this budget. Values estimated include land runoff, non-gaged sources, and the relatively weak concentrations for N and P in rainfall. The Kerr Dam discharge used in this budget, 8859 c.f.s. is based on 365 daily samples whereas the average of the 12 samples taken monthly is 9219 c.f.s. The concentrations of N and P in these 12 samples were used to calculate the discharge (kgs/yr.) using the flow of 8859 c.f.s. The validity of this computation appears to be justified by the budgets computed for Cl^- and $SO_4^=$.

*Creel census data provided by the N. C. Wildlife Resources Commission.

Table 22

John H. Kerr Reservoir

Nutrient Budget - Nitrogen and Phosphorus
April 1974 - March 1975

Source of Flow	Average Discharge c.f.s.		kgs/yr Total-N	kgs/yr Total-P
ROANOKE ARM				
Principal Rivers				
Roanoke	2,725			
Banister	615			
Dan	2,803	6,143	3,612,770	590,205
Three Streams - North Drainage	91		35,040	5,402
Five Streams - South Drainage	475		227,030	23,068
Point Sources Discharges, 7 Municipal and Industrial to Kerr Reservoir or to Flows Downstream of Sampling Points	2.7		85,733	23,411
Total		6,712		
Discharge J. H. Kerr Dam		8,859		
Average Discharge Nutbush Arm All Sources		115		
Net Flow Roanoke Arm (8859-115)		8,744		
Net Flow Non-Gaged Streams and Other Flow - Roanoke Arm (8744-6712) (T-N, .378 mg/l, T-P - .032 mg/l, averages of five non-polluted streams)		2,032	686,565	58,035
Rainfall, 43"/yr/30,866 acres		153		
Evaporation[1]		-153		
N and P Contribution by rain[2,3,4] NO_3; .62 mg/l; Total-P, 0.1 mg/l 43"/yr/30,866 acres			18,556	13,644
Total Roanoke Arm			4,665,694	713,765
NUTBUSH ARM				
Flat Creek	14		5,110	474
Nutbush Creek	6.4		54,750	22,265
Flow Non-Gaged Sources (115-20.4)	94.6		31,967	2,697
Rainfall 43"/yr/12,452 acres)			7,488	5,506
Total Nutbush Arm			99,315	30,942
Total J. H. Kerr Reservoir			4,765,009	744,707
Kerr Dam Discharge			4,026,680	377,045
% Retained			16	50

[1]Yonts and Giese, 1974
[2]Gambell and Fisher, 1966
[3]T-P determined on rainfall samples collected at Chapel Hill, N.C. 13 and 25 April 1972.
[4]Uttormark and Chapin, 1974.

Fe, Cl^- and $SO_4^=$

Utilizing the same flow data used to compute the budget for nitrogen and phosphorus, the pass through for Fe, total, dissolved and particulate and Cl^- and $SO_4^=$ was also calculated, Table 23. The expected large reduction of total and particulate Fe was confirmed and to a somewhat lesser degree the dissolved Fe. The close agreement of both Cl^- and $SO_4^=$ for net pass through endorses the validity of the flow values used.

DISCUSSION

The limnological characteristics of a reservoir are basically defined by the velocity change as the inflowing rivers and streams encounter the standing water of the impoundment. In turn downstream flow through the impoundment and average retention time becomes a function of the relative inflow volume and rate of discharge. This down reservoir movement is also generally over an increasing mean depth since the deepest point and in many instances the maximum surface area of an impoundment is at or adjacent to the dam. The dimensional parameters of mean depth ($\bar{z}$), residence time (τw), flushing rate ($1/\tau w$), retention coefficient (R) and areal loading (qs) for each of the segments of each arm of John H. Kerr Reservoir have been calculated and arranged in Table 24. With these dimensions the associated phosphorus fraction concentrations and loading (L_p, $g/m^2/yr$) are also presented. The changing magnitude of all dimensions in downstream movement and the considerable difference in flushing rate and loading between the two arms of this reservoir provides an opportunity to test the validity of the Vollenweider numbers against the observed trophic state of each compartment (Vollenweider and Dillon, 1974). The relationships of areal loading (qs) versus phosphorus loading is shown in Figure 1 and phosphorus loading and productivity as determined by chlorophyll a production is examined in Figure 2. The differences in behavior of the two arms of John H. Kerr Reservoir is clearly seen with the

Table 23

John H. Kerr Reservoir
Load Budget - Conservative Components Fe, Cl^-, $SO_4^=$
April 1974 - March 1975 (Monthly Samples)

	Average c.f.s.	Total Fe mg/l	Total Fe kg/yr[1]	Dissolved Fe mg/l	Dissolved Fe kg/yr[1]	Particulate Fe mg/l	Particulate Fe kg/yr[1]	Cl^- mg/l	Cl^- kg/yr[1]	$SO_4^=$ mg/l	$SO_4^=$ kg/yr[1]
Roanoke River	2,725	1.96	4,776	.13	317	1.83	4,459	5.1	12,428	6.6	16,083
Banister River	615	2.49	1,369	.43	237	2.06	1,133	3.4	1,870	5.6	3,080
Dan River	2,803	3.29	8,247	.25	626	3.04	7,620	7.3	18,298	7.4	18,549
Station 58-15 Roanoke Arm	-	.90	-	.18	-	.82	-	5.6	-	6.9	-
North Drainage											
Bluestone Creek*	47	1.12	47	.41	17	.71	30	5.1	214	6.9	290
Little Bluestone Creek	19	1.12	19	.27	4.6	.85	14	7.8	133	7.4	126
Butcher Creek*	25	.94	21	.18	4.0	.76	17	5.2	116	4.9	110
South Drainage											
Hyco River	309	1.17	323	.27	75	.90	249	13.6	3,758	12.3	3,399
Aarons Creek*	48	1.00	43	.30	13	.70	30	6.6	283	6.9	296
Grassy Creek*	64	1.44	82	.28	16	1.16	66	4.6	263	6.4	366
Island Creek*	33	1.23	36	.34	10	.89	26	5.4	159	6.6	195
Little Island Creek	21	1.47	28	.27	5.1	1.20	23	4.5	85	14.4	270
Nutbush Arm											
Flat Creek-1326	14	1.22	15	.31	3.8	.91	11	5.6	70	6.6	83
Flat Creek-39	-	.57	-	.12	-	.45	-	11.4	-	7.8	-
Nutbush Creek	6.4	2.71	16	.51	2.9	2.20	13	353.0	2,020	25.7	147
[2]Rainfall (43"/yr./12,452 acres)	153	-	-	-	-	-	-	0.5	68	2.5	342
Other Runoff to Reservoir Non-gaged sources	2,241	1.14*	2,281	.30*	600	.84*	1,680	5.4*	10,804	6.3*	12,606
Total			17,303		1,931		15,371		50,927		55,942
Kerr Dam Discharge	8,859	.63	4,991	.17	1,347	.46	3,644	6.1	48,325	7.4	58,624
% Change			-71		-30		-76		-5		+5

*Ave. concentration in indicated streams used to calculate non-gaged sources.
[1]X10^3 [2]Gambell and Fisher, 1966

Table 24

John H. Kerr Reservoir
Phosphorus Budget Parameters
(Calculations Based on Reservoir Elev. 300 and 8859 cfs Annual Average Discharge for April 1974-March 1975)

Roanoke Arm Segments	Area $m^2 X 10^3$	Mean Depth $\bar{z}$(m)	Residence Time τ_w(yr.)	Flushing Rate $\rho=1/\tau_w$	Retention Coef. R	Areal Loading $(\bar{z}/\tau_w)$ qs	Station	mg/1* PO_4-P	mg/1* Total-P	Flow Into Segment	cfs***	LP $g/m^2/yr$
Stations 24-20	11,975	5.9	.011	191	.231	536	24	.008	.104	Enter	6,503	50.4
20-14	15,706	10.3	.025	40.0	.182	412	20	.005	.073	Enter	7,130	29.6
14-8**	20,720	11.1	.03	33.3	-	370	14	.005	.059	Enter	7,220	18.3
8-4	28,673	11.6	.043	23.2	.233	270	8	.007	.057	Enter	8,565	15.2
4-2	23,776	15.0	.045	22.2	.336	333	4	.007	.043	Enter	8,699	14.0
							2	.006	.028	Leave	8,744	
Nutbush Arm Nutbush Creek								2.62	4.45	Enter	6.4	
Other Flow above 1308								.009	0.34		16.6	22.5
Nutbush Segments Stations							119	.089	.222			
Above 1308	1,355	5.1	.16	6.2	.959	31.8	118	.066	.138			
1308 - Buoy L	6,232	8.1	.76	1.3	.552	10.7	1308 116 114	.014 .008 .005	.097 .052 .033	Enter	46	.64
Buoy L - H	12,882	8.3	1.22	0.81	.019	6.8	111	.006	.022	Enter	74	.17
Buoy H - E	12,525	9.7	1.28	0.78	-	7.6	108	.005	.020	Enter	98	.14
Buoy E - A	12,096	10.5	1.23	0.81	-	8.5	103	.006	.020	Enter	106	.16
										Leave	116	

*Phosphorus fractions are average values of all vertical profile samples at each station.

**Main channel only, Grassy and Butchers Creek volume and area subtracted.
***Non-gaged flow, 2241 cfs, into Roanoke arm allocated to segments by percent of drainage area of each segment.

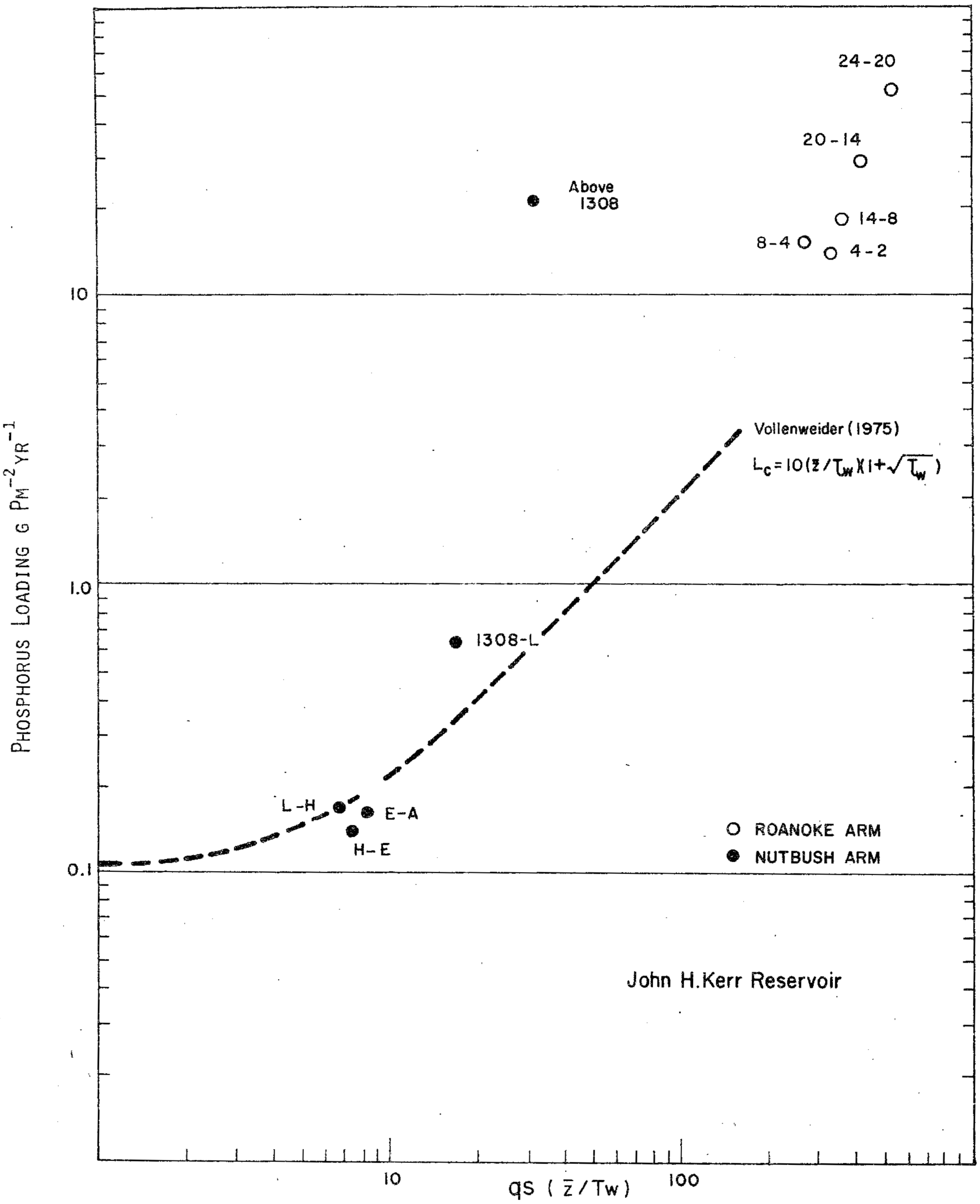

Figure 1. Areal Loading and annual phosphorus load. Dashed line, Vollenweider's boundary condition, acceptable below, unacceptable above.

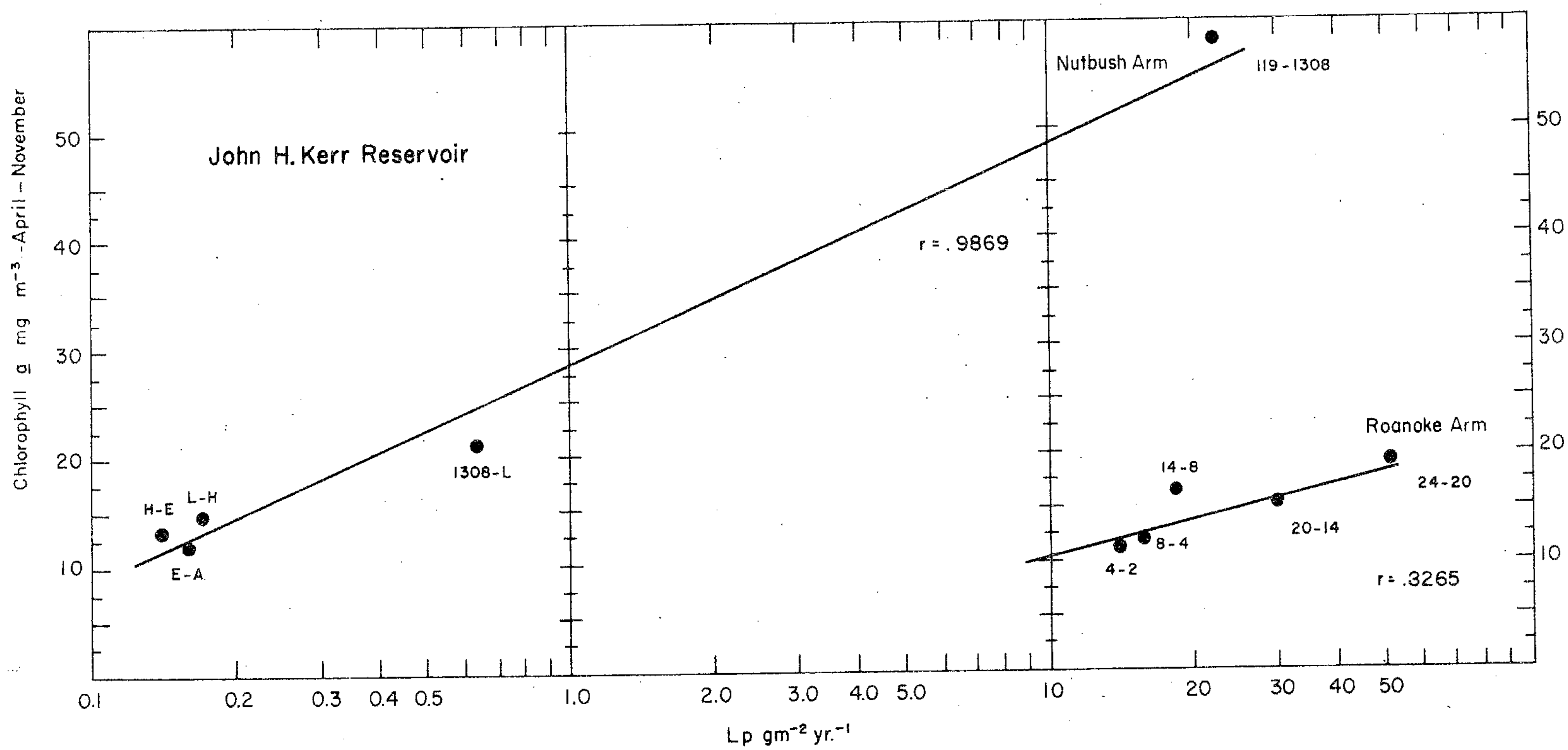

Figure 2. Phosphorus load and chlorophyll a, Roanoke and Nutbush Arms by indicated segment.

flushing rate as a major controlling variable in establishing both the net available phosphorus as well as its rate of utilization. In a final analysis the several independent variables that describe the conditions for algal growth are compared to growth as determined by chlorophyll a and productivity (Ps), Table 25. Again the consistent high correlations of the Nutbush Arm as compared to the low or non-correlations of the Roanoke Arm indicate a major dependency of the system on flushing rate to establish growth limiting conditions.

A preliminary analysis of productivity at secondary levels, total zooplankton numbers and associated algal cell density shows that in the Roanoke Arm the correlation has a r value of -.340 whereas in the Nutbush the r value is .871.

SUMMARY

The John H. Kerr Reservoir, a hydro-power flood control impoundment of 48,900 acres, receives a substantial nutrient load of nitrogen and phosphorus from upstream municipal and industrial waste water discharges. Because of the relative flow into the two major arms of the reservoir, that of the Roanoke being about 80-90 times greater than the Nutbush the residence time of the two arms varies by a factor of 30, 60 days versus 1,800 days. Nutrient budgets for nitrogen and phosphorus indicate for the period April 1974 to March 1975 about 16% of the nitrogen was retained in the impoundment and 50% of the phosphorus. Budgeting for Cl^- and $SO_4^=$, utilizing the same flow values, accounted for essentially the entire load, -4% for Cl^- and +5% for $SO_4^=$.

Examination of P budget parameters and the response of the system of each arm subdivided into five segments verifies with high correlations the validity of areal loading and Total-P as predictive dimensions when retention time is high. At high flushing rates the correlation values are lower. Impoundments with high phosphorus retention coefficients exhibit considerable capacity to

Table 25

John H. Kerr Reservoir
Correlation Coefficients - Growth Controlling Variables

April-November Data

Independent Variables	Dependent Variables: Chlorophyll *a* mg/m^3 R	N	Primary Production $gC/m^2/d$ R	N
Loading $Pg/m^2/yr.$	.836	.986	.757	.989
Total P - Total Profile	.923	.985	.755	.991
Total P - Epilimnion	.608	.975	.419	.965
Total Soluble P - Epilimnion	.561	.953	-.060	.951
Areal Loading	.817	.993	.743	.999
Flushing Rate	.871	.995	.602	.995
Secchi Depth	-.805	-.830	-.077	-.745

R - Roanoke Arm stations

N - Nutbush Arm stations

remove phosphorus from downstream systems. Even with comparatively short residence time, such as the Roanoke Arm, phosphorus removal by adsorption on iron rich sediments may be of considerable magnitude.

ACKNOWLEDGEMENTS

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TROPHIC STATUS AND NUTRIENT LOADING FOR LAKE TAHOE

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I. INTRODUCTION

Lake Tahoe, located in the Sierra Nevada on the California-Nevada border, is a large, deep, ultra-oligotrophic lake which is usually monomictic. The lake, formed in a graben fault, has steep sides, a flat bottom and very little shallow water for its size (Fig. 1). Over 60 tributaries flow into the lake which is drained by one major outflow.

Lake Tahoe is particularly renowned for the great transparency of its water and the beauty of its deep blue color. It is surrounded by high mountains that are covered with snow during several months of the year. These characteristics make the lake basin an ideal place for year-round recreational activities which attract thousands each year. During the last few decades, there has been a dramatic increase in both the resident and tourist population at Tahoe resulting in serious environmental disturbance. By 1962 sewage discharge, even after treatment, was shown to greatly stimulate phytoplankton primary productivity in the nutrient poor Lake Tahoe water. The export of treated effluent from the basin was started shortly thereafter, with completion of most of the sewage diversion process by 1970.

As the population continues to increase in the Tahoe basin with a concomitant rise in construction activities (road building, housing developments), serious damage to the watershed of the lake continues. The exposure of mineral soil to erosion and the resultant leaching of nutrients is a factor in the cultural eutrophication of the lake. Large plumes of sediments extending from tributary streams into the lake and the appearance of luxuriant growths of attached algae around the lake margin in the last ten years were the first clearly visible signs of change in the lake (Goldman 1974).

During the last 16 years Lake Tahoe has been the subject of intensive limnological research with emphasis on the lake's primary productivity, nutrient limiting factors and the process of eutrophication. Since 1972 the research program has expanded into a multidisciplinary research project (supported by NSF-RANN Grant GI-22) with the principal objective being the identification and measurement of the impacts (physical, chemical, biological, and social) of commercial and recreational development of the Lake Tahoe basin. The Ward Creek watershed has been chosen for intensive studies of nutrient flux and sediment transport through the watershed and their impacts of lake water quality.

Primary productivity has proven to be one of the most sensitive indicators of eutrophication in Lake Tahoe (Goldman 1974, Goldman and Amezaga in press). Its level has increased alarmingly over the last 15 years of research.

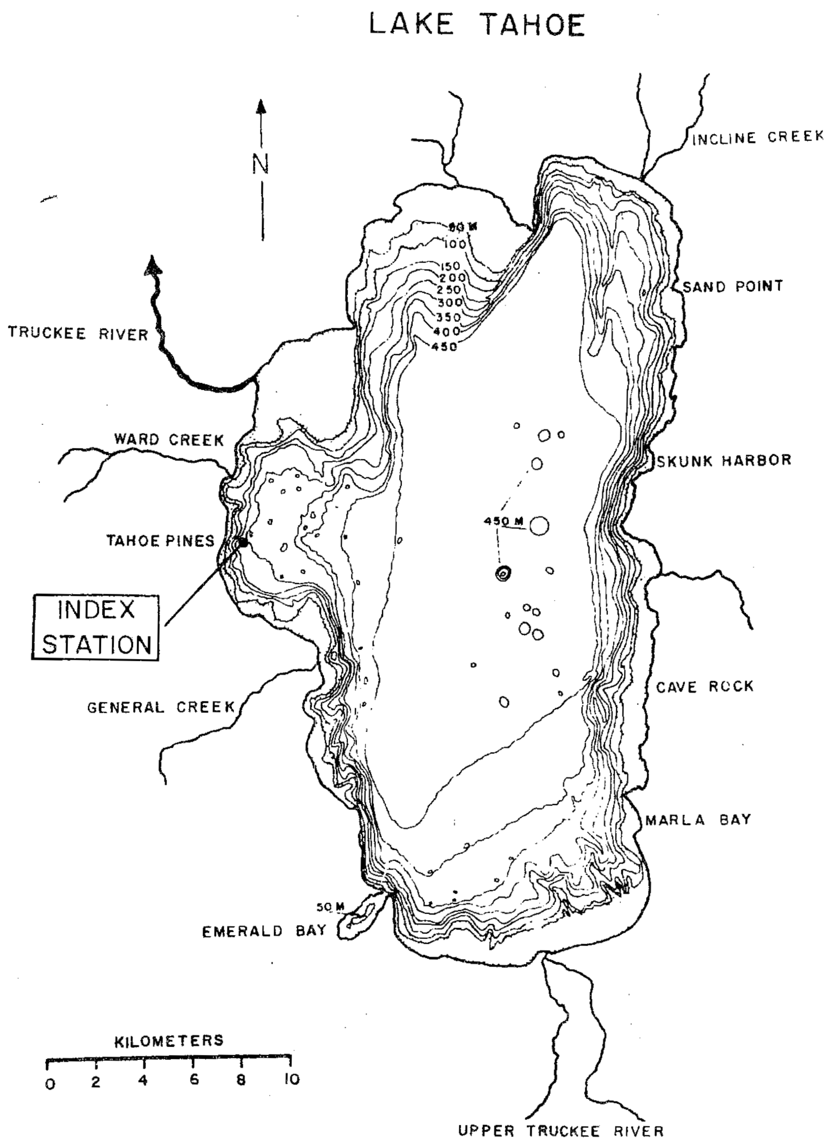

Figure 1. Bathymetric map of Lake Tahoe indicating the index station and other locations. The contour interval is 50 m. The shaded area indicates the littoral zone which extends to 100 m depth. A few tributaries only are shown. (After Goldman 1974)

II. GEOGRAPHIC DESCRIPTION OF WATER BODY

A. LATITUDE AND LONGITUDE

Lake Tahoe is situated in the Sierra Nevada mountains at latitude 39° 06' and longitude 120° 02' (centroid of water area).

B. ALTITUDE OF THE LAKE ABOVE SEA LEVEL

The natural level of the outlet from the lake is 1897 meters and the lake surface level is regulated for the purpose of water storage. A low dam located at the Truckee River outlet serves this purpose and maintains the level between 1897 meters and 1899 meters.

C. CATCHMENT AREA

The total catchment area, including the area of surface water, extends over 1310 square kilometers.

D. GENERAL CLIMATIC DATA

Lake Tahoe never freezes; only some harbors and marinas have periodic ice coverage in the winter. Its large volume of water stores enough summer heat to prevent Lake Tahoe from freezing in the winter.

Average monthly air temperatures for two locations on the shore of the lake are shown in Table 1.

Table 1. Average Air Temperature Data in °C (McGauhey et al. 1963)

	At Tahoe City for 22 year period 1931-52	At Glenbrook for 17 year period 1945-61
January	-3.2	-0.6
February	-2.1	0.2
March	0.2	2.2
April	3.8	5.6
May	7.8	9.2
June	11.7	14.1
July	16.1	18.8
August	15.8	18.8
September	12.4	15.2
October	7.2	9.4
November	1.7	4.4
December	-1.2	0.8
Annual	5.8	7.2

The climate is influenced primarily by marine air masses moving inland from the Pacific Ocean. Continental influence occurs occasionally.

Predominant winds are from the west, southwest or northwest. Within the basin the lake forms an extensive plain which tempers the strong gusty winds usually associated with montane areas.

Evaporation is estimated to average about 90 cm per year.

Evapotranspiration is about 60 cm per year.

E. GENERAL GEOLOGICAL CHARACTERISTICS

The Lake Tahoe basin is bordered on its west side by the main crest of the Sierra Nevada and by the Carson Range on the east. The lake basin is the southernmost of a series of tectonic depressions that form a NNW-trending graben complex extending northward to the area of Mt. Lassen.

Granitic rocks of the Sierra batholith comprise the bedrock of the entire southern half of the basin and along the eastern side as far north as Incline Creek. Extensive flows of principally andesitic volcanic rocks of the Cenozoic age occur at the north end. Little or no granitic rock crops out in the basin west of Crystal Bay with the exception of Stateline Point. Volcanic rocks are predominant on the west side as far south as Blackwood Creek.

Approximately 70% of the runoff in the basin comes from granitic terrain, 25% from volcanic rocks and about 5% from metamorphic rocks (Court, Goldman and Hyne 1972). Sparse protective vegetation in many areas and erodible soils result in appreciable erosion during heavy rain and spring snowmelt. Large sediment plumes extend into the lake during heavy runoff.

F. VEGETATION

The lake shores are forested with coniferous trees. Some meadows exist in the tributary stream valleys with abundant stands of nitorgen fixing alders in many places. Rock exposures and steep slopes near the rim of the basin may be almost devoid of vegetation. Most of the forest is second growth having been extensively lumbered in the late 1800's for mining activity in Nevada. Because of the dry summers and cold winters revegetation is a slow process. Some ski slopes have remained barren for over a decade.

G. POPULATION

The Lake Tahoe region is a recreational area. Precise population estimates are difficult to acquire and soon become obsolete. Various components of total population are present (listed below). In view of the dynamic nature of population in the Tahoe region, a peak seasonal population number is commonly used as the population indicator.

The Tahoe Regional Planning Agency (TRPA) has derived a population estimate from census data and economic activity analysis. The total estimate is 129,700 broken down into the following categories:

Permanent residents	26,100
Seasonal residents	10,000
Second-home residents	32,000
Motel/hotel visitors	32,400
Camper visitors	6,700
	107,200
Estimate day use visitors	22,500
Total	129,700

The total value of 129,700 represents the existing peak seasonal population. (After preliminary draft of Lake Tahoe Study Section 114, PL 92-500, U.S. Environmental Protection Agency, October 1973).

H. LAND USAGE

The Tahoe basin is used extensively for recreation (skiing, gaming, tourism, watersports). Land ownership is:

1. 62% public land* (57% National Forest and 5% State Parks)
2. 38% private land (67% of lake shoreline)

The majority of the developments are second-home subdivisions. Legalized gambling in Nevada has spawned several large casinos at the south and north ends of the lake adjacent to the state line.

*Considerable land is being acquired by governmental agencies.

I. USE OF WATER

Tahoe is operated as a fluctuating reservoir to provide water for downstream users. High water in spring causes shore line erosion and late summer low water may leave some piers high and dry during unusually dry years. There are 34,000 acre feet ($42 \times 10^6 m^3$) allocated for use in the basin (also see B). Approximately 19,000 acre feet are actually being used. Fishing, boating, and some skin diving are recreational uses of the lake. With continued development a water shortage could develop.

J. SEWAGE AND EFFLUENT DISCHARGE

Sewage is collected by conventional sewage lines in most parts of the basin and is given tertiary treatment at the south end of the lake (South Tahoe Public Utility District plant) before being pumped out of the basin for recreation and irrigation. Sewage at the north end of the lake has been pumped into a cinder cone out of the Tahoe drainage after primary treatment. This natural filter is now overloaded and the rate of infiltration into sewer lines may be great. A few septic tanks persist but the majority of sewage effluent is exported. There is no significant industrial effluent discharge, if any, in the basin.

III. MORPHOMETRIC AND HYDROLOGIC DESCRIPTION OF WATER BODY

A. SURFACE AREA OF WATER

Lake Tahoe surface area is 499 km^2. Its maximum length is 34.7 km, its maximum width is 19.2 km. It has an average length of 32.9 km and an average width of 15.4 km. Lake Tahoe's shoreline measures about 113 km (including bays and inlets).

B. VOLUME OF WATER

Lake Tahoe has a volume of 156 km^3 of water. The top 1.86 meters of the lake (elevation 1896.77 m to 1898.63 m), with a volume of about 0.9 km^3 serves as a storage reservoir for the Truckee Carson Irrigation District which operates it on behalf of the United States Government.

C. MAXIMUM AND AVERAGE DEPTH

Lake Tahoe has a maximum depth of 501 meters and an average depth of 313 meters.

D. EXCEPTIONAL DEPTHS AND SURFACE AREA RATIO OF DEEP TO SHALLOW WATERS

The lake basin has steep sides, a flat bottom and very little shallow water for its size (Fig. 1). Several large mounds (about 50 meters high) occur on the floor of Lake Tahoe.

The shallow littoral zone of Lake Tahoe extends to about 100 meters (Goldman and Amezaga 1974). The surface area ratio of deep to shallow water is 4.35.

E. RATIO OF EPI- OVER HYPOLIMNION

The epilimnion of Lake Tahoe extends down to about 15 meters and its hypolimnion is located below 25 meters. This gives Lake Tahoe a ratio of epi- over hypolimnion of about 0.05 (7 km^3/143 km^3).

F. DURATION OF STRATIFICATION

Stratification lasts from 6 to 7 months beginning about May and lasting until November. Complete mixing occurs in late winter if sufficiently high winds and low temperatures persist.

G. NATURE OF LAKE SEDIMENTS

The areal distribution of volcanic constituents of sand and gravel fractions reflects volcanic sources in the north and northwest parts of the basin. Volcanic areas contribute montmorillonite to clay fractions whereas vermiculite and chloritic intergrades are characteristic weathering products of granitic sources.

Two distinct types of sediment are present. Pollen-rich diatomaceous ooze (organic ooze) is characterized by the following: (a) abundant diatoms and pollen; (b) chloritic intergrades in the clay fraction; (c) all samples from flat-lying, well stratified beds. The other sediment type (non-organic) is typified by: (a) diatoms and pollen rare or absent; (b) vermiculite/mica/montmorillonite clay fraction; (c) not present in "flat-lying" beds; (d) texturally more varied than organic ooze.

Non-organic samples represent exposed depositional products of the Tioga glaciation, reflecting relatively rapid erosion and slumping into deeper parts of the basin. The principal source of non-organic material was the west side where volcanic rocks constitute about half of the area. In contrast, organic ooze samples result from relatively passive post-glacial fluvial erosion. The relative abundance of biogenic components in organic ooze reflects low depositional rates and the clay fraction, rich in chloritic intergrades, points to the dominance of granitic source rocks in the present basin-wide source (Court, Goldman and Hyne 1972).

H. SEASONAL VARIATION OF MONTHLY PRECIPITATION

Table 2. Precipitation data in cm (McGauhey et al. 1963)

	At Tahoe City for 43 year period 1910-52	At Glenbrook for 17 year period 1945-61
January	15.39	7.57
February	13.82	6.22
March	9.91	7.16
April	5.28	3.84
May	2.82	3.89
June	1.52	1.02
July	0.66	0.94
August	0.38	0.74
September	1.17	1.22
October	4.24	2.44
November	8.41	5.23
December	13.54	6.25
Annual	77.14	44.45

Precipitation increases dramatically with altitude in the basin. For example, 100 cm of precipitation fell during the 1973 water year and 187 cm during 1974 at 2195 m altitude in Ward Valley.

I. INFLOW AND OUTFLOW OF WATER

The total average inflow of water:

from runoff water is about	0.382 km^3
from precipitation on lake	0.259 km^3
Total inflow	0.641 km^3

The total average outflow of water from the lake:

from evaporation is about	0.410 km^3
from discharge	0.217 km^3
from diversion	0.006 km^3
Total outflow	0.633 km^3

Ground water movements in the basin are for the most part unknown.

J. WATER CURRENTS

Surface and mid-depth currents have been observed to be generally southerly. Bottom currents were found to be generally southeasterly (McGauhey et al. 1963).

Periods of the surface seiches in Lake Tahoe have been determined to be about 19 min. (uninodal seiche) or less (binodal and transverse). Water level records have indicated fluctuations with periods of about 12 and 24 hours and amplitudes of a few millimeters. These fluctuations are possibly surface reflections of internal seiches (McGauhey et al. 1963).

K. WATER RENEWAL TIME

Retention time for Lake Tahoe has been estimated to be about 700 years.

IV. LIMNOLOGICAL CHARACTERIZATION SUMMARY

* Indicates that data on the parameter are (or have been) collected regularly and that the representative values given here have been selected from data covering a period of at least one year.

** Indicates that data on the parameter are (or have been) collected occasionally. However, the values given here are known to be quite representative.

All concentration values given below, for range limits, are actual point measurements in the water column, unless otherwise indicated. They are not mean concentrations for the whole water column. When mean concentrations over a period of time are given, values have been calculated over both depth and time.

A. PHYSICAL

* 1. Temperature range 4.57 - 20.35 ^{o}C (reversing thermometer)
Representative profiles of temperature in the euphotic zone are shown in Fig. 2

** 2. Conductivity range 86.9 - 104.3 μmho/cm 25^{o}C mean = 92 μmhos

* 3. Light penetrates to great depths in Lake Tahoe
Secchi depth range 15.5 - 43.0 m mean for 5 years = 28.3 m
Some representative profiles of light transmission in Lake Tahoe are shown on Fig. 2
Depth of 1% light transmission range 59 - 105 m
Extinction coefficient range 0.044 - 0.078

** 4. Color measurements have been taken on Lake Tahoe waters by Smith, Tyler and Goldman (1973), by using a spectroradiometer to measure absolute values of spectral irradiance and a transmissometer to

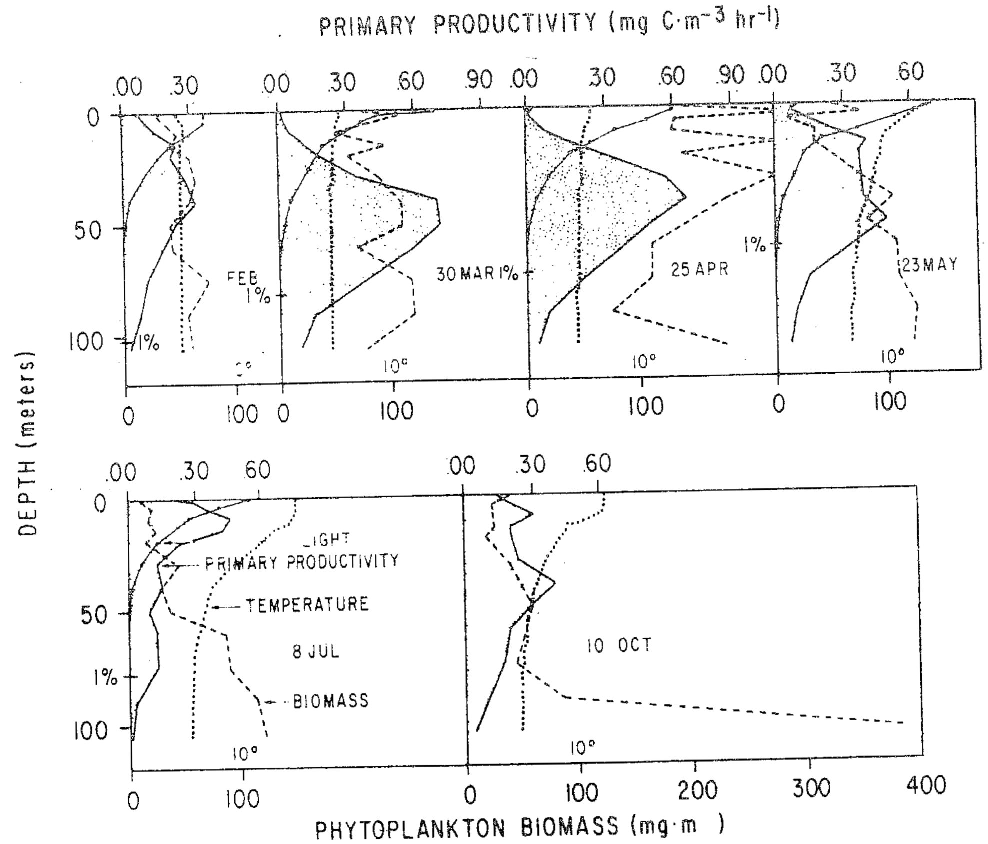

Figure 2. Representative vertical profiles of primary productivity, temperature, light, and phytoplankton biomass at the index station of Lake Tahoe in 1969. (After Goldman and Amezaga 1975)

measure beam transmittance. They measured the spectral composition of the radiant energy up- and down-welling from lake waters and evaluated the color attributed to lake waters in terms of the C.I.E. chromaticity coordinates. They reported the tristimulus values and showed the plot of the tristimulus values on the C.I.E. chromaticity diagram which gives the numerical specification of the colors.

* 5. Solar radiation range: 20-750 $cal \cdot cm^{-2} \cdot day^{-1}$ Total = 150,863 $cal \cdot cm^{-2} \cdot year^{-1}$ (average of 4 years)

B. CHEMICAL

** 1. pH range 7.3 - 8.0

* 2. Dissolved oxygen range 6.85 - 11.57 mg/l
There is essentially no oxygen depletion in Lake Tahoe.

* 3. Total phosphorus range:

epilimnion (0-15 m)	0.7 - 20.4 μg/l	with mean = 2.8 μg/l for growing season
euphotic zone (0-105 m)	N-D - 25.0 μg/l	(N-D = non detectable)
whole lake (0-400 m)	N-D - 25.0 μg/l	with mean = 3 μg/l

We have not been able to measure satisfactorily the fractions of phosphorus which are present in very small amounts. The mean dissolved phosphorus value is less than 5 μg/l.

4. Total nitrogen

* NO_3-N range:

epilimnion (0-15 m)	N-D - 25 μg/l	with mean = 4.3 μg/l for growing season
euphotic zone (0-105 m)	N-D - 26 μg/l	with mean = 7 μg/l (one year)
whole lake (0-400 m)	N-D - 26 μg/l	with mean = 13 μg/l (one year)

Other forms of nitrogen have not been measured on a regular schedule in the past in Lake Tahoe. Based on some past measurements, however, an average concentration has been estimated (McGauhey et al. 1963). These values are:

Nitrate as N	8 μg/l	Nitrite as N	1 μg/l
Ammonia as N	2 μg/l	Organic nitrogen as N	50 μg/l

There is considerable variation in nitrate during the year with depletion occurring in the euphotic zone.

** 5. Alkalinity range 40 - 45 mg/l with mean = 43 mg/l

6**Ca^{++} range: 8.8 - 9.9 mg/l	**Mg^{+} range: 2.1 - 2.9 mg/l
**Na^{+} range: 5.8 - 7.0 mg/l	**K^{+} range: 1.6 - 1.8 mg/l
**$SO_4^{=}$ range: 1.5 - 3.6 mg/l	**Cl^{-} range: 1.7 - 2.1 mg/l
*Fe^{+} range: N-D-126 μg/l	

** 7. Trace metals. Where levels were below detection, the analytical limit is indicated (e.g. Beryllium <0.3). Values are in micrograms per liter.

Aluminum	16	Chromium	<0.07	Molybdenum	0.51
Beryllium	<0.3	Copper	trace	Nickel	<0.3
Bismuth	<0.3	Gallium	<2.8	Lead	<0.06
Cadmium	<0.7	Germanium	<0.3	Titanium	<0.6
Cobalt	<0.6	Manganese	2.6	Vanadium	15
				Zinc	<14

C. BIOLOGICAL

1. Phytoplankton

Representative profiles of phytoplankton biomass and productivity are shown in Fig. 2.

* a. chlorophyll a ranges (unit is mg/m^3)

epilimnion (0-15 m)	9.06 - 0.31	mean = 0.18 for growing season
euphotic zone (0-105 m)	0.03 - 1.25	mean = 0.275 for year round
whole lake (0-400 m)	0.01 - 1.49	

Maximum average concentration for the euphotic zone = 0.59 mg/m^3.
Maximum average concentration for the whole lake = 0.39 mg/m^3.

* b. primary productivity (expressed as carbon)
annual productivity = 55 g m^{-2} $year^{-1}$ (average of 6 years)
epilimnion (0-15 m) range 8.79 - 76.46 $mg \cdot day^{-1} \cdot m^{-2}$ (of epilimnion)
average = 38 $mg \cdot day^{-1} \cdot m^{-2}$ for growing season
euphotic zone range 42 - 322 $mg \cdot m^{-2} \cdot day^{-1}$
average 150 $mg \cdot m^{-2} \cdot day^{-1}$ (for 6 years)

Note on Fig. 2 that the euphotic zone extends well below the depth at which 1% of surface light is transmitted.

c. Algal growth is limited primarily by nitrogen and iron. EDTA or NTA additions can effectively stimulate primary productivity through chelation. Phosphorus is stimulating only when additional nitrate is provided.

* d. There are over 160 species of phytoplankton in Lake Tahoe, 112 of which are diatoms. Only 10 are centric forms. One of the dominant diatoms of the late 1960's *Cyclotella bodanica* has been replaced by *Cyclotella stelligera*. Other dominant species include *Dinobryon sertularia*, *Fragilaria crotonensis* and *Melosira crenulata*. As few as three species often account for 80% of the total phytoplankton biomass. A large number of very small ~3 μ forms are now abundant at about 90 m in summer. There taxonomic status is still uncertain.

Phytoplankton fresh weight biomass = 8.3 $g \cdot m^{-2}$ of the euphotic zone for an average day (average of 2 years).

The average number of cell concentration is about 100 cells per milliliter.

Duration of a "bloom" at Lake Tahoe is about 3 to 4 months. The time of its occurrence is highly variable from one year to the next, and it would not be considered a bloom by most observers.

* 2. Zooplankton

There is an average of about 1000 zooplankters per cubic meter in Lake Tahoe. The community is composed of about 25 species of crustaceans and 14 species of rotifers. It is dominated by the rotifers *Kellicottia longispina*, *Ascomorpha* and *Chromogaster* and the copepods *Epischura nevadensis* and *Diaptomus tyrrelli*. The major pelagic cladocerans *Daphnia* and *Bosmina* that were found often in Lake Tahoe have almost completely disappeared in recent years, perhaps from predation by the introduced *Mysis relicta* (Richards et al. in press).

3. Bottom Fauna

The most abundant benthic animals are sculpins and the California crayfish *Pacifastacus leniusculus*. There are an estimated 56 million adult crayfish in the lake (Abrahamsson and Goldman 1970). Aquatic oligochaets and an endemic stone fly are also abundant in some areas of the lake.

4. Fish

The fish fauna is largely composed of exotic species. These include the lake trout, rainbow and brown trout as well as kokanee. Sculpins, suckers, and dace make up the rest of the fauna.

5. Bacteria

Bacteria are the usual pseudomonad varieties which are often associated with detrital particles (see Paerl and Goldman 1972). Measurements of primary productivity in Tahoe together with measures of heterotrophy remain the most important and sensitive indicators of eutrophication in the lake.

6. Bottom Flora

Aquatic mosses, attached diatoms and chara make up most of the benthic flora which grows to a depth of 100 meters (Goldman and Amezaga 1974).

7. Macrophytes

Some pond weeds are to be found in marinas and protected areas. Higher aquatic plants are for the most part absent in the lake.

V. NUTRIENT BUDGETS

Computations of nutrient budgets were based on the following:

Source from land runoff:

1. Total monthly water discharge calculated from daily measurements taken by the U.S. Geological Survey on nine major tributaries of Lake Tahoe.

2. The estimated yearly runoff of each of the other 54 creeks and tributaries of Lake Tahoe (see Table 3-XV in McGauhey et al. 1963). The total monthly water discharge was estimated for these 54 creeks from this yearly runoff and the measured discharge of the nine major tributaries.

3. Chemistry data collected on nine major tributaries by four different organizations. The major data source was the Tahoe Research Group of the University of California at Davis. Other groups were: The California-Nevada Federal Joint Water Quality Investigation, Lake Tahoe Area Council, Water Resources Information Series of the State of Nevada.

All chemistry concentration data were integrated monthly to get a mean monthly concentration of nutrient per creek.

Creeks for which data for a specific month had not been collected were assumed to have a concentration of nutrients equal to the average concentration of the other creeks that month.

For every creek the total amount of each nutrient that was discharged into the lake was calculated for each month by multiplying the total flow data by the mean concentration. All creeks total discharge of nutrient were summed by month and all monthly values summed to obtain the total load of nutrients entering Lake Tahoe in one year from land runoff. This was done using 1969 data.

Precipitation

1. Average estimate total precipitation on the lake surface.

2. Measurements of average ammonium-nitrogen and nitrate-nitrogen were made in the Lake Tahoe watershed (Coats, Leonard, Fujita and Goldman in prep.) and various estimates have been made of total nitrogen content of rain waters. Only traces of phosphorus are assumed to be present in the precipitation.

Groundwater

According to the water balance ground water does not contribute any significant amount of water to the lake.

Waste Discharge

Waste is being diverted out of the basin. Information on seepage is not available, although exfiltration from sewer lines may be important. High nitrate runoff is still occuring from a temporary land disposal site at South Tahoe (Perkins et al. in press).

A. PHOSPHORUS

Source	$kg \cdot year^{-1}$	$g \cdot m^{-2} \cdot year^{-1}$
Land runoff	23,404	0.047
Precipitation	trace	---
Total	23,404	0.047

B. NITROGEN ($kg \cdot year^{-1}$)

	Land Runoff	Precipitation*	Total
NO_3-N	21,832	20,116	41,948
Organic N	104,645	no information	
NH_3-N	24,480	7,165	31,645
NO_2-N	2,731	no information	
Total Nitrogen	153,688	104,000	257,288

Total nitrogen surface area loading = 0.5156 $g \cdot m^{-2} \cdot year^{-1}$.

*A recent estimate for (NO_3-N + NH_3-N) in precipitation was obtained using the water-year 1973-1974. The new value is 49,900 $kg \cdot year^{-1}$. Preliminary results were obtained recently on measurements of organic nitrogen. These new results confirm that our preliminary value of 104,000 $kg \cdot year^{-1}$ is a very reasonable estimate of total nitrogen loading from precipitation, although it was not obtained by direct measurements.

VI. DISCUSSION

Lake Tahoe remains a classic example of a subalpine, ultraoligotrophic lake whose remarkable clarity gives record Secchi readings to forty meters. Oxygen shows no measurable depletion, even at depths of 500 meters, and the dilute rain of organic matter into the abyssal zone is almost completely mineralized before it reaches the sediment. The lake, at present nitrogen levels, is rather insensitive to phosphorus and can be considered a classic example of a nitrogen limited system. It would appear to be highly sensitive to nitrogen loading and the increased loading that has certainly accompanied the development of the basin has caused an increase in primary productivity of about five percent per year.

VII. SUMMARY

Because of its relatively small watershed and great volume Lake Tahoe is at the extreme lower end of lakes classified on the basis of loading. In all probability this also makes it one of the lakes most sensitive to nutrient loading. Some confirmation of this is seen from measures of primary productivity during the last 15 years (Fig. 3). The rate of increase appears to have peaked out during the last two or three years, perhaps in response to the extensive sewage diversion from the lake. The loss of _Daphnia_ and _Bosminia_ as dominant zooplankters and the increase of ultra plankton at the lower level of the euphotic zone is of great interest.

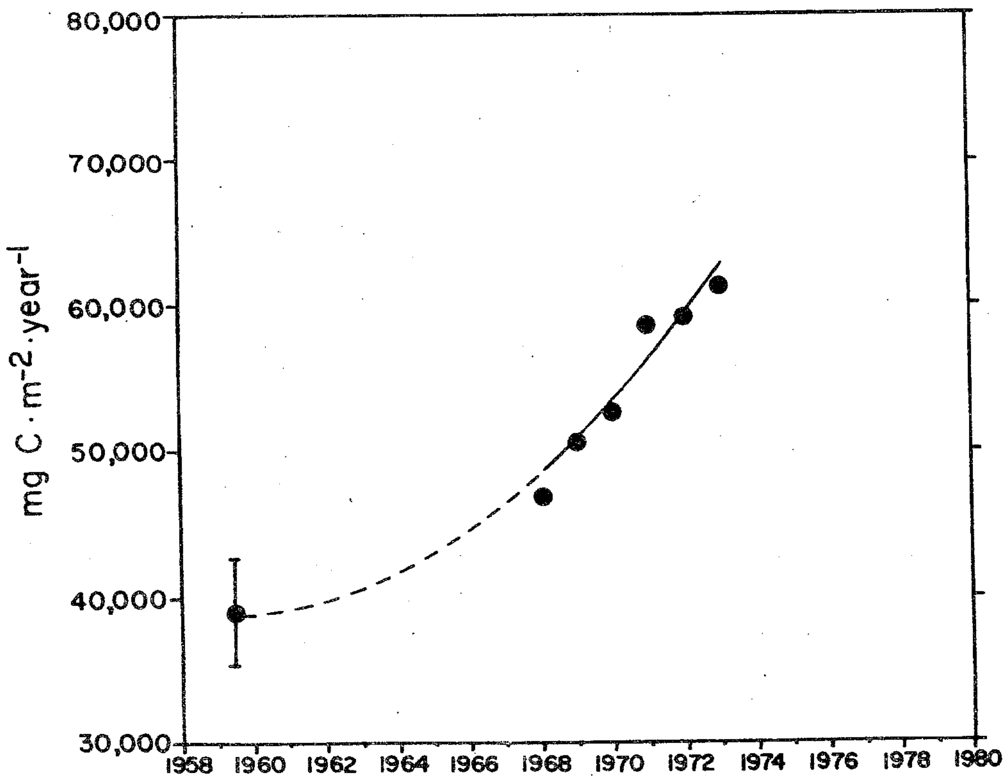

Figure 3. Annual primary productivity at Lake Tahoe between 1959-60 and 1973. Preliminary results indicate that the 1974 value is very close to the 1973 value for primary productivity.

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REPORT ON NUTRIENT LOAD - EUTROPHICATION RESPONSE FOR THE OPEN WATERS OF LAKE MICHIGAN

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INTRODUCTION

Concern over the potential overfertilization of the waters of Lake Michigan prompted the Water Pollution Control Administration (now the Environmental Protection Agency) and the states bordering on the lake to take action. They adopted regulations in 1968 that sought to reduce the phosphorus input from waste treatment plants by 80 percent by December 1972. In addition, the United States and Canada have reached an agreement to reduce effluent phosphorus concentrations to 1.0 mg/l for waters entering Lakes Ontario and Erie. It is conceivable that this requirement might also eventually apply to Lake Michigan. This paper discusses the effects on loading that the reduction in effluent phosphorus has produced. It also discusses the implications of this reduction on water quality in the open waters of Lake Michigan.

PHOSPHORUS LOADING ESTIMATES

Lee (1974) compiled the phosphorus loadings to Lake Michigan in 1971 using a report by the Phosphorus Technical Committee to the Lake Michigan Conference (Zar, 1972). The total estimated loading of phosphorus from all sources was 18.1 million pounds per year (Table 1). This is somewhat higher than the values estimated by Bartsch (1968), by US EPA report (1971), and by the Region V Office of the Environmental Protection Agency (Zar, 1972). However, these latter estimates probably do not include storm sewer overflow or direct precipitation and dry fallout contributions (see Table 1).

Lee (1974) also included predicted phosphorus loading to the lake for 1973 which incorporated the 80 percent reduction in phosphorus by waste treatment facilities in the basin that was agreed to in the late 1960's by the states bordering on Lake Michigan. This value, included in Table 2, assumes 13.2 million pounds of phosphorus yields 2.6 million pounds per year of phosphorus, which leads to the 7.5 million pounds per year total loading of phosphorus shown in Table 2. This predicted phosphorus loading of 7.5 million pounds per year is expected to be reached by approximately 1976-77, assuming the projected goal of 80 percent removal of phosphorus from domestic wastewaters is attained. This would place Lake Michigan in an oligotrophic category (Figure 1) relative to

Table 1. ESTIMATED PHOSPHORUS LOAD TO LAKE MICHIGAN, 1971[1]

Source	Load (Million lbs/yr)
Direct wastewater	3.9
Indirect wastewater	9.3
Total wastewater	13.2
Erosion and other diffuse sources	3.0 (1 to 7)
Combined sewer overflow	0.8
Precipitation and dustfall on surface of lake	1.1
Total diffuse source	4.9
Total	18.1
Bartsch 1968 Estimate	14.6
US EPA 1971 Estimate	14.3
Zar 1971 Estimate	16.7

[1]After Lee (1974).

Table 2. ESTIMATED PHOSPHORUS LOADING
TO LAKE MICHIGAN, 1974

Source	Load (Million lbs/yr)
Wisconsin contribution[1,2]	4.5
Michigan contribution[3]	4.7
Indiana*[4]	0.6
Illinois**[5]	0.7
Combined sewer overflow[6]	0.8
Precipitation and dustfall[6]	1.1
Total	12.4
Lee Estimate for 1973[6] (assumes 80 percent P removal from domestic wastewaters)	7.5
IJC Goal for 1973[7]	11.7

[1,2] From Schraufnagel (1974) and Wisconsin DNR report (1973).

[3] From McCracken (1974).

[4] From Miller (1971).

[5] From US EPA (1971).

[6] From Lee (1974). Based on 80 percent removal.

[7] From Great Lakes Water Quality Board (1973).

* Tributary input of phosphorus into Lake Michigan, 1970 data.

** Represents 1971 data.

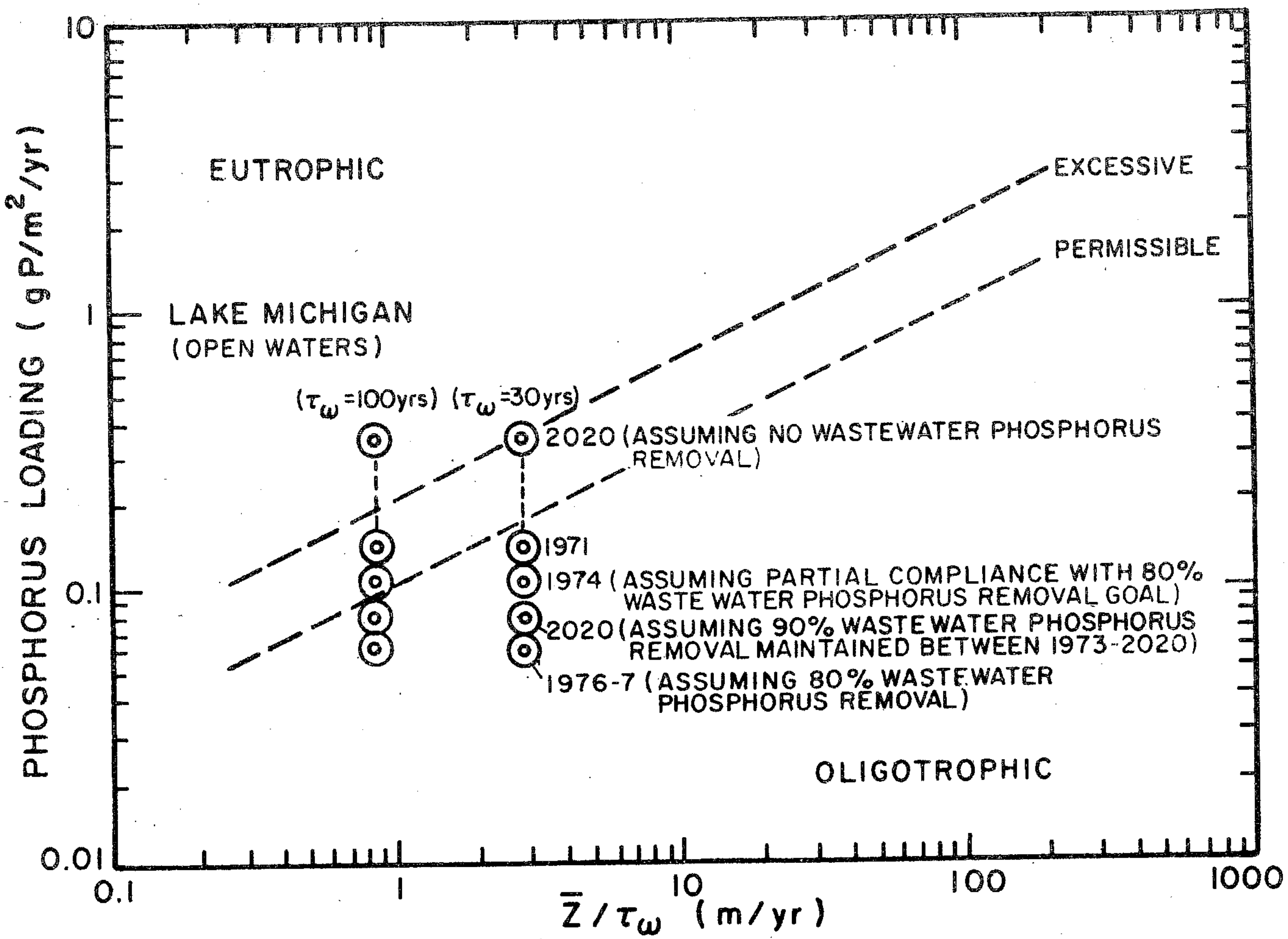

Figure 1. Vollenweider Loading Relationship

Vollenweider's (1975) loading criteria (the effects of different hydraulic residence times in assessing the trophic status of Lake Michigan is discussed in a following section). Vollenweider (1975) has used this relationship (Figure 1) to indicate the relative trophic status of a large number of lakes based on phosphorus loadings and mean depth and hydraulic residence time.

Table 2 also includes the best available estimates for phosphorus loadings to Lake Michigan as of January, 1974. The pollution control agency of each of the states bordering on Lake Michigan was contacted for updated information on nutrient sources. These values are based on information provided by each state. Values for Michigan and Wisconsin reflect improved phosphorus removal by sewage treatment plants by the end of 1973. Current values for Indiana and Illinois were not available. The values for these states presented in Table 2 are probably somewhat high.

If one assumes that the decrease in phosphorus loading from 1971 to 1973 was due entirely to improved sewage treatment plant phosphate removal, then the 1973 estimates reflect a 43 percent reduction in this source of phosphorus, based on Lee's (1974) estimates.

Without any phosphorus removal from domestic wastewaters or a change from phosphorus to nonphosphorus-type detergents, it would be expected that approximately 30 million pounds of phosphorus per year would be entering the lake by the year 2020 over what was expected to enter Lake Michigan in 1973. This would make the 2020 loading approximately 42.2 million pounds of phosphorus annually. This would place Lake Michigan in a eutrophic or eutrophic-mesotrophic status, depending on the hydraulic residence time used in calculation of the $\bar{z}/\tau_{\omega}$ term in the Vollenweider diagram, relative to its phosphorus loading (Figure 1). However, since the wastewaters will be treated for at least 80-90 percent phosphorus removal, only a 2.4 million pound increase per year (assuming 90 percent removal) in the phosphorus loading from the sewered population is expected by the year 2020.

This will be countered by a combination of diversion of sewage and elimination of combined sewer overflow (Table 3a) with the result that the new change in phosphorus loading to Lake Michigan between 1973 and 2020 is expected to be only about 2.1 million pounds per year (Lee, 1974). This is a total phosphorus loading to Lake Michigan of 9.6 million pounds per year by the year 2020. This would leave Lake Michigan in an oligotrophic status relative to the Vollenweider (1975) plot (figure 1), regardless of whether 30 or 100 years is taken as the hydraulic residence time, even with the 2.1 million pound per year increase in phosphorus load.

The nitrogen loading to the lake was estimated by Bartsch (1968) to be about 166 million pounds per year. It is not expected that the nitrogen loadings would have changed much since 1968.

PROBLEMS IN ESTIMATING NUTRIENT LOADS

The estimates presented in Tables 1 and 2 are necessarily based on a number of assumptions. Much of the wastewater data is calculated using 3.6 pounds of phosphorus per person per year. For direct wastewater sources (i.e., discharge directly into the lake), this value is probably quite good, but for indirect sources (i.e., discharge to a tributary to the lake), it is probably too high. No attempt was made to determine what percentage of the phosphorus from indirect wastewater sources is actually reaching the lake in an available form.

Diffuse source estimates were based on an average contribution per square mile of watershed area, usually about 100 pounds of phosphorus per square mile per year. These assumptions, although the best available at this time, result in considerable uncertainty in the loading estimates.

Table 3a. EXPECTED CHANGES IN PHOSPHORUS LOADING OF LAKE MICHIGAN, 1973-2020[a]

Factors Influencing Future Phosphorus Load	Change in Load (Million lbs/yr)
Diversion of North Shore Sanitary District	-0.1
Eliminate Combined Sewer Overflow	-0.8
Increase in Sewered Population (90 percent phosphorus removal by year 2020)	+2.4
Increased Urban Area (conversion of rural to urban land)	+0.6
Rural Runoff Input Reduction	magnitude unknown
Urban Runoff Input Reduction	magnitude unknown
Improvement in Advanced Waste Treatment	magnitude unknown
Net Change in Phosphorus Load to Lake Michigan, 1973-2020	+2.1

[a]From Lee (1974)

PROBLEMS IN ESTIMATING THE HYDRAULIC RESIDENCE TIME FOR LAKE MICHIGAN

For the purposes of this discussion, and as it is used in the Vollenweider phosphorus loading diagram (Figure 1), the hydraulic residence time is defined as the water body volume/annual inflow volume. Thus, it constitutes the water body's "filling time." If the annual precipitation onto the water body surface was approximately equal to its annual evaporation, the annual outflow volume could be used in the same manner as the inflow volume. There are advantages to both methods. The necessity of having to account for precipitation and evaporation in calculation of the hydraulic residence time is avoided if the inflow volume is used. However, the inflow to a water body is frequently through numerous tributary inputs, as well as from runoff directly into the water body and precipitation directly onto the water body surface. It is usually difficult to measure accurately all such inflows to a water body. By contrast, the outflow for most water bodies is usually through a single outlet, allowing it to be more easily measured. The outlet is often gaged and, therefore, the computed total outflow is usually more accurate than the total inflow. Since several methods were possible, it was decided early in the US OECD study that the hydraulic residence time of the US OECD water bodies would be determined on the basis of their annual inflow volumes (Jaworski, 1974).

For Lake Michigan, an examination of the literature indicates there is considerable confusion concerning its hydraulic residence time. Most investigators used the outflow or discharge volume rather than the inflow volume to calculate Lake Michigan's hydraulic residence time. Because precipitation is approximately equal to evaporation, calculation of the hydraulic residence time using the outflow volume will give a reasonable estimate of its magnitude. A summary of the reported hydraulic residence times for Lake Michigan is presented in Table 3b. In some cases, the hydraulic residence time was stated by the investigators, while

Table 3b. SUMMARY OF HYDRAULIC RESIDENCE TIMES REPORTED FOR LAKE MICHIGAN

Hydraulic Residence Time (yr)	Source of Information
99.4	Beeton and Chandler (1963)
30.8	Rainey (1967)
31.2	Patalas (1972)
	Vollenweider and Dillon (1974)
31.2	a) Table 1
100	b) Figure 2
94-113	Vollenweider (1975, 1976, 1977a)
30	Watson (1976)
105	International Joint Commission (1976)
30	Sonzogni _et al_. (1976)
100	Schelske (1977)
100	Bennett (1977)

in other cases it was calculated by these reviewers based on the data presented by the indicated sources. Examination of Table 3b indicates that, with one exception, the hydraulic residence time estimates for Lake Michigan aggregate around the two values of 30 and 100 years, depending on the source of the data. There are several reasons for this three-fold difference in the hydraulic residence time estimates. One reason is that previous investigators have not adequately defined their terms. They did not clearly indicate whether annual inflow or outflow volumes were used in their calculations. However, this factor alone is not sufficient to account for the large differences in the reported hydraulic residence times for Lake Michigan.

Recently, Quinn (1977) has conducted a study of the annual and seasonal flow variations through the Straits of Mackinac from Lake Michigan to Lake Huron. While Lakes Michigan and Huron have historically been treated as a single body of water in hydraulic and hydrologic studies, Quinn has indicated that the actual water mass transport between these two lakes has generally been ignored. Consequently, Quinn developed a water mass continuity technique which he applied to Lake Michigan for the 1950-1966 period to determine average annual and monthly flows through the Straits of Mackinac. His model indicated a 500+ percent variation between maximum and minimum annual flows through the straits during the 17-year period. He also compared his predicted flows with the results of a direct current measurement of flow through the straits for a 100-day period in 1973 and found the results agreed within 2 percent. Using his technique, the annual variations and seasonal cycle of the flow through the straits were quantified. Based on his study, Quinn has determined that calculation of two different hydraulic residence times is possible for Lake Michigan. If the annual mean flow, with no regard for seasonal variations, is used in the computation of the hydraulic residence time, Quinn obtains a value of 137 years. However, Quinn also found that there is a deep return flow of water into Lake Michigan through the straits during stratification. If this return flow is considered in the computations as part of the annual inflow water volume, a hydraulic residence time of 69 years is obtained. This value of 69 years falls approximately in the middle of the 30-100 year range reported in Table 3b.

There is mixing of this "backflow water" with the water at the upper end of Lake Michigan, although the extent of this mixing is not known. It is likely that the backflow has limited effect on the waters of lower Lake Michigan, but it influences the discharge through the Straits of Mackinac into Lake Huron. Because of the uncertainty concerning the correct value, both the 30 and 100-year hydraulic residence time values were used in calculation of the mean depth/hydraulic residence time term (i.e., $\bar{z}/\tau_{\omega}$) in the Vollenweider phosphorus loading diagram (Figure 1). A value between these two extremes is likely the correct hydraulic residence time for Lake Michigan (e.g., Quinn's (1977) value of 69 years). Consequently, 30-100 years can be used as a range of the hydraulic residence times, depending on the actual outflow volume of Lake Michigan during a given year. The use of 100 years in the $\bar{z}/\tau_{\omega}$ expression produces a value of 0.84 m/yr, while 30 years produces a $\bar{z}/\tau_{\omega}$ value of 2.8 m/yr. However, it should be noted that while 30-100 years was used as a range for the hydraulic residence time values in this report, based on the work of Quinn (1977), a range of 70-100 years is likely a more realistic estimate of the present hydraulic residence time for Lake Michigan.

The effect of these two hydraulic residence time values (i.e., 30 and 100 years) on the relative position of Lake Michigan on the Vollenweider diagram can be seen in Figure 1. A τ_{ω} value of 100 years indicates Lake Michigan was in the mesotrophic zone of the Vollenweider diagram, based on its 1971 phosphorus load, and is approximately at the oligotrophic-mesotrophic boundary, based on its 1974 phosphorus load. This characterization of Lake Michigan is reasonable for its nearshore water zones, but is not indicative of the open water trophic conditions of the lake, which are generally considered as oligotrophic. A τ_{ω} value of

30 years indicates Lake Michigan plots at the oligotrophic-mesotrophic boundary in 1971 and in the oligotrophic zone of the Vollenweider diagram in 1974. This oligotrophic characterization is accurate for Lake Michigan's open waters, but does not describe its nearshore zones, which are in a relatively more productive condition than its open waters. Thus, the effects of the two τ_ω values make a difference in delineation of Lake Michigan's predicted trophic condition, as indicated by its position on the Vollenweider phosphorus loading diagram (Figure 1). The lower 1974 phosphorus load to Lake Michigan, relative to its 1971 load, implies an improvement in its water quality, as indicated by its more oligotrophic position on the Vollenweider diagram (Figure 1). Such an improvement is likely when Lake Michigan has reached a new equilibrium condition relative to its reduced phosphorus loading (Sonzogni et al., 1976).

NUTRIENT LOADS AND PRODUCTIVITY IN LAKE MICHIGAN

Schelske and Callendar (1970) surveyed the phytoplankton productivity and the nutrient levels of Lakes Michigan and Superior during the summer of 1969. Table 4 contrasts data for nutrient and productivity parameters for the two lakes. Productivity, as measured by carbon fixation, is about eight times greater in Lake Michigan than in Lake Superior. Conversely, SiO_2 concentrations are considerably lower in Lake Michigan because of the larger diatom population. Schelske and Callendar (1970) suggest that the large difference in SiO_2 concentrations between surface and bottom waters in Lake Michigan also indicates a substantial diatom population. NO_3^--N is higher in Superior and shows little concentration change down the water column. NH_4^+-N and ortho-$PO_4^{\equiv}$-P show no apparent correlations but are included in Table 4 to facilitate comparison. A summary of nutrient loadings and productivity characteristics is presented in Table 5.

Table 4. NUTRIENT AND PRODUCTIVITY PARAMETERS FOR LAKES SUPERIOR AND MICHIGAN, 1969[1]

Lake	Carbon Fixation $mg\text{-}C/M^3/hr$	SiO_2, mg/l	NO_3^--N, μg/l	NH_4^+-N, μg/l	Ortho-$PO_4^{\equiv}$-P μg/l
Northern Michigan					
Surface	3.06±1.29 (16)	0.26±0.07 (18)	116±7 (17)	16.7 (18)	0.7 (18)
Bottom	--	1.26±0.16 (12)	216±36 (12)	16.0 (12)	1.2 (12)
Southern Michigan					
Surface	3.55±2.32 (14)	0.15±0.07 (16)	101±18 (15)	10.2 (15)	0.9 (16)
Bottom	--	1.63±0.49 (12)	218±29 (12)	23.9 (15)	1.9 (16)
Superior					
Surface	0.39±0.11 (16)	1.87±0.11 (20)	269±21 (19)	4.2 (22)	0.5 (22)
Bottom	--	2.01±0.14 (22)	276±16 (21)	16.5 (22)	1.5 (21)

[1]After Schelske and Callender (1970).

NOTE: Parentheses designate number of samples.

Table 5. LAKE MICHIGAN SUMMARY OF PRODUCTIVITY AND NUTRIENT LOADING CHARACTERISTICS

Loading values	
Phosphorus[1] (total)	0.1 $g/m^2/yr$
Nitrogen[2] (total)	1.3 $g/m^2/yr$
Productivity[3,4]	3.4 $mg\text{-}C/m^3/hr$
	150 $g\text{-}c/M^2/yr$
Yearly average chlorophyll *a*[3]	2.3 mg/m^3
Euphotic zone	8 meters
Mean depth	84 meters
Mean water residence time	30 - 100 years

[1]From this report.

[2]After Bartsch (1968).

[3]After Schelske and Callender (1970).

[4]After Vollenweider (1975).

According to the Vollenweider (1975) phosphorus loading diagram, the current phosphorus loading to Lake Michigan places it in the oligotrophic zone of the diagram, below its "permissible" loading line, regardless of whether the 30 or 100 year hydraulic residence time value is used (Figure 1). Table 5 contains loading estimates as well as the mean depth and hydraulic residence times used in the Vollenweider diagram. This trophic classification is roughly in agreement with the productivity status of the open waters of this lake. Further reductions in the phosphorus loading to the lake would tend to move Lake Michigan to a relatively more "oligotrophic" position on the Vollenweider diagram. Table 5 also contains values for productivity, yearly average chlorophyll a, and the depth of the euphotic zone in the open waters of Lake Michigan.

IMPACT OF NUTRIENT REDUCTION ON WATER QUALITY IN LAKE MICHIGAN

Three more or less distinct regions of the lake must be considered in evaluating the potential impact that 80 to 90 percent phosphorus removal from domestic wastewater will have on water quality in Lake Michigan. These are the open waters of the lake (which are the primary focal point of the report), the nearshore waters, and the areas of restricted circulation such as river mouths, harbors, etc. Lee (1974) has discussed the characteristics of each of these regions and the probable impact which 80 to 90 percent phosphorus removal from domestic wastewaters will have on water quality in each of these areas. As pointed out by Lee (1974), there will likely be a small improvement in water quality in the open waters of the lake which should be manifested several years from now in the form of reduced phytoplankton growth. The greatest improvement in water quality will likely occur in the nearshore waters where phosphorus is already or can be made the limiting factor controlling planktonic and attached algal growth. As noted by Lee (1974), in areas of restricted circulation such as southern Green Bay, little or no improvement in water quality will likely occur from the 80 percent removal of phosphorus from domestic wastewater sources, since it would be insufficient to make phosphorus the limiting element controlling algal growth in these areas.

From an overall point of view, the information available today strongly supports the decision that was made in the late 1960's by the federal government and the states bordering on Lake Michigan to provide for 80 percent removal of phosphorus from domestic wastewater sources. Failure to take this step would have resulted in a very significant deterioration of water in Lake Michigan due to the increased urbanization of the lake's watershed. Instead of the steady, slow decline in water quality of the lake which would have resulted without the removal of phosphorus from domestic wastewaters, water quality in this lake should improve in the next 50 years due to the decision that was made in the late 1960's bringing about phosphorus removal from domestic wastewaters.

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TROPHIC STATUS AND NUTRIENT LOADING

FOR LAKE MICHIGAN

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INTRODUCTION

Lake Michigan is the world's sixth largest lake in terms of volume or surface area (Hutchinson 1957). Limnologically it has not been studied extensively. Studies on lakewide eutrophication, with the exception of fisheries, are relatively recent (Beeton 1965; Ayers and Chandler 1967; Beeton 1969). Limnological characteristics have been summarized and compared with other Laurentian Great Lakes by Beeton and Chandler (1963) and Schelske and Roth (1973). The lake serves many uses including municipal and industrial water supply, recreation, transportation, and commercial and sport fishing (Beeton and Chandler 1963; Beeton 1969).

The purpose of this paper is to review the effects of nutrient loading on biological communities and processes and to assess current and past trophic conditions in Lake Michigan. In this paper, the classical definition of oligotrophic and eutrophic will be used, i.e. that the terms imply variation in nutrient content. Eutrophication therefore results from nutrient enrichment or more specifically from increased supplies of limiting nutrients.

The discussion of nutrient loads for Lake Michigan is the subject of a separate paper by Piwoni, Rast and Lee in this volume.

Phytoplankton growth and primary production in Lake Michigan are limited by supplies of phosphorus, an environmental characteristic common to Lake Superior and Lake Huron as well. Evidence to support this statement is available from numerous field and laboratory experiments (Schelske and Stoermer 1972; Schelske et al. 1972; Schelske et al. 1974; Schelske, Simmons and Feldt, In press). It has also been shown that supplies of nitrogen have little if any effect on phytoplankton growth

(Schelske et al. 1974). Other evidence for control of eutrophication by inputs of phosphorus in the Laurentian Great Lakes is provided by data that show concentrations of phosphorus increase as primary productivity and chlorophyll *a* increase and Secchi disc transparency decreases (Fig. 1). Concentrations of chlorophyll *a* from the open-lake stations are lowest in Lake Superior, larger in Lake Huron, and largest in Lake Michigan, with phosphorus having the same relationship among the three lakes. These results clearly show that standing crops of algae (measured by concentrations of chlorophyll *a*) are positively correlated with concentration of phosphorus in the water. There also is nearly an order of magnitude range in rates of primary productivity for Lake Superior, the most oligotrophic of the lakes, and Lake Michigan.

Historical data on trends in levels of nutrients are not available, but ample evidence exists that conservative elements have increased (Beeton 1965). Whether changes in levels of conservative elements affect trophic state is an unresolved question.

INSHORE-OFFSHORE DIFFERENCES

Investigations of Lake Michigan, particularly the southern half, have revealed extreme differences in nutrients, phytoplankton productivity and standing crop between the nearshore waters and the open parts of the lake (Ladewski and Stoermer 1973; Stoermer 1972). Maximum chlorophyll *a* concentrations and standing crops of phytoplankton in the spring were 15 mg/m^3 and 8,000 cells/ml in the nearshore waters, several times greater than the offshore waters. In the spring during the presence of the thermal bar these differences may not be surprising since the nearshore area may be 6-10°C warmer than the offshore region; but these differences persist throughout the year so factors other than temperature are important (Holland and Beeton 1972). Nutrient input from rivers obviously contributes to the inshore-offshore differences, and this factor is important to consider since the input is not distributed uniformly over the lake.

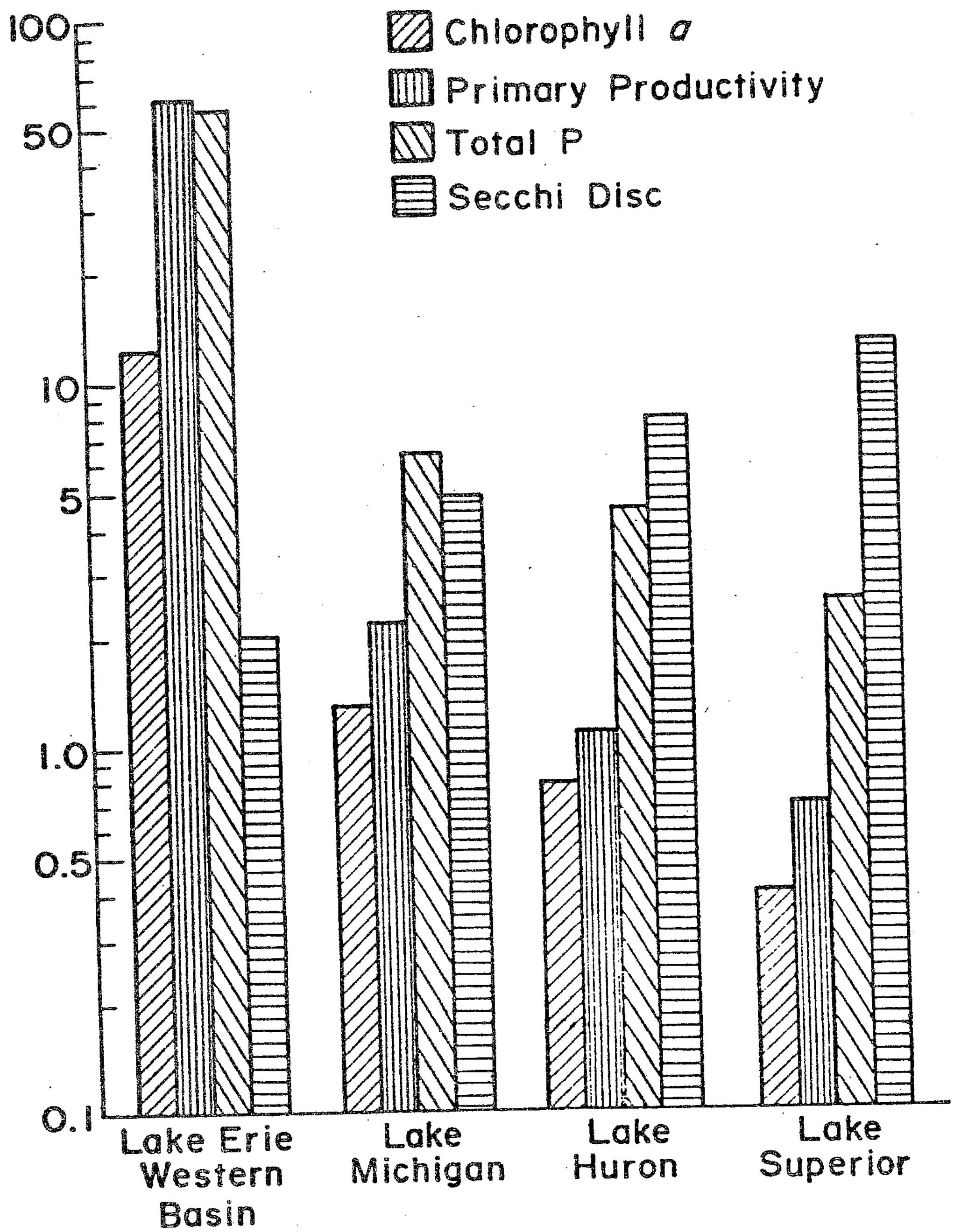

Figure 1. Chlorophyll *a* (mg/m^3), primary productivity ($mg\ C/m^3/hr$), total phosphorus ($mg\ PO_4\text{-}P/m^3$) and Secchi disc transparency (m) in the Great Lakes. (From Schelske 1974.)

No estimates of differences in nearshore nutrient loading have been made, but the significance of unequal loadings is obvious from data on nutrient input by various tributaries. Forty percent of the nutrient loading as total soluble phosphorus occurs on about five percent of the shoreline or from the input of the Muskegon, Grand, Kalamazoo, and St. Joseph rivers along the southeastern part of the lake. Another 35 percent is contributed by the Fox and Menominee rivers flowing into Green Bay, leaving only 25 percent of the loading for the remainder of the nearshore zone (Schelske 1974).

It is obvious that the effects of nutrient loading to Green Bay are manifested primarily in the bay and do not affect greatly the water quality in the northern part of Lake Michigan. Evidence for this statement is based primarily on the fact that the chemical and biological characteristics in northern Green Bay are very similar to those found in northern Lake Michigan (Schelske and Callender 1970). The nutrient contribution from Green Bay to northern Lake Michigan is therefore diffuse and represents only a relatively small, if any, source of nutrient enrichment for the open-lake waters. If there is a difference in nutrient composition, the loading factor might be significant due to the relatively large volume of water flowing out of Green Bay into Lake Michigan.

Due to the large difference in nutrient loading within the nearshore zone and between the nearshore zone and the offshore zone, it will not be possible to discuss the nearshore zone in this paper. The nearshore zone is also much more variable biologically than the offshore zone. This restriction is not too serious when one considers nutrient loading for the system, since the nearshore zone divided arbitrarily at the 20-m contour represents a small fraction of the lake (Table 1).

Table 1. MORPHOMETRIC AND HYDROLOGIC CHARACTERISTICS OF LAKE MICHIGAN.

	Nearshore	Offshore	Lake
Depth (m)	0–20	20–281	0–281
Length (km)	–	–	490
Breadth (km)	–	–	188
Water surface (km^2)	7,440	46,110	53,550[a]
Land drainage basin (km^2)	–	–	117,840
Land and water (km^2)	–	–	170,390
Maximum depth (m)	20	281	281
Average depth (m)	10	–	97.5
Volume of water (km^3)	74	4,796	4,870
Mean outflow (km^3/yr)	–	–	49.1

[a] Excluding Green Bay.

GENERAL MORPHOMETRY AND HYDROLOGY

Lake Michigan is deep, with a maximum depth of 281 m, and has two principal basins, the northern and the southern. The long axis is 490 km in a nearly north-south direction (Fig. 2). Morphometric and geological characteristics have been described by Hough (1958) and geological research has been reviewed by Sly and Thomas (1974). The drainage basin is large, 170,390 km^2 with the lake occupying one-third of the surface area (Table 1). Because of the large proportion of lake surface to drainage area, inputs of water and nutrients from precipitation directly on the lake surface are significant. The other large input of water is surface runoff from streams and rivers, although ground water inflow may be an important consideration in local areas. Water is lost mainly

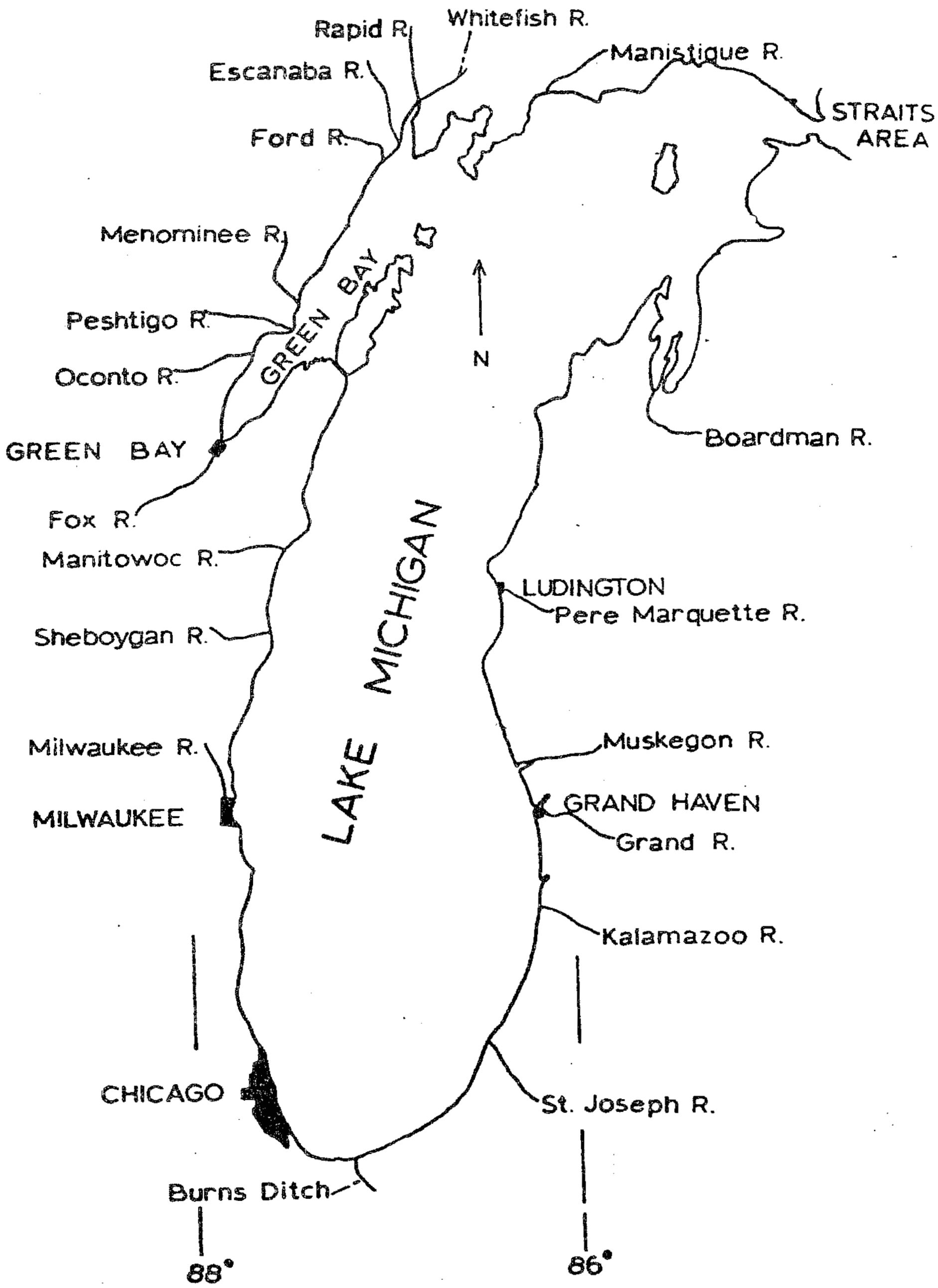

Figure 2. Major tributaries of Lake Michigan. (From Schelske 1974.)

through evaporation and the main outflow at the Straits of Mackinac (Table 1). A small amount of water in relation to the outflow (3 km^3/yr) is used to supply water for the city of Chicago and is not returned because waste water is diverted to the drainage system of the Mississippi River through the Chicago Sanitary Canal.

PHYSICAL AND CHEMICAL CONDITIONS

Areas of all five Laurentian Great Lakes are extensive so meteorological factors, particularly wind energy, can produce large-scale physical changes resulting in seiches, upwelling, and surface and subsurface currents. Two recent review papers provide excellent treatments of physical processes of large lakes (Mortimer 1974; Boyce 1974), but few studies have been made on how these physical processes influence chemical or biological processes. Physical characteristics are determined by the facts that the basins are closed and large enough so water transport is affected by Coriolis force (rotation of the earth), wind is the principal source of mechanical energy, and the water is thermally stratified in the summer (Boyce 1974).

Surface currents generally behave as expected due to the influence of Coriolis force with movement in a counterclockwise direction, i.e. currents moving south on the western shore and north on the eastern shore. The mean surface currents indicate two and possibly three cells or areas of counterclockwise circulation along the long axis of the lake (Millar 1952). Ayers et al. (1958) found this general pattern during periods of the normal westerly winds in June but not in August when winds were more easterly. It is well known that circulation patterns are transient and can change within a day, given normal shifts in wind speed and direction.

The thermal structure and stratification differ from either first-class or second-class temperate lakes, described by Hutchinson (1967). Lake Michigan is an atypical temperate lake in that it is not truly dimictic but is probably monomictic. After thermal stratification breaks down in the fall, the lake does not stratify again until the next summer, mixing at least periodically during storms all winter. Due to the long fetch and mixing in the winter, the entire water mass cools to less than 4.0°C with temperatures as low as 0.5°C being common in mid-lake during March (Rousar 1973). Investigators have observed inverse thermal stratification after the water has cooled below 4.0°C--the warmer water remains on the bottom until the lake is mixed completely by a storm. Very unusual winter conditions are needed for the lake to freeze completely for extended periods of time, but ice formation is extensive in shallow waters. Temperature ranges in the open lake are at least 0.5-22.9°C (Rousar 1973).

In addition to being monomictic, Lake Michigan differs from shallower temperate lakes in that spring warming produces a thermal bar in the lake. This phenomenon has been well described previously for other large lakes and for the Laurentian Great Lakes, by Rodgers (1965) for Lake Ontario, by Huang (1972) for Lake Michigan, and generally by Mortimer (1974). Differential warming in the spring produces the thermal bar; nearshore waters being shallower warm more rapidly than the deeper offshore waters. The thermal bar is the downwelling water at maximum density (4.0°C) that develops between water nearshore that is warmer and water offshore that is colder than 4.0°C. This sharp horizontal temperature gradient restricts mixing of inshore waters with the open lake and affects biological problems related to nutrient loading. Different and greater numbers of phytoplankton were present on the shoreward side of the thermal bar (Stoermer 1968).

Seasonal, physical and chemical data are available from Rousar (1973). These data are summarized in Table 2 as they are the most extensive data set available for open-water conditions during an entire year. These data also appear to represent chemical conditions during 1970-1971 because results available from other investigators sampling at the same time are comparable in magnitude.

Table 2. PHYSICAL AND CHEMICAL DATA FOR LAKE MICHIGAN WATER COLLECTED FROM A DEPTH OF 4 METERS. Nearshore values are Station 1 near Milwaukee and offshore values are for Stations 3 and 4 between Milwaukee, Wisc. and Ludington, Mich. Data presented are averages and ranges for an 18-month period in 1970-1971. Data are from Rousar (1973).

	Nearshore	Offshore
Temperature (C)	11.4 (0.1-20.8)	11.4 (0.5-22.9)
pH	8.3 (7.9-8.8)	8.3 (8.1-9.0)
Total alkalinity (meq/liter)	2.14 (2.06-2.25)	2.12 (2.04-2.17)
Specific conductance (μmhos/cm @ 25 C)	265 (257-278)	259 (251-273)
NO_3-N (mg/liter)	0.19 (0.10-0.29)	0.19 (0.12-0.27)
SiO_2 (mg/liter)	0.75 (0.2-1.6)	0.85 (0.2-1.5)
Total P (μg/liter)	15.2 (8.2-32.9)	8.1 (2.4-16.0)
Soluble reactive P (μg/liter)	1.9 (ND-10.8)	1.1 (ND-4.0)

A distinct difference is obvious in the total phosphorus concentrations between nearshore and offshore, but there is no difference in the averages for the other parameters (Table 2). The lack of significant differences is due partly to the technique of averaging, partly to the parameters listed, and partly to the locations of the stations.

Distinct differences between offshore and inshore stations were obtained for Secchi disc transparency during two years of intensive sampling (Fig. 3). The transparency varies seasonally and, with the exception of the minimum in September, was correlated inversely with cell counts and chlorophyll concentrations (Ladewski and Stoermer 1973). September transparency was reduced by upwelled light from "milky water," probably suspensions of precipitated calcium carbonate.

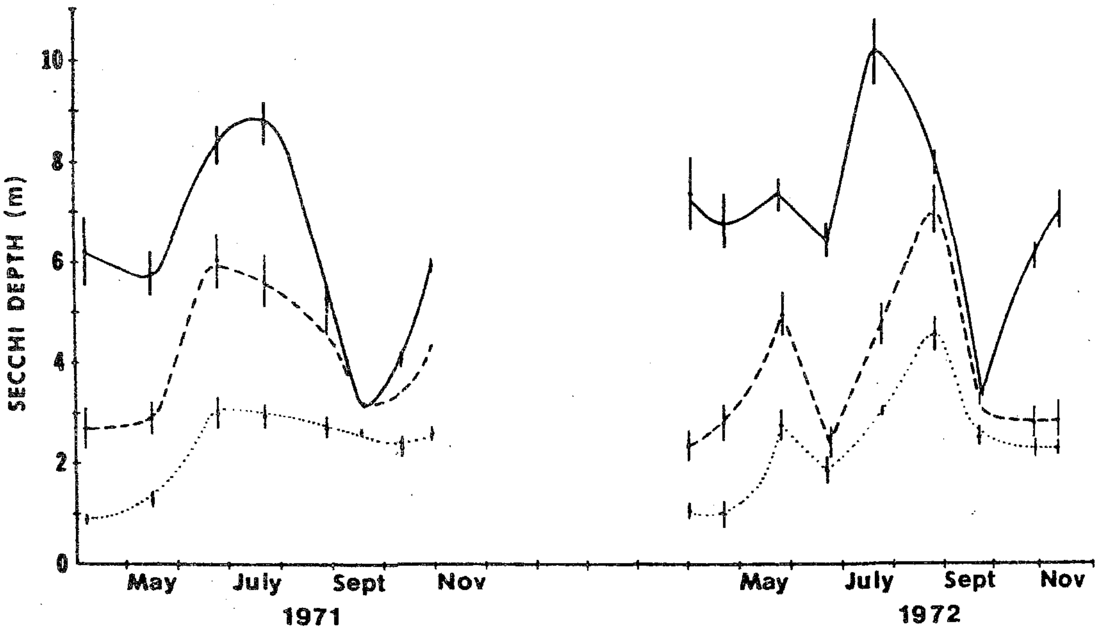

Figure 3. Secchi disc transparency averaged by depth range and month. Key: Dotted line shows mean value for stations less than 10 m deep, dashed line shows mean value for stations between 10 m and 40 m deep and solid line shows mean value for stations deeper than 40 m. For each cruise there are nominally 12 stations shallower than 10 m, 16 between 10 and 40 m deep and 13 deeper than 40 m. Error flags show the standard error of the mean. (From Ladewski and Stoermer 1973.)

"Milky water" is associated with increases in pH of surface waters during summer. Maximum open-water values for pH are now much greater in the summer than 8.0-8.2 commonly cited in many papers, as pointed out previously by Schelske and Roth (1973). This fact is confirmed by the maximum pH of 9.0 recorded by Rousar (Table 2). Data for the conservative elements and representative values for a number of the trace elements are given in Table 3.

Table 3. LAKE MICHIGAN WATER CHEMISTRY FOR CONSERVATIVE AND TRACE ELEMENTS. Data on trace elements are from Rossmann (1973).

Element	Concentration (mg/liter)
Ca	36
Mg	11
Na	3.8
K	1.4
SO_4	18
Cl	8.0
Fe	0.007
Mn	0.00084
Cu	0.0027
Zn	0.004
Co	<0.001
Ni	0.0065
Mo	0.0018
Ba	0.026

MAJOR NUTRIENT CYCLES

Some data are available for the seasonal cycles of phosphorus, nitrogen and silica in Lake Michigan--these elements and carbon are the major nutrients for phytoplankton. Supplies of carbon are more than adequate for phytoplankton growth.

Phosphorus concentrations in the lake are low, with total phosphorus averaging 8.0 μg P/liter (Table 2). Allen (1973) reported averages of less than 7.0 μg P for samples collected in 1965. Soluble reactive phosphorus being frequently below 1.5 μg P/liter leads one to question the utility of this measurement, particularly on a routine basis (Schelske and Callender 1970). No clear seasonal cycle of either soluble reactive or total phosphorus is evident from Rousar (1973) or Allen (1973) but summer values for total phosphorus appear to be smaller than at other times of the year.

Of the three forms of combined inorganic nitrogen, only nitrate is quantitatively significant. Concentrations of ammonia and nitrite are low being only a few percent of the nitrate concentrations. Nitrate varies seasonally with a maximum in the winter of 0.27 mg N/liter followed by a steady decline to 0.12 mg N/liter in August (Rousar 1973). This decline is due to nitrogen utilization for phytoplankton growth.

The seasonal cycle for silica is similar to that for nitrate. A maximum value of 1.4-1.5 mg SiO_2/liter occurred for the winter of 1970-1971 (Rousar 1973). Data from the Great Lakes Research Division, University of Michigan, collected in the spring of 1971, agree well with these values for the winter, and it seems reasonable therefore to conclude that the maximum open-lake concentration was no greater than 1.5 mg/liter (Stoermer 1972). The minimum value reported by Rousar of 0.2 mg/liter occurred in August. Minimum values during the summer may not be accurate due to technical problems associated with detecting concentrations lower than 0.1 mg/liter, but it is clear from unpublished GLRD data that the minimum is presently less than 0.1 mg/liter.

A more detailed discussion of the seasonal cycle of silica has been derived from data collected in 1971 by the GLRD as part of a study of algal quality in southern Lake Michigan (Stoermer 1972). For this discussion, silica depletion is defined as concentrations equal to or less than 0.2 mg/liter.

Silica depletion in 1971 had occurred in localized nearshore areas at the time of our first collections (in late March and early April). Concentrations of 1.4 mg/liter were measured at mid-lake stations, indicating little utilization by diatoms at this time, but were significantly less than 1.4 mg/liter at distances as great as 6.4 km offshore.

The zone of silica depletion increased in May and June and by June extended to at least 6.4 km offshore. The mid-lake values ranged from 0.9 to 1.3 mg/liter in May and were generally less than 1.0 mg/liter in June. Silica concentrations in June over much of the lake were less than 0.7 mg/liter, indicating that half the silica reserve in the euphotic zone had been utilized by diatoms.

Silica was essentially depleted in the euphotic zone to a depth of 20 m in July. In August and September concentrations remained low, generally < 0.2 mg/liter over the study area.

Increases in silica in surface waters from the spring-summer lows were not evident until the October cruise when concentrations ranged from 0.3 to 0.5 mg/liter. Concentrations did not increase greatly on the next and last cruise at the end of October, as most values ranged from 0.3 to 0.6 mg/liter. Rousar did not find maximum values in the epilimnion until January.

PHYTOPLANKTON

Based on the report of "more than 700 morphologically distinguishable entities" in a study of plankton diatoms from Lake Michigan (Stoermer and Yang 1969), one might conclude that the phytoplankton had been studied extensively. Such a conclusion would be erroneous for a number of reasons recognized by Stoermer and Yang. First, many of the data analyzed were from nearshore regions that differ physically, chemically and biologically from the open lake. Second, there is a lack of seasonal data. The only data collected seasonally are those obtained by sampling municipal water intakes that are necessarily located close to shore. Third, many of the data were obtained from vertical plankton tows, so the absolute abundance of organisms in the water cannot be estimated. Fourth, there are therefore few quantitative data on cell counts or even on chlorophyll concentrations which might be used to estimate the biomass of phytoplankton. Stoermer and Yang therefore studied the relative abundance of different organisms in available collections. With this approach, it was possible to obtain a comparative data set for samples collected from as early as the 1880's to 1967 and to make a detailed analysis of 44 common species (Table 4). Because many of the older samples were preserved as material for diatom identification or in samples in which other types of algae were destroyed, it was not possible to work with the complete phytoplankton assemblage.

Historically, the plankton flora of Lake Michigan was dominated by diatoms (Stoermer 1967; Ahlstrom 1936) as would be expected for a pristine flora (Stoermer and Yang 1969). It is obvious from other studies of the Laurentian Great Lakes, particularly Lake Superior which is the most oligotrophic, that diatoms are the dominant phytoplankton organisms (Holland 1965; Schelske et al. 1972). It appears from limited studies of populations in Lake Superior that the phytoplankton assemblage is the typical *Cyclotella* plankton of oligotrophic lakes as characterized by Hutchinson (1967). The available collections from Lake Michigan indicate that the oligotrophic *Cyclotella* plankton is presently never as dominant as it is in Lake Superior (Holland 1969).

The species composition of phytoplankton in Lake Michigan changed markedly in the past 100 years as the result of accelerated eutrophication and possibly from other forms of pollution. A change from 1930-1931 to the 1960's was documented by Stoermer (1967). During this time interval, the number of euplanktonic species that could be considered indicators of eutrophication increased 70 percent. New species, such as *Stephanodiscus binderanus* and *Stephanodiscus hantzschii*, characteristic of eutrophic conditions became dominant (Stoermer and Yang 1970). In addition, more recently the predicted shift (Schelske and Stoermer 1971) of phytoplankton assemblages dominated by diatoms to those dominated by blue-green algae has occurred in summer populations (Stoermer 1972).

Seasonally the spring pulse occurs as early as February and as late as April at the Chicago water intakes, a nearshore location (W. F. Danforth, Ill. Inst. Tech., personal communication), but based on chlorophyll it

Table 4. DOMINANT DIATOMS IN LAKE MICHIGAN.
As taken from Stoermer and Yang (1970).

Amphipleura pellucida (Kütz.) Kütz.	*Nitzschia bacata* Hust.
Asterionella formosa Hass.	*N. dissipata* (Kütz.) Grun.
Cyclotella comta (Ehr.) Kütz.	*N. recta* Hantz.
C. kuetzingiana Thwaites	*Nitzschia* sp. #2.
C. meneghiniana var. *plana* Fricke	*Rhizosolenia eriensis* H. L. Smith
C. michiganiana Skv.	*Stephanodiscus alpinus* Hust.
C. ocellata Pant.	*S. binderanus* (Kütz.) Krieger
C. operculata (Agardh) Kütz.	*S. hantzschii* Grun.
C. pseudostelligera Hust.	*S. minutus* Grun.
Diatoma tenue var. *elongatum* Lyngb.	*S. niagarae* Ehr.
D. tenue var. *pachycephala* Grun.	*S. subtilis* (Van Goor) A. Cleve
Fragilaria capucina Desm.	*S. tenuis* Hust.
F. capucina var. *lanceolata* Grun.	*S. transilvanicus* Pant.
F. capucina var. *mesolepta* Rabh.	*Synedra delicatissima* var. *angustissima* Grun.
F. crotonensis Kitton	*S. demerarae* Grun.
F. intermedia Grun.	*S. filiformis* Grun.
F. intermedia var. *fallax* (Grun.)	*S. ostenfeldii* (Krieger) A. Cleve
F. pinnata Ehr.	*S. ulna* var. *chaseana* Thomas.
Melosira granulata (Ehr.) Ralfs	*S. ulna* var. *danica* (Kütz.) V.H.
M. granulata var. *angustissima* O. Müll.	*Tabellaria fenestrata* (Lyngb.) Kütz.
M. islandica O. Müll.	*T. fenestrata* var. *geniculata* A. Cleve
M. italica subsp. *subartica* O. Müll.	*T. flocculosa* (Roth) Kütz.

does not occur until May or June in the offshore waters between Milwaukee and Ludington (Rousar 1973). One would expect the pulse to be delayed from south to north due to cooler air temperatures and from the shore to open lake due to slower increases in offshore water temperature. These effects are related to the length of the lake and the thermal bar. The latter effect is evident in the data presented by Ladewski and Stoermer (1973) and by Stoermer (1972). The summer minimum occurs in late August and September followed by an autumnal pulse smaller than the spring maximum. According to Rousar's data, chlorophyll *a* concentrations for the open lake average 4.5 mg/m^3 during the spring pulse, 3.0 mg/m^3 for the fall pulse and about 1.0 mg/m^3 during the summer minimum. Ladewski and Stoermer (1973) found a spring maximum of 2.7 mg/m^3, a fall maximum of 1.8 mg/m^3 and a summer minimum of 0.7 mg/m^3 from data averaged for 13 offshore stations (Fig. 4). Allen (1973) reported a minimum of 0.5 mg/m^3 and a maximum of 2.4 mg/m^3.

In terms of species composition the spring pulse is dominated by diatoms and the summer minimum by blue-greens (Stoermer 1972). Blue-greens in surface samples in 1971 comprised a major fraction of the phytoplankton counts from late August until late October when sampling was terminated (Stoermer 1972); percentages of blue-greens exceeded 80% in many of the samples. The dominant species was *Anacystis incerta*.

The spring pulse from 1968-1972 at Chicago was dominated by *S. binderanus* and *S. hantzschii*, while the autumn maximum consisted mainly of *Asterionella*, *Fragilaria*, and *Tabellaria* (Danforth, personal communication). *S. hantzschii* and *S. binderanus* were not common at the Chicago filtration plants until 1956 and 1960, respectively -- these species apparently replaced *Melosira islandica* indicating severe environmental perturbation (Stoermer and Yang 1970), presumably nutrient enrichment. Holland (1968, 1969) found that *M. islandica* was dominant in the open waters during May and June, but was replaced by other species of *Melosira* in more eutrophic areas; one of the species of *Melosira*, *M. binderana* is a synonym for *S. binderanus*. Fluctuations in the standing crops of

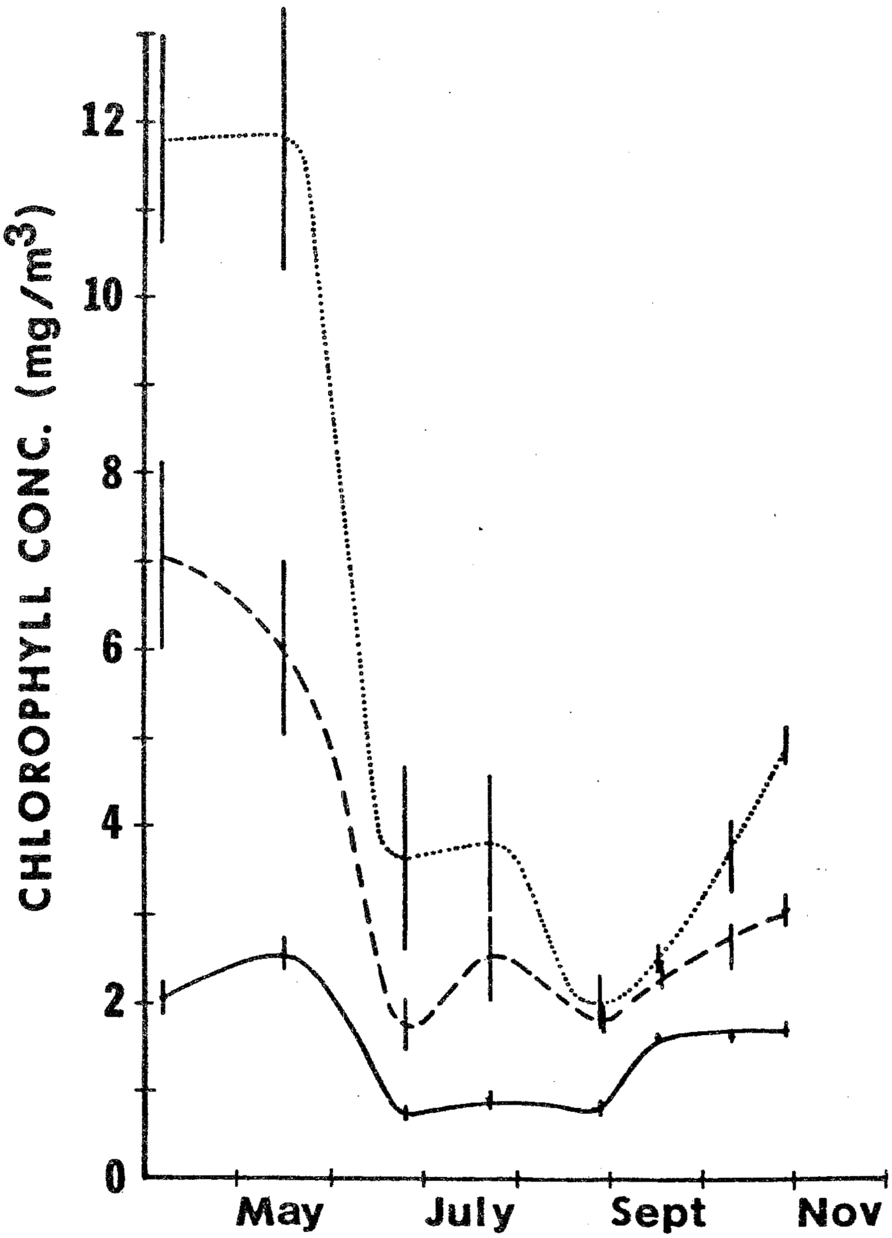

Figure 4. Chlorophyll concentration in 1971 averaged by depth range and month. See Figure 3 for key. (From Ladewski and Stoermer 1973.)

common diatoms from May to October in northern Lake Michigan have been studied by Holland (1969).

The relative abundance of Chlorophyta, Chrysophyta, and Cyanophyta in the offshore plankton was studied by Stoermer (1967). Each group comprised only a minor part of the phytoplankton compared to the Bacillariophyta. Some of the more abundant forms of green algae were *Botryococcus braunii* Kütz., *Closterium aciculare* West, *Dictyosphaerium pulchellum* Wood, *Sphaerocystis schroeteri* Chodat, and several species of *Oöcystis* Nägeli. The main genus of Chrysophyta was *Dinobryon* Ehr.. Five species were recorded, but the dominant one was *D. divergens* Imhof. Blue-greens were represented by *Chroococcus limneticus* Lemm., *C. minutus* (Kütz.) Nägeli, *Coelosphaerium naegelianum* Unger, *Gomphosphaeria lacustris* Chodat, and *Oscillatoria mougeotia* Kütz.

Phytoplankton productivity is relatively low with summer values averaging about 4.0 mg $C/m^3/hr$ (Schelske and Callender 1970). Annual rates of carbon fixation for offshore waters ranged from 121-139 g C/m^2 at three offshore stations between Ludington and Milwaukee as compared to a maximum value of 247 g C/m^2 at a nearshore station near Milwaukee (Fee 1973). Phytoplankton productivity in the Great Lakes has been compared by Vollenweider et al. (1974).

ZOOPLANKTON

Studies of zooplankton have concentrated mainly on the Crustacea (Table 5) with very little being known about Protozoa or Rotifers including their taxonomy (Gannon 1972a). There are no data on open-water zooplankton prior to 1954 (Wells 1970) or for the winter months. Zooplankton studies on the Great Lakes have been reviewed by Watson (1974).

Wells (1970) found changes between 1954 and 1966 in both the size and species composition of zooplankton populations. Size-selective preda-

Table 5. ZOOPLANKTON CRUSTACEA OF LAKE MICHIGAN.

Common Species	Gannon (1972a)	Wells (1970)	Roth and Stewart (1973)
Copepods			
Diaptomus ashlandi Marsh	X	X	X
Diaptomus minutus Lilljeborg	X	X	X
Diaptomus oregonensis Lilljeborg	X	X	X
Diaptomus sicilis Forbes	X	X	X
Epischura lacustris Forbes	X	X	X
Eurytemora affinis (Poppe)	X	X	X
Limnocalanus macrurus Sars	X	X	X
Senecella calanoides Juday	X	X	
Cyclops bicuspidatus thomasi Claus	X	X	X
Cyclops vernalis Fischer	X	X	X
Eucyclops agilis (Koch)			
Mesocyclops edax (Forbes)	X	X	
Tropocyclops prasinus mexicanus Kiefer	X		X
Canthocamptus robertcokeri M.S. Wilson	X		X
Cladocera			
Bosmina longirostris (Müller)	X	X	X
Eubosmina coregoni Baird	X	X	X
Daphnia galeata-mendotae Birge	X	X	X
Daphnia longiremis Sars	X	X	
Daphnia retrocurva Forbes	X	X	X
Daphnia schødleri Sars	X		
Ceriodaphnia lacustris Birge	X	a	
Ceriodaphnia quadrangula (Müller)	X	a	X
Alona affinis (Leydig)	X		
Chydorus sphaericus (Müller)	X		X
Holopedium gibberum Zaddach	X	X	X
Leptodora kindtii (Focke)	X	X	X
Polyphemus pediculus (L.)	X	X	X
Diaphanosoma leuchtenbergianum Fischer	X	X	X

[a]*Ceriodaphnia* species.

tion by alewife was given as the most likely factor for the change. The largest species, *Leptodora kindtii* *Daphnia galeata-mendotae*, *D. retrocurva*, *Limnocalanus macrurus*, *Epischura lacustris*, *Diaptomus sicilis* and *Mesocyclops edax*, declined in abundance with *D. galeata-mendotae* and *M. edax* decreasing from abundant to extremely rare. At the same time smaller species increased in abundance. Species such as *M. edax*, *D. galeata-mendotae*, *Diaptomus oregonensis*, *Diaphanosoma leuchtenbergianum* and *L. kindtii* that decreased in Lake Michigan due to selective predation are abundant in Green Bay. Gannon (1972b) concluded that size-selective predation had a smaller effect on zooplankton Crustacea in Green Bay than in open Lake Michigan because higher rates of primary productivity support greater rates of zooplankton production in Green Bay. If greater primary productivity is a causal factor it would be related directly to eutrophication.

Eurytemora, a marine species, has invaded the lake — it was recorded by Wells (1970) in 1966 but not in 1954 and by other workers (Table 5).

Seasonal distribution of the major groups appears quite simple. Copepods are present throughout the year, but cladocerans are found only during the summer. Gannon's data indicate that cladocerans do not appear until thermal stratification is present and persist until the lake becomes homothermous. *Cyclops bicuspidatus thomasi* is the most abundant copepod with *Diaptomus ashlandi*, *Diaptomus minutus*, *Diaptomus oregonensis*, *Diaptomus sicilis* and *Limnocalanus macrurus* being common but less abundant. Cladocera are most abundant from June to September with the maximum populations occurring in July and August when copepods are also most abudant. In the summer the dominant species are the copepods, *C. bicuspidatus thomasi*, *Diaptomus ashlandi* and the cladocerans, *Bosmina longirostris*, *Daphnia retrocurva*, and *Daphnia galeata-mendotae*. *Bosmina* dominates the zooplankton in July and August. Later in the summer during August and September *Daphnia* replaces *Bosmina* as the dominant cladoceran.

Quantitative data on zooplankton abundance are lacking, with only four studies of major importance. Each of these studies was restricted to small areas of the lake or to few stations. In addition, the data are not comparable as two different methods were used. Wells (1970) used horizontal tows with a calibrated Clarke-Bumpus sampler and 0.366 mm silk net; Gannon (1972a), Roth and Stewart (1973), and Stewart (1974) used vertical tows with a 0.5-m diameter nylon net. Gannon used a 0.256-mm aperture and Roth and Stewart used a 0.158-mm aperture. All of the data are reported as individuals/m^3. Gannon's station was 60 m deep and Roth and Stewart's stations were 14 m for the nearshore station and 40 m for the offshore station. Gannon reported a maximum crop of 8300 individuals/m^3 in August, whereas Roth and Stewart (1973) reported maxima of 130,000 individuals/m^3 in July at the offshore station and 280,000 individuals/m^3 at the nearshore station in August. Roth and Stewart considered that some of these differences may have been due to more immature forms being collected by their smaller mesh net, but concluded that the main factor was greater productivity in their study area.

Roth and Stewart (1973) also noted differences between their offshore and inshore station. The cladoceran bloom appeared to start earlier at the nearshore station and contained a greater proportion of cladocerans than the offshore station, presumably an effect attributable to earlier warming at the inshore station. The large population of *Bosmina longirostris* at the nearshore station was attributed in part to size-selective feeding by abundant nearshore fish on zooplankton. Whether the inshore station contained larger standing crops of zooplankton is a question of data interpretation, since the offshore station was three times deeper than the nearshore station. Maximum standing crops on an areal basis therefore would have occurred at the offshore station, because abundances at the nearshore station were seldom greater than a factor of two larger than the offshore station. Actually the largest standing crop of zooplankton, 360 mg dry weight/m^3, occurred at the offshore station in July; the largest standing crop at the nearshore station was 280 mg dry weight/m^3 in August. Biomass values were as low as 30 mg dry weight/m^3 in April.

BENTHOS

Knowledge of the benthos is restricted mainly to the macroinvertebrates. Numerically the fauna in the main lake is dominated by Amphipoda, Oligochaeta, Sphaeriidae, and Chironomidae in that order (Robertson and Alley 1966). Nearshore Gastropoda, Hirudinea, other Insecta and other Crustacea may be numerous (Mozley 1974). In the main lake, *Pontoporeia affinis* is the only species of Amphipoda. Cook and Johnson (1974) have reviewed several aspects of studies on macrobenthos of the Great Lakes.

Powers and Alley (1967) investigated the depth distribution of benthos. Maximum numbers occurred at 30 m, with large numbers being present at 40 and 50 m. At depths of 20 m and greater, *Pontoporeia* was the dominant organism, comprising more than 50 percent of the total counts. Proportions of *Pontoporeia* increased with depth, with 75 percent of the total being at depths greater than 80 m. At 30 m, the mean and standard deviation of total counts/m^2 was 15,000 $\pm$ 6,700 and at 40 and 50 m it was 11,000 $\pm$ 5,000. At 30, 40 and 50 m, the mean and standard deviation for *Pontoporeia* was 8,600 $\pm$ 3,700, 6,800 $\pm$ 3,100 and 6,300 $\pm$ 2,800. The mean number of benthic organisms declined to less than 1,000 at depths greater than 200 m. Average standing crops (dry weight) of macrobenthos ranged from highs of 10 and 20 g/m^2 at 30 and 10 m to approximately 0.3 g/m^2 at depths greater than 200 m.

At depths less than 30 m, the second most abundant group of organisms is the Tubificidae (Mozley 1974). This group of oligochaetes, composed of a number of species, is dominated by *Limnodrilus hoffmeisteri*, *Potamothrix moldaviensis*, *Peloscolex freyi*, *P. ferox* and *Tubifex tubifex*. In addition, the lumbriculid oligochaete *Stylodrilus heringianus* is abundant at depths greater than 20 m (Hiltunen 1967).

Sphaerids are most abundant at 30 m and also occurred abundantly at depths of 60 m and less. There are three species of *Sphaerium*, *S. nitidum*, *S. striatinum*, and *S. corneum*, and at least nine species of *Pisidium* (Robertson 1967). The deep water species, *P. conventus*, is the most abundant *Pisidium*. In shallow water, *P. casertanum*, *P. henslowanum* and *P. lilljeborgi* are the most abundant species.

Chironomids form a relatively minor part of the benthos, with maximum counts averaging 200/m^2 at 40 and 50 m (Powers and Alley 1967). This group is complex from the taxonomic standpoint, so species identification is either not attempted or is questionable in many studies. Certain species have been considered pollution tolerant and others as pollution intolerant, thereby types of chironomids have been used to assess environmental quality (Brinkhurst, Hamilton and Herrington 1968; Mozley, In prep.).

The benthic fauna in deep waters has been affected little by environmental changes. Although abundances of *Pontoporeia* and oligochaetes were significantly greater in 1964 than in 1931, Robertson and Alley (1966) stated "no definite conclusions can be reached concerning long-term trends" due to expected year-to-year variations in abundances. Severe changes in benthic organisms have occurred in localized areas, harbors, bays, and river mouths, but extensive changes over large areas have been observed only in southern Green Bay (Howmiller and Beeton 1970). In southern Green Bay, drastic changes have been documented, including the disappearance of mayfly nymphs (*Hexagenia*), a change that also occurred in the western basin of Lake Erie as the result of oxygen depletion (Britt 1955).

FISH

Much has been written about the fish fauna, partly reflecting the concern over decreases in the abundance of species in the commercial fishery. Wells and McLain (1973) summarized many of the papers dealing with the history of fish and list 38 species (Table 6) as "all common fish of Lake Michigan, past and present." A much longer list of species, numbering 70 to 75, resulted from intensive sampling of species that "are extremely rare or transients that normally inhabit streams, inland lakes or protected bays" (Jude et al. 1975). Only 32 of these 70 species were

Table 6. PAST AND PRESENT COMMON FISH OF LAKE MICHIGAN.

Sea lamprey[a]	*Petromyzon marinus*
Lake sturgeon	*Acipenser fulvescens*
Alewife[a]	*Alosa pseudoharengus*
Lake whitefish	*Coregonus clupeaformis*
Blackfin cisco	*Coregonus nigripinnis*
Deepwater cisco	*Coregonus johannae*
Longjaw cisco	*Coregonus alpenae*
Shortjaw cisco	*Coregonus zenithicus*
Bloater	*Coregonus hoyi*
Kiyi	*Coregonus kiyi*
Shortnose cisco	*Coregonus reighardi*
Lake herring	*Coregonus artedii*
Round whitefish	*Prosopium cylindraceum*
Lake trout	*Salvelinus namaycush*
Brook trout	*Salvelinus fontinalis*
Rainbow trout (steelhead)[a]	*Salmo gairdneri*
Brown trout[a]	*Salmo trutta*
Coho salmon[a]	*Oncorhynchus kisutch*
Chinook salmon[a]	*Oncorhynchus tshawytscha*
Rainbow smelt[a]	*Osmerus mordax*
Northern pike	*Esox lucius*
Carp[a]	*Cyprinus carpio*
Emerald shiner	*Notropis atherinoides*
Spottail shiner	*Notropis hudsonius*
Longnose sucker	*Catostomus catostomus*
White sucker	*Catostomus commersoni*
Channel catfish	*Ictalurus punctatus*
Bullheads	*Ictalurus* spp.
Trout-perch	*Percopsis omiscomaycus*
Burbot	*Lota lota*
Ninespine stickleback	*Pungitius pungitius*
Smallmouth bass	*Micropterus dolomieui*
Yellow perch	*Perca flavescens*
Walleye	*Stizostedion vitreum vitreum*
Freshwater drum	*Aplodinotus grunniens*
Slimy sculpin	*Cottus cognatus*
Spoonhead sculpin	*Cottus ricei*
Fourhorn sculpin	*Myoxocephalus quadricornis*

[a] Species that have been introduced or invaded the lake.

collected in both years of the two-year study of a nearshore area in the southeastern part of the lake.

During the period of historical record, eight common species were either introduced or gained access to the lake (Table 6). The rainbow trout or steelhead, brown trout and carp have been present in Lake Michigan since the turn of the century. A few chinook salmon were introduced between 1873 and 1880 but they did not become established. Extensive stocking of coho salmon and chinook salmon began in 1966 and 1967. It was originally thought that these species would not reproduce in the Great Lakes and, although there is ample evidence that spawning runs have been established in streams in Wisconsin and Michigan, it is still not certain if these spawning runs would continue if stocking was discontinued. Another salmon from the Pacific Ocean was introduced with the release of 2,000 masu salmon (*Oncorhynchus masou*) in 1920, and some 645,000 Atlantic salmon (*Salmo salar*) were released in the lake between 1872 and 1932, but neither became established. The pink salmon (*Oncorhynchus gorbuscha*) is the only introduced salmon that has established self-sustaining populations. It was introduced in Thunder Bay, Lake Superior in 1956 and has recently spread to Lakes Huron and Michigan.

Three of the species presently common in the lake are native to the Atlantic Ocean. First records of the rainbow smelt in 1923, the sea lamprey in 1936 and the alewife in 1949 represent relatively recent introductions. The smelt in Lake Michigan originated from a planting in Crystal Lake, Michigan in 1912 (Van Oosten 1937). The alewife and sea lamprey probably entered the Great Lakes drainage via the Erie Canal which linked the Mohawk-Hudson River system entering the Atlantic Ocean with the Oneida-Oswego River system entering Lake Ontario (Smith 1970; Aron and Smith 1971). They became established in Lake Ontario and subsequently circumvented the natural barrier at Niagara Falls and reached the other Great Lakes via the Welland Canal.

Some of the introduced species have caused severe environmental problems as well as (at least on the short term) environmental benefits. Most experts on the problems of the commercial fisheries of the Great Lakes attribute some of the cause to the invasion by the sea lamprey. In Lake Michigan, during its period of maximum abundance in the 1950's, the sea lampreys destroyed from five to twelve million pounds of fish per year. At this time the prey was largely deepwater ciscoes, as most of the lake trout had disappeared (Smith 1968). Control measures for sea lamprey have been successful. Adult fish are trapped in weirs on the spawning streams, and the larvae are killed in the spawning streams with the selective larvicide, 3-trifluoromethyl-4-nitrophenol.

With the decline in lake trout and other large predators, the population explosion of alewives resulted, reaching a climax in 1967 with massive dieoffs of alewives. Dead fish clogged municipal and industrial water intakes and littered beaches, detracting from their desirability for swimming and creating costly problems of removal. Coho salmon and chinook salmon from the Pacific Ocean have been introduced since 1966 and 1967, partly to overcome the problem of over-population with alewives, providing a predator to check population increases and large fish for a sport fishery. Results were dramatic, the salmon flourished on the abundant alewives and provided an excellent sport fishery. In Lake Michigan the largest fish caught by angling have been a 13.7 kg coho, only .4 kg less than the world's record, and a 19.6 kg chinook.

Although largely successful, stocking Pacific salmon has not been without its environmental repercussions. When salmon mature they return to spawn in the stream in which they were stocked. These spawning runs of salmon have been extensive, creating a bonanza for fishermen in the streams. They also provide large numbers of fish that can be removed by unsportsmanlike means including snagging. Dead fish often litter the streams as they die after spawning or are caught and discarded by fishermen who do not value fish in poor spawning condition.

Historically and continuing to the present time, the most common native and introduced fish, especially the large predators, are species characteristic of oligotrophic environments. This fact alone leads one to conclude that some of the characteristics of open Lake Michigan are definitely oligotrophic. The original fish fauna included 10 species of coregonids (Table 6) and one salmonid. Of these the lake whitefish, lake herring and lake trout were taken in the largest quantities by the fishery. Whitefish are recovering as a result of sea lamprey control, and lake trout are being maintained through artificial stocking, but the other species are no longer abundant. All of these species have been sought actively by commercial fishermen and have been significant in the commercial fishery.

Commercial fishing in Lake Michigan began at least as early as 1843 primarily for abundant nearshore populations of lake whitefish. "By 1860 certain grounds for this species were becoming depleted and by the 1870's complaints about the scarcity of whitefish were common" (Wells and McLain 1973). Logging and dumping sawdust in streams were factors in the decline. Other species were abundant in the fishery and then declined. For example, catches of lake trout averaged eight million pounds in the early 1900's, declined to five million pounds in the 1930's and increased to more than six million pounds in the early 1940's before declining sharply in the late 1940's. By the mid-1950's, the lake trout was believed to be extinct.

The total commercial production was greatest from 1893-1908 when the average harvest was 41 million pounds yearly. Production dropped between 1908 and 1911, due primarily to a decrease in catches of lake herring, and then fluctuated with catches ranging from 20 to 30 million pounds until 1966 when catches of alewives became large. The peak year was 1967 when the total catch was 60 million pounds, of which 42 million pounds were alewives.

Causes for changes in the fish fauna are not fully explained (Smith 1972). Some of the factors have been modification of the drainage basin first by logging and then by agriculture, influences of urbanization and industrialization, invasion of the sea lamprey, and undoubtedly the effects of overfishing for commercial species (Smith 1972).

ASSESSMENT OF TROPHIC CHANGE

Basically either chemical or biological approaches can be used to determine changes in the trophic status of lakes. Detectable changes in the biota should be those that result from nutrient enrichment of the system or in the case of Lake Michigan from increased loading of phosphorus.

Of the biota only the phytoplankton can be used to determine changes in trophic state of Lake Michigan. Qualitatively, the benthic community has been affected little by nutrient enrichment, and quantitatively the problems of large variances in sampling invoke the need for large differences to detect changes in standing crops (Mozley 1974). No zooplankton data are available for long-term assessments of standing crop and there is no direct evidence that changes can be used to assess trophic changes--the latter aspect also is confused by size-selective predation by fish and its resultant shifts in species composition. Many changes in the abundance and species composition of fish have been documented, but these "have been largely unexplained and have been a subject of uncertainty and controversy" (Smith 1972). One of the reasons that changes in trophic state have not been obvious is that the biological communities of the open waters, with the exception of the phytoplankton, continue to be those characteristic of oligotrophic environments.

Changes in the phytoplankton species composition have been documented, and the cause for the change has been related to increased inputs of phosphorus. Changes in species composition have occurred that apparently reflect nutrient enrichment. Changes in the diatom flora have been documented in papers by Stoermer and Stoermer and Yang from data collected in the early 1960's. Another shift in species composition has occurred due to depletion of silica in the euphotic zone during the summer--this shift is the replacement of diatoms by blue-green algae as the phytoplankton dominant. These blue-greens, however, are not *Anabaena flos-aquae* or *Microcystis*, the most common nuisance forms, and the standing crops are not large due to the relatively low levels of phosphorus in the system.

It is possible that much of this change is relatively recent, perhaps the most serious changes have occurred in the last 20 years. Nuisance species of *Stephanodiscus*, *S. hantzschii* and *S. binderanus*, did not appear in the phytoplankton at the Chicago water intake prior to 1956. In the mid-1950's (Ayers et al. 1958) and in the early 1960's (Risley and Fuller 1965), reported levels of silica during the summer that would not limit diatom growth. But by 1969, levels over the entire lake basin were no greater than 0.2 mg SiO_2/liter in the epilimnetic waters (Schelske and Callender 1970). Further evidence for a relatively recent effect caused by increased inputs of phosphorus comes from recent mathematical modelling of phytoplankton growth in Lake Ontario (Thomann et al. 1975). This model indicates that three detention times or 24 years are needed to reach a steady state. In Lake Michigan it would be more appropriate to use three residence times for phosphorus or approximately 18 years as the time needed to reach equilibrium. The model suggests that effects of increased phosphorus loading will not be manifested in the lake immediately and would have been delayed from the first large increases in loading, probably in the 1940's.

If there is a significant delay between the time of phosphorus inputs and the effects produced in the biological system, then one can accept the hypothesis that major changes in the open-water phytoplankton have occurred in the past 20 years. If this is true it may also be one of the reasons why there has not been a measurable effect in the benthos and other components of the biological system.

Assessing differences or changes in trophic state of oligotrophic waters from biomass of phytoplankton or concentrations of chlorophyll *a* may not be practical as relatively large differences or intensive sampling may be required. It is not certain that a statistically significant increase of 0.5 mg/m^3 in the spring maximum of chlorophyll *a* could be detected on a lakewide basis as this would represent an increase of 20 percent. It seems obvious more sensitive techniques are needed. One technique that offers increased sensitivity is the use of environmental parameters that integrate environmental processes. Hypolimnetic oxygen depletion has been used for this purpose in shallower lakes, but there is little change in hypolimnetic oxygen concentrations in Lake Michigan. Schelske (1975) proposed that silica and nitrate depletion in the euphotic zone during summer stratification could be used to assess trophic state.

Chemical changes or utilization of nitrate and silica in the upper Great Lakes are related to trophic state in the upper Great Lakes. Reserves of these nutrients in surface waters (photic zone) are depleted by phytoplankton during summer stratification. Reserves, as indicated by differences in concentrations between bottom and surface waters, are related inversely to trophic state: Lake Superior, the most oligotrophic, has the greatest reserves, and Lake Michigan, the most eutrophic, has the smallest reserves (Fig. 5).

In conclusion, one would expect eutrophication of Lake Michigan to be a function of increasing phosphorus concentrations. Unfortunately adequate data are not available for consideration. The rate of silica depletion may be the most sensitive means of assessing changes in trophic state since the dominant phytoplankton are diatoms. Diatoms require silica for growth, so as nutrient enrichment increases standing crops of diatoms utilize silica in increasing amounts and rates. Even though silica is depleted in the summer, rates or annual quantities of silica utilized by diatoms can still be used to assess trophic state. Rates of depletion would have to be calculated for the spring bloom period. Finally, if eutrophication of Lake Michigan continues, the total amount of silica in the lake would continue to decline. Conversely, if eutrophication is reversed, the total amount of silica in the lake should remain constant or increase.

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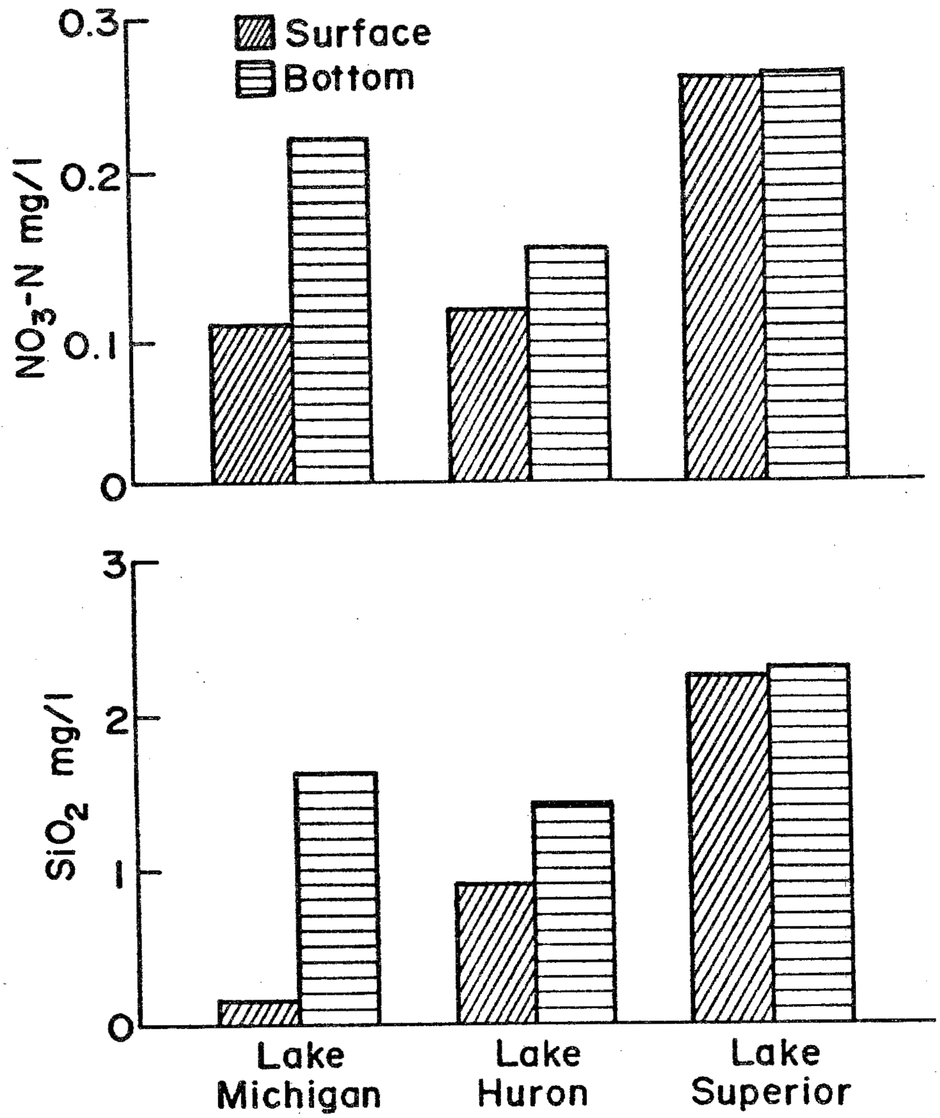

Figure 5. Silica and nitrate in the upper Great Lakes. (From Schelske 1974.)

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