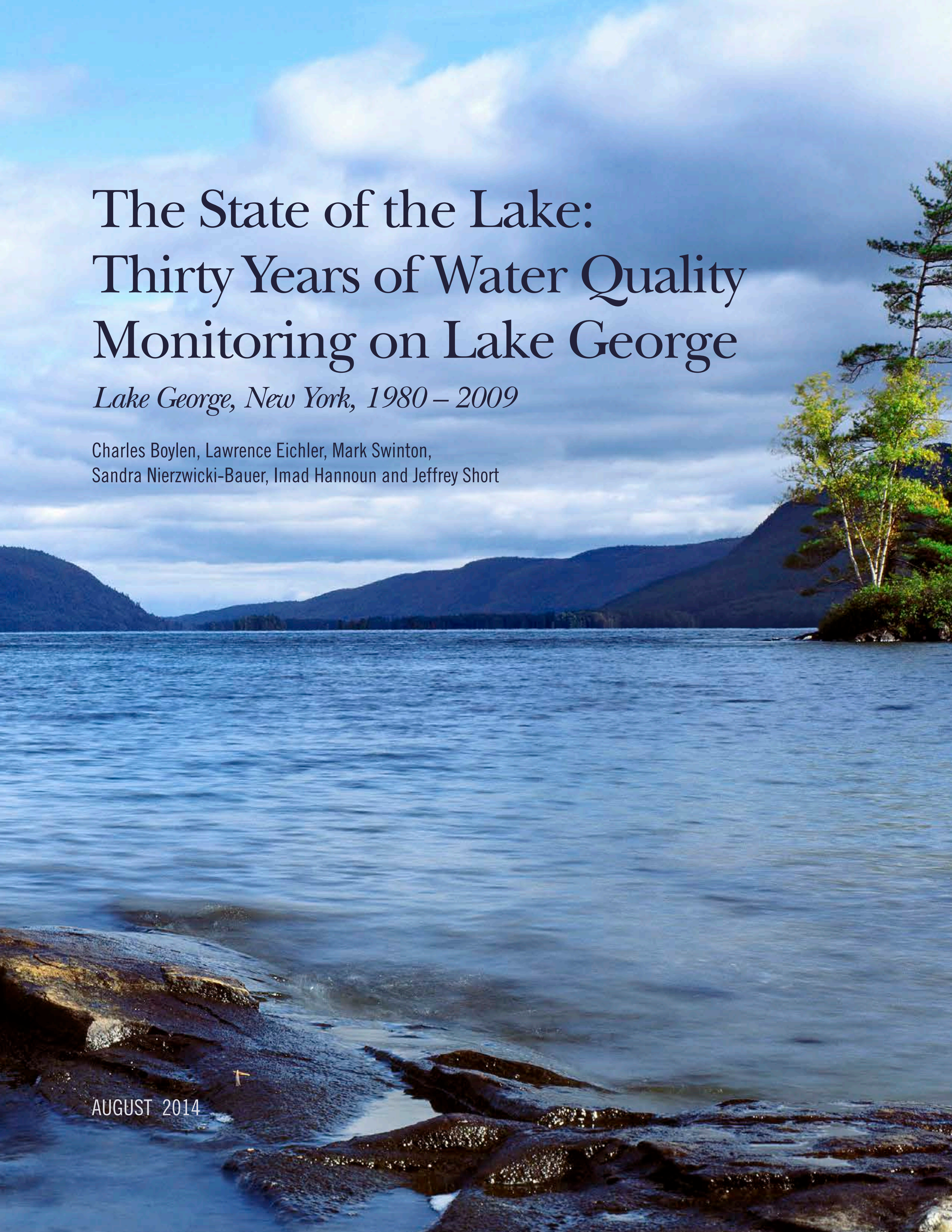

The State of the Lake: Thirty Years of Water Quality Monitoring on Lake George

Lake George, New York, 1980 – 2009

Charles Boylen, Lawrence Eichler, Mark Swinton,
Sandra Nierzwicki-Bauer, Imad Hannoun and Jeffrey Short

AUGUST 2014

Cover photograph courtesy of Carl Heilman II, www.carlheilman.com.
Designed by Tamarack Media Cooperative.

The State of the Lake: Thirty Years of Water Quality Monitoring on Lake George, New York, 1980 – 2009

Charles Boylen[1], Lawrence Eichler[1], Mark Swinton[1],
Sandra Nierzwicki-Bauer[1*], Imad Hannoun[2] and Jeffrey Short[3*]

August 2014

[1] Darrin Fresh Water Institute & Department of Biological Sciences, Rensselaer Polytechnic Institute, 5060 Lakeshore Drive, Bolton Landing, NY 12814

[2] Water Quality Solutions, Inc., 1726 Three Springs Road, McGaheysville, VA 22840

[3] JWS Consulting LLC, 19315 Glacier Highway, Juneau, AK 99801
* Corresponding Authors

The DFWI Offshore Chemical Monitoring Program has been jointly funded by Rensselaer Polytechnic Institute and The FUND for Lake George.

DFWI is grateful for the steadfast support of the David M. & Margaret A. Darrin endowments, including the David M. Darrin endowment to support summer interns, and for support from the United Parcel Service endowment. Both DFWI and The FUND for Lake George wish to give special recognition to the Helen V. Froehlich Foundation for its long term and ongoing support. The dedicated role of our many contributors has been essential to the progress we have made in understanding the state of Lake George water quality and its changing nature over time.

We appreciate the thoughtful comments of Dr. Jay Bloomfield, Dr. Nicholas Clesceri, Dr. Stephen Norton, and Dr. James Sutherland on an earlier draft of this report. However, the final views and opinions presented herein remain solely those of the authors. In addition, we want to thank past and present FUND trustees, especially Dr. Margaret Schadler, Dr. Robert Dickson, and other members of the Lake Science Committee, as well as Dr. Carol Collins.

Rensselaer

Darrin Fresh Water Institute
A Research Center of Rensselaer Polytechnic Institute

THE FUND *for* LAKE GEORGE

CONTENTS

FIGURES & TABLES

FIGURES

TABLES

I. EXECUTIVE SUMMARY

ON HIS VISIT DURING THE SPRING OF 1791, the stunning natural beauty and exceptionally clear waters of Lake George led Thomas Jefferson to declare it to be "without comparison, the most beautiful water I ever saw." Centuries later, the lake's natural character remains largely intact, providing the basis for a nearly $1 billion annual local tourist economy. Lake George is a headwater lake with limited human perturbation, contributing to its State of New York water quality rating of Class AA-Special. It serves as the primary drinking water supply for both residents and visitors. Approximately 90 to 95 percent of the Lake George watershed remains as natural forestland, 46 percent of which is "forever wild" state-owned Forest Preserve.

Increasing development has, however, had environmental consequences, raising concerns regarding the ecological integrity and resilience of the lake, especially over the last 50 years. The most serious include eutrophication from wastewater and urban runoff, the toxicity of pesticide residues on the lake's fish and other biota, acidification from atmospheric pollutants, salt loading from road de-icing in winter, and the effects of invasive species, introduced inadvertently or deliberately in the case of several fish species. Together these concerns have engendered sustained scientific efforts to understand their effects, including an extensive water quality monitoring program begun in 1980. Conducted by Rensselaer Polytechnic Institute's Darrin Fresh Water Institute (DFWI) and with financial support from The FUND for Lake George, the core results from 30 years of chemical surveillance of the lake waters are summarized herein.

At the outset of the DFWI monitoring program, the major concerns identified by the scientific community were associated with nutrient loading and with introduced fish species. Research conducted at Lake George by the New York State Department of Environmental Conservation produced the first evidence of the adverse effects of DDT on fish anywhere in the world, but national restrictions on this pesticide in 1972 led to dramatic reductions in its use. In contrast, loadings of phosphorus and nitrogen from septic tanks and sewage treatment plants, augmented by high-phosphorus detergents in domestic wastewaters were increasing the biological productivity of the lake, raising concerns that these loadings could lead to areas of low-oxygen water on the lake bottom during early fall resulting in inhospitable conditions for fish, blooms of nuisance plants near shore and of toxic phytoplankton farther offshore, and a general decline of water clarity, all of which threatened the local economy. Research conducted during the 1960s and 1970s indicated that nutrient loadings had at least doubled over the natural background levels prior to European settlement, and as an oligotrophic (*i.e.* low-nutrient) lake, Lake George was especially susceptible to strong biological responses. By the end of the 1970s, regulations and other efforts to limit nutrient loadings from sewage, wastewaters and detergents sharply reduced phosphorus loading, but the efficacy of these efforts was not yet clear by 1980.

Alteration of the fish community posed another potential threat to Lake George. Introduced rainbow smelt became established as a self-sustaining population in the early 1970s, and were later suspected of causing a shift in the composition of the phytoplankton community by improving conditions for cyanobacteria (or blue-green algae). Rainbow smelt prey on large-bodied zooplankton more efficiently than the indigenous fish species in Lake George, and the reduced zooplankton population consumes less phytoplankton, increasing competition for scarce nutrients among phytoplankton species. In the nutrient-poor waters of Lake George, this increased competition may have favored cyanobacteria, especially during late summer when nutrients are most scarce, because cyanobacteria can grow at lower nutrient levels than many other phytoplankton species. The less nutritious cyanobacteria are preyed upon by fewer zooplankton species thereby reducing the food source for forage fishes including rainbow smelt that are in turn consumed by game fish. Whatever the cause, this shift clearly occurred during the 1970s, and because cyanobacteria are less nutritious than diatoms, the result may have lowered the lake's carrying capacity for fish.

The DFWI monitoring program was designed to assess the water quality of Lake George, in particular to help address concerns related to nutrient loading. Sampling focused on phosphorus, nitrogen and silica compounds that are essential nutrients for aquatic plants, on the major cations including sodium and the biologically important potassium, calcium and magnesium ions, along with their most common anions including chloride, bicarbonate and carbonate, and sulfate, to help infer sources. Oxygen and chlorophyll were measured because they reflect respiration and photosynthesis, the two most important biological processes within the lake, and temperature was measured because it governs the rates of these processes, how readily water masses within the lake mix, and the solubility of oxygen and of carbon dioxide in water. These measurements were made at biweekly intervals during spring and fall, and at monthly intervals during summer, at a core set of six mid-water stations: two in the north basin, three in the south basin, and one in the constricted channel connecting the two basins. Samples were collected using a depth-integrated sampler from the surface to 10 m depth with grab samples taken at 20 – 30 m depths, typically within 1-m off the lake bottom to characterize the hypolimnetic water. Temperature and oxygen were measured at discrete depths extending to 30 m to characterize the density structure of the water column and its oxygen concentration.

The clearest trend evident from the monitoring is a steady rise in sodium chloride, nearly tripling from about 9 mg NaCl L^{-1} in 1980 to nearly 26 mg NaCl L^{-1} in 2009. The salt comes from road de-icing applications within the watershed during winter, and was well above the background of < 1 mg NaCl L^{-1} typical of Adirondack lakes in undeveloped watersheds. Sodium chloride has displaced calcium carbonate as the dominant salt in the lake, and is now near or possibly above the threshold for altering the phytoplankton community composition based on statistical analysis of diatom presence in sediment cores. Continued increases of salt loading in the watershed may also affect the circulation of the lake, which might result in subtle alteration of its biological functioning.

Concentrations of total phosphorus had declined by 1980 to around 60% of the concentrations measured during the 1960s and 1970s, and remained nearly constant thereafter through 2009, indicating that measures to reduce loading to the lake during the 1970s were effective. Concentrations of total nitrogen had also declined, with additional declines in the 1990s resulting from Clean Air Act restrictions on nitrogen and sulfur oxide emissions from combustion sources nationally. These restrictions led to a concurrent decline of sulfate in Lake George, and an associated decrease of acidity and increase of alkalinity (or acid-neutralizing capacity of the lake water).

Despite the apparent stability of phosphorus in the lake since the early 1980s, chlorophyll increased by about 33%. During this same period average water clarity measured as Secchi depth decreased by half a meter, or about 6%. At least four potentially interacting factors may have contributed to these changes. The gradual chlorophyll increase may reflect a phytoplankton response to a slowly rising increase of nutrient loading associated with development in the watershed that is too small to detect in the phosphorus monitoring. Alternatively, changes in the composition of the lake's fish community may have increased predation on zooplankton, thereby continuing to reduce grazing pressure on phytoplankton, resulting in more abundant phytoplankton standing stocks that reduce water clarity. Also, a shift in the community composition of the phytoplankton might have increased the annualized seasonal average of chlorophyll per unit phytoplankton biomass, leading to higher measurements of annual average chlorophyll measurements that are not commensurate with increases of phytoplankton biomass. And finally, Lake George hosts a submerged meadow of macroalga (*Nitella* spp.) that covered an estimated 14 - 20% of the lake bottom during the late 1970s. *Nitella* can compete with phytoplankton for nutrients such as phosphorus and nitrogen from the water column; consequently, if the *Nitella* meadow has declined since the 1970s it may have increased nutrient availability to phytoplankton. Understanding how these factors interact to cause the observed changes in chlorophyll could improve management of the lake considerably, through curtailing further losses of the lake's water clarity and through possible enhancement of the lake's capacity to support fish. Future research and monitoring efforts should focus on improving this understanding.

Temperature measurements of surface waters across the lake clearly show an average increase of 1.8 °C (3.2 °F) from 1980 to 2009, reflecting warming trends observed throughout North America. In the northern hemisphere this trend increases in strength at higher latitudes, as less persistent snow cover converts the landscape to a more absorptive surface for solar radiation. The effects on Lake George may include a longer growing season for the aquatic plants, possibly leading to less oxygen in surface water during summer, faster metabolic rates of fish and invertebrates including zooplankton and phytoplankton, and fewer days of ice cover on average during winter.

Oxygen measurements reflect a classic pattern of seasonal variation for lakes of similar size, depth and latitude as Lake George. The water column remains almost completely saturated with oxygen throughout winter. Development of the thermocline during spring eventually isolates the deeper waters of the lake from the surface water by creating a density barrier that impedes vertical mixing. As the thermocline deepens during late summer, the biological demand for oxygen caused by decaying plant and animal matter depletes the oxygen beneath the thermocline. If too much decaying organic matter concentrates in a limited volume of bottom water, the biological demand can deplete oxygen concentrations to levels inadequate for respiration by fish. Oxygen levels inhospitable for fish usually occur near the bottom of Caldwell Basin during late summer and early fall. In addition, hypoxic sediments release phosphorus to the overlying waters of Caldwell Basin, providing an added increment of phosphorus that stimulates phytoplankton productivity during fall. The extent of these low-oxygen waters vary considerably from year to year, but warrant careful monitoring to provide early warning of a long-term increasing trend.

Overall, Lake George remains in remarkably good condition environmentally, and for the most part, is reasonably stable with respect to adverse trends that could degrade water quality. Threats to water quality of greatest concern include the rising concentrations of salt from continued applications to control winter road ice, the high sensitivity of the lake's phytoplankton abundance and community composition to even modest increases of nutrient loading, and changes in how the lake's food web responds to alterations in the lake's fish community through stocking programs, or in response to invasive species. Previous efforts to protect the lake's water quality, both regulatory and otherwise, have thus far largely succeeded in averting serious degradation from environmentally persistent pesticides such as DDT, from nutrient-stimulated eutrophication, and from acidification by atmospheric pollutants.

Thirty years of water quality monitoring has provided a fundamental understanding of changing lake health as it has also identified serious questions requiring answers to prevent ecological decline. Over the coming decades, increasing pressure from development, use, and a changing climate will challenge the lake's natural resilience. Enlightened management, informed by rigorous science focused on identifying and remediating new threats, can ensure Lake George continues to provide an example of how to protect a natural resource that serves as an economic cornerstone and world-class destination for present and future generations of both residents and visitors.

II. INTRODUCTION

AFTER AN INTENSIVE PERIOD OF STUDIES supported by the National Science Foundation's International Biological Program (IBP) during the 1960s and '70s (Ferris and Clesceri 1974), the Darrin Fresh Water Institute (DFWI) developed a lake assessment program in partnership with The FUND for Lake George (Long et al. 1982a). The Offshore Chemical Monitoring Program began in 1980 to primarily assess physical and chemical characteristics of Lake George that continued in fundamentally the same way, with some expansion, to the present. The original stations have been sampled at the same depths and at largely the same times, and employing the same sampling and analytical protocols. This has produced a remarkable long-term data set that can be used to assess environmental change within the lake on decadal time scales, and is one of the longest and most consistent sets of such measurements in North America, providing a benchmark for long-term change in Lake George.

Environmental perturbation of Lake George from human activities began centuries ago. Lake George was already altered significantly well before the current monitoring program started, and an understanding of responses to these perturbations, both scientific and managerial, provides helpful context (see Appendix I for an historical summary of Lake George). During the 1960s and 1970s, increasingly serious concerns nationwide about declining water quality prompted numerous studies aimed at evaluating the effects of nutrient overloading. One of the most important reasons Lake George was selected for monitoring under the IBP was its modest extent of environmental degradation in comparison with other North American lakes of similar size and latitude. This relatively limited degradation made it a favorable location for detecting environmental change, especially if environmental conditions are documented prior to substantive change. This was indeed the case for Lake George. Data produced by the IBP studies of Lake George provided clear evidence of increased phosphorus and nitrogen loading as human settlement within the watershed intensified, likely exacerbating development of hypoxia in the deepest parts of the lake's basin just prior to the fall turnover (Clesceri and Williams 1972, Ferris and Clesceri 1974, Siegfried and Quinn 1986).

As a lake of exceptional water clarity, Lake George seemed especially vulnerable to water quality degradation resulting from a strong phytoplankton response to small increases in nutrient loading. However, other IBP studies documented the extent and role of a large meadow of *Nitella* spp., an attached charophyte alga inhabiting the lake bottom at depths of 7 – 16 m (Needham et al. 1922, Stross et al. 1988). Unlike most submerged aquatic vegetation, *Nitella* obtains nutrients directly from the water column, thereby competing with phytoplankton for nutrients. *Nitella* was consequently thought to both depend upon and to maintain water clarity (Stross et al. 1988).

Growing appreciation of the lake's sensitivity to water clarity losses from increased nutrient loading prompted vigorous efforts in the late 1970s to identify and limit sources of nutrients in the watershed from inadequate septic systems and from sewage treatment plants (Aulenbach et al. 1981). These efforts were aided by regulatory limitation on the phosphorus content of detergents, a major source, by the State of New York in October 1976 (New York State Department of Environmental Conservation 1985). This set the stage for arresting further increases of nutrient loading into Lake George, followed by their subsequent decline.

Another potentially important event affecting the water quality of Lake George was establishment and subsequent enhancements of a self-perpetuating population of rainbow smelt (*Osmerus mordax*). Studies by Siegfried and co-authors provided substantial evidence that introduction of rainbow smelt to the lake may have altered the phytoplankton community through a trophic cascade effect. Siegfried and Quinn (1986) argued that smelt predation of large-bodied zooplankton reduced grazing on phytoplankton, and ensuing competition for nutrients favored cyanobacteria during late summer and early fall. This could have serious consequences for the lake's capacity to support higher trophic levels such as fish that have strong direct or indirect dependence on phytoplankton productivity. This effect would also likely result in overall increases of the chlorophyll standing stock in the lake. Concerns regarding progressive eutrophication, alteration of the *Nitella* meadow and the consequences of the rainbow smelt introductions provided much of the motivation for continuing to monitor the water quality of Lake George after funding for the IBP studies ended in 1974.

Three other concurrent lake monitoring and research programs provide additional context for interpreting the results of the Lake George monitoring program. Research conducted at the Experimental Lakes Area in Ontario, Canada demonstrated the paramount role of phosphorus loading in causing eutrophication (Schindler and Fee 1974). A water quality monitoring program beginning in 1992 evaluated effects of phosphorus and other nutrient loading in Lake Champlain, as well as effects of acid rain from fossil fuel combustion (Vermont and New York State Departments of Environmental Conservation 2012). Acid rain concerns also prompted creation of the Adirondack Lakes Survey Corporation (ALSC) in the early 1980s to monitor chemical and biological responses of wilderness and lightly-developed Adirondack lakes (Baker et al. 1990a,b). Comparison of monitoring results from ALSC lakes and Lake George permits evaluation of effects from watershed development in Lake George. Likewise, because Lake George is relatively small compared to Lake Champlain and serves as a relatively undeveloped headwater basin, comparison of results may provide insights regarding processes and trends in both lakes after accounting for their differences geologically.

This report provides a synthesis of the Lake George monitoring results, with the primary goal of summarizing the data to distill the major patterns and their inter-connections. While anything approaching an exhaustive synthesis might easily take years, we have attempted here to glean the most prominent and readily accessible trends reflected by the data spanning the period from 1980 through 2009. In the following sections, we first describe the current environmental setting of the lake and its general hydrology, followed by sections summarizing trends in major ions and conductivity, nutrients, chlorophyll and clarity, and temperature and oxygen. Finally, we present conclusions and recommendations for future monitoring and research, recognizing that the monitoring to date has provided an unparalleled foundation for continued limnological research on Lake George.

III. THE LAKE GEORGE WATERSHED

NATURAL HISTORY

The Lake George basin was created by the Wisconsin Glaciation, and began accumulating sediment after deglaciation around 15,000 years ago (Hutchinson et al. 1981). Like many glacially-formed lakes across the Adirondack Park, Lake George is oriented on a north-northeast axis and drains to the north. The glacier changed drainage patterns to create a lake with a single northern outlet out of what had been two river systems flowing in opposite directions. One stream originated in what is now Northwest Bay Brook and flowed south to the Hudson River; the second flowed from the Narrows northward into Lake Champlain (Newland and Vaughan 1942). During glacial recession, three glacial lake stages in the southern portion of current Lake George with southern flow may have existed. Several layers of sand below Lake George Village are believed to be a result of delta formations from the three lake outlets. With the glacier's final retreat these southern outflows became dammed by their own deposits (Chadwick 1928 in Shuster 1994). Lake George now drains to the north with a single outlet at the LaChute River, descending 69 m to Lake Champlain.

GEOLOGY AND SOILS

The Lake George watershed consists predominately of pre-Cambrian rock, with small patches of Cambrian bedrock at the southern end (Shuster 1994). Bedrock geology indicates the Lake George basin is a mix of dominantly granitic gneisses, charnockitic gneisses, garnet-biotite-quartz-plagioclase gneisses, quartzites, meta-anthrosites, and metagabbros with smaller quantities of marble, calcsilicates and amphibolites. Most of the watershed is covered with shallow sandy till overlaying bedrock with numerous granite outcrops and large boulders. The sandy tills have high hydraulic conductivities and rapid groundwater infiltration rates. The northern portion of the watershed has more fine silts and clays associated with deposition from seasonally melting glaciers. These overburdens have much lower hydraulic conductivities (McClelland and Isachsen 1986, Shuster 1994).

The bottom sediments of Lake George include three major units, defined as undifferentiated till, glaciolacustrine clay and Holocene lake deposits (Hutchinson et al. 1981). Glacially deposited sand and gravel occurs mostly on the west side of the lake and in the deep bedrock basins. Glaciolacustrine clay formed deposits up to 30 m thick in the deepest basins, but eroded in water depths less than 20 m. Holocene muds, rich in organic matter (~10%), generally accumulate in water depths greater than 30 m and form thick layers, up to 15 m, in the deep basins.

PLANTS AND ANIMALS

The Lake George watershed is surrounded by northern upland forests that contain a mix of coniferous and deciduous trees. The watershed supports a mixture of 38 native coniferous and deciduous tree species, 30 native shrub species and 49 species of other native plants. Dominant trees include sugar maple (*Acer saccharum*), yellow birch (*Betula alleghaniensis*) and beech (*Fagus grandifolia*), with lesser amounts of white ash (*Fraxinus americana*), hemlock (*Tsuga canadensis*), red maple (*Acer rubrum*), white and red oak (*Quercus alba* and *Q. rubra*), and white pine (*Pinus strobus*; Nicholson et al. 1979). There are 54 species of mammals that live around the lake and over 200 bird species. The lake supports 47 aquatic plant species (Ogden et al. 1976), including five classified as rare, threatened or endangered, and two that are classified as invasive (Young 2010).

The Lake George watershed supports at least 36 species of fish, of which 11 are non-native (George 1981). The Lake George fishery is classified as two-story, indicating the presence of both cold and warm water species. In recent years, management has focused on stocking programs for 14 fish, including rainbow trout (*Salmo gairdneri*), brook trout (*Salvelinus fontinalis*), lake trout (*Salvelinus namaycush*) and Atlantic salmon (*Salmo salar*). George (1981) reported that rainbow smelt (*Osmerus mordax*) were introduced in the 1970s based on their presence in streams. Natural reproduction of land-locked salmon is assumed to be minimal due to the limited tributary area available for spawning, thus stocking is required to support this put-and-take fishery. An extensive area of spawning habitat for lake trout is protected from fishing pressure allowing a naturally reproducing population.

WETLANDS

Large wetland systems border Lake George on its southeast margin, Dunhams Bay (527 ha), and at the head of Northwest Bay (162 ha). Smaller wetlands of varying sizes dot the Lake George shoreline in Warner Bay, Huddle Bay, East Brook and the Shelving Rock area (Fig. 1a). These emergent wetlands are frequently or continually inundated with water, characterized by soft-stemmed vegetation, such as grasses, sedges, and forbs that live in shallow water. Plants adapted to the moisture-saturated soil conditions of wetlands include cattails (*Typha*), bulrushes (*Scirpus*), pickerelweed (*Pontederia*), and arrowheads (*Sagittaria*) (Ogden et al. 1976).

Another important wetland type in the Lake George watershed is a deep water marsh, which is a permanently flooded area that does not exceed a seasonal water depth of six feet and is defined by free floating vegetation, rooted vegetation with floating leaves, or submerged vegetation. This type of wetland occurs in many locations along the shoreline of Lake George. Duckweed (*Lemna*), water lilies (*Nuphar* and *Nymphaea*), coontail (*Ceratophyllum*), bladderwort (*Utricularia*), wild celery (*Vallisneria*) and pondweeds (*Potamogeton*) grow in this type of wetland. It may be an important food source for waterfowl and is a critical area for fish spawning and nurseries.

POPULATION AND THE BUILT ENVIRONMENT

The last assessment based on 45-year old maps indicated approximately 3% of the watershed area as developed, with 75% of the development concentrated in the south basin (Hetling 1974, Sutherland et al. 1983). Developed areas are largely limited to the southern and western shores of the lake. Developed areas have expanded considerably since the 1980s, and may now cover perhaps as much as 10% of the watershed.

Eight towns, one village and three counties border Lake George. These are largely rural communities. The Town of Queensbury is the largest municipality with a population of 25,441 in the 2010 census. The Town of Bolton has a population of 2,117, Lake George 3,578, Hague 854, Ticonderoga 5,200, Putnam 645, Dresden

677 and Fort Ann 6,417. Many of these residents live outside the Lake George watershed as the towns are not completely within the watershed. During the summer months, the Lake George watershed experiences a large increase in population from seasonal residents.

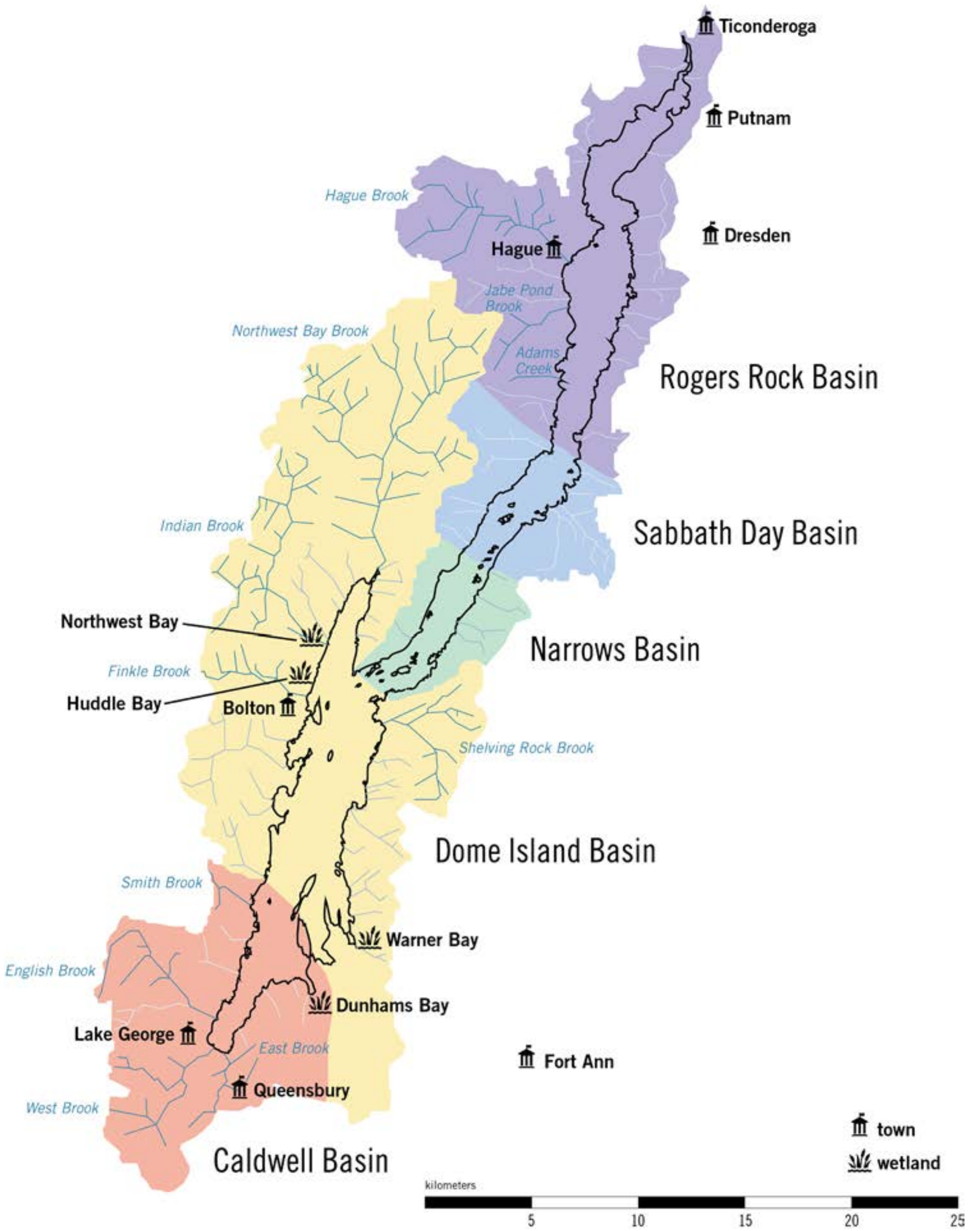

FIGURE 1a. *Lake George watershed sub-basins.*

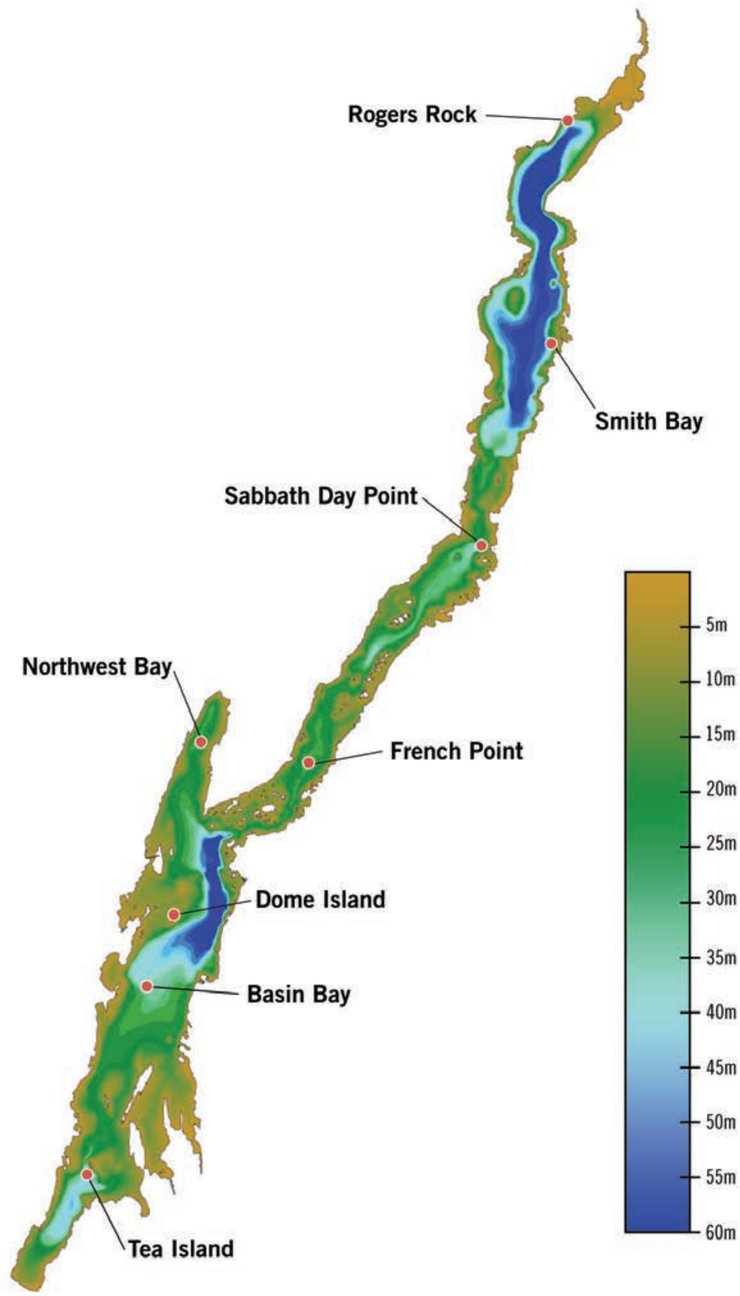

FIGURE 1b. *Lake George bathymetry, and locations of the eight sampling stations, six of which were monitored from 1980 through 2009 to the present.*

There are over 1,090 km of roads within the Lake George watershed including 178 km (16.3%) of county roads, 646 km (59.3%) of town and village roads, and 266 km of state and federal roads (24.4%). In 2009, there were 9,009 buildings in the watershed, with 3446 built between 1980-2009 (a 62% increase) and with 82% of new buildings occurring in the south basin[1].

1 Data for the land use was obtained from analysis of the assessment rolls for the eight towns around Lake George. Only data from areas included with the Lake George watershed was studied. Data was provided by the New York State Office of Real Property Services. In addition the assessors in the Towns on Lake George, Bolton and Hague were consulted as was the Code Officer in Ticonderoga, the Queensbury Planning Office and the Real Property Offices in Washington and Essex Counties, and the Building Codes and Fire Prevention Office in Warren County..

SOURCES OF LAKE GEORGE WATER

Overall, streams account for 57 percent of the lake's annual hydrologic budget, while 25 percent comes from direct precipitation on the lake and 18 percent from groundwater fed to the lake through underground springs (Shuster 1994). More than 141 streams drain the Lake George watershed, 92 of which are large enough to be mapped on the USGS 7.5 minute topographic maps (Madsen et al. 1989, Sutherland et al. 2001). The largest ten streams drain about 48% of the lake's land catchment (Table 1).

	Watershed Area (ha)	Percent of Land Catchment
Northwest Bay Brook	8,423	17.2%
Indian Brook	3,012	6.2%
Hague Brook	2,764	5.7%
West Brook	2,244	4.6%
English Brook	2,092	4.3%
Shelving Rock	1,889	3.9%
Finkle Brook	1,110	2.3%
East Brook	869	1.8%
Sucker Brook	636	1.3%
Foster Brook	476	1.0%
Total	23,537	48.2%

TABLE 1. *The ten largest tributaries to Lake George by watershed area (Lake George Association 2012).*

LAKE GEORGE PHYSIOGRAPHY

Lake George includes two large basins of nearly equal size to the north and south, separated by a shallow, island-studded area known as the Narrows (Table 2; Aulenbach et al. 1981). The lake contains five major catchment sub-basins (Fig. 1a), two in the south (Caldwell and Dome Island) and three in the north (The Narrows, Sabbath Day and Rogers Rock). The lake is 97.5 m above sea level with a maximum width of 3.3 km, and lake level is regulated by the LaChute Hydro Company Inc. usually to within 1 m by a dam on the discharge stream, the LaChute River. The most recent compilation of bathymetric data is presented in Fig. 1b (Boylen and Kuliopulos, 1981). On average, the hydraulic retention time has been estimated to be 7 to 8 years (Shuster 1994, Colon 1972, Clesceri and Williams 1972). Other physical characteristics of the lake are summarized in Table 2.

	Entire Lake	South Basin	North Basin Including the Narrows
Volume (km^3)	2.10	1.02	1.08
Surface Area (km^2)	114	57.6	56.4
Shoreline (km)	210	76	134
Watershed (km^2)	606	371	235
Length (km)	51	22.4	28.6
Max. Depth (m)	58	58	53.3
Average Depth (m)	18	15.5	20.5

TABLE 2. *Basin characteristics of Lake George (from Aulenbach et al. 1981).*

IV. LAKE GEORGE HYDRODYNAMICS

THE TRANSPORT AND MIXING OF WATER MASSES WITHIN A LAKE determine the distribution and interaction of dissolved compounds with the atmosphere and with the lake bed. The internal circulation is largely determined by lake physiography and bathymetry, especially in a long, narrow lake such as Lake George, where surface winds blowing across a long fetch can induce relatively strong surface water currents. These processes provide essential context for interpreting the results of the monitoring program. This section provides a brief summary of the major hydrodynamic processes as influenced by the physiography and bathymetry of Lake George.

Lake George is a classic temperate dimictic lake, meaning it has two periods of vertical mixing alternating with summer and winter stratification periods. Stratification results primarily from the interaction of meteorological factors and the dependence of water density on temperature. The density of water achieves a maximum at 4°C. As water cools below or warms above 4°C, its density decreases, albeit at a non-linear rate. Another factor affecting the density of water is its dissolved solids content. At the same temperature, saline water has a higher density than fresh water. Changes of water density can have profound effects on circulation in deep lakes (Fischer et al. 1979). In addition to thermal stratification, there are several other factors that affect water circulation and movement. These include additional meteorological parameters such as diurnal temperature fluctuations, wind, inflow intrusions, outflow withdrawal dynamics, and the lake's complex bathymetry.

THERMAL STRATIFICATION AND WATER COLUMN MIXING

In Lake George, the water temperature is near 0 °C at the surface during winter, usually beneath ice cover, and increases with depth to 4 °C, the temperature of maximum water density, near the bottom. Heating during spring results in an isothermal water column that is readily mixed by wind-induced surface turbulence, and continued surface heating again stabilizes the water column as the surface water becomes warmer and hence less dense than the deep water. The water column continues to warm, establishing a well-defined thermocline at about 10 m depth during summer, which then deepens to about 20 m during maximum stratification and weakens by early fall. As the surface cools during fall, the density of the surface water increases, enabling wind mixing to cause another turnover. Continued cooling during late fall and winter increases the stability of the stratified water column, with water cooler than 4 °C floating above warmer water beneath.

Spatially uneven solar heating can generate horizontal currents near the surface. At night, surface water cooling may result in the formation of falling plumes that transport water (and other substances) vertically while entraining the water below, producing turbulence, and generating upward water movement to replace the descending plumes.

WIND-INDUCED CURRENTS

Winds blowing over a lake exert a shear stress on the water surface that causes waves to form, break and transfer momentum to the lake. A relatively strong sustained wind will cause the surface layer of water to accelerate in the general direction of the wind. Because the water surface remains nearly horizontal, a return reverse flow at depth is typically established (Fig. 2). A series of horizontally opposing flows at different elevations, attenuated with depth, have been modeled to occur in Lake Mead (Hannoun et al. 2005). The resulting vertical shear produces turbulent mixing, which leads to the deepening of the thermocline. Aside from initiating bulk water movement, the wind induces motions that can generate seiches and internal waves at the thermocline, with amplitudes in Lake George on the order of several meters or more, and periods on the order of hours to days (Manley 2006). With continued forcing, internal waves generally strengthen and amplify, then break and cause locally strong turbulence and intense localized vertical mixing (Hannoun and List 1988).

In Lake George, Haines and Bryson (1961) reported that Langmuir (1938) established that water velocities near Lake George's surface are typically about 2% of the wind speed. Given an approximately 8 km h^{-1} average wind speed in 2012 (www.wunderground.com), the resulting average surface water velocity is computed to be approximately 0.16 km h^{-1}, or 3.8 km d^{-1}. Applying the 2% wind to water velocity ratio to a sustained wind speed of 30 km h^{-1} yields surface water velocities on the order of 14 km d^{-1}. While these calculations are empirically based, they nonetheless show that strong sustained winds can significantly influence surface water transport. Such water velocities can move surface water across the length of the lake within a few days to a few weeks.

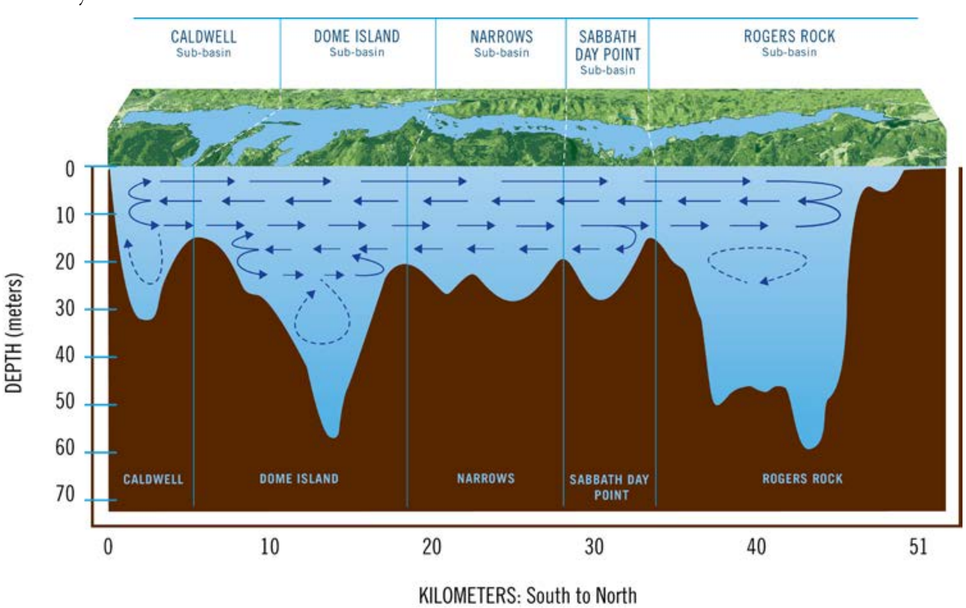

FIGURE 2. *Conceptual pattern of sub-surface currents induced by surface wind stress along a longitudinal bathymetric section of Lake George.*

These calculated average surface water velocities are of the same order of magnitude as those observed in rhodamine dye studies in the vicinity of the water intake pipe for the town of Lake George (Sklenar 1996, Manley 2006). While the location and movement of the dye transport plumes varied from 1994 to 1995, the time to reach the intake pipe was 13 and 26 hours (Sklenar 1996), yielding a horizontal velocity of 2 and 1 km d^{-1}, respectively, of comparable magnitude to the value of 3.8 km d^{-1} estimate based on Langmuir's general observations.

INFLOW DYNAMICS

Precipitation imparts momentum downwards while mixing with ambient surface water. The fate of precipitation water depends on the relative density difference between the precipitation and ambient waters. For example, a relatively cold rain in summer may drop within the water column to its point of neutral buoyancy, after mixing with ambient water.

A stream entering a lake will nearly always be at a different temperature and salinity, and thus density, than the surface lake water. Upon entering a lake, the initial momentum of a stream serves to entrain ambient water, until its velocity is diminished and buoyancy forces start to dominate. The inflowing water will then either flow over the surface as an overflow if it is less dense or descend towards the bottom as an underflow if more dense than the ambient water (Fischer et al. 1979). A submerged inflow will eventually find its level of neutral buoyancy. If a sufficiently strong thermocline is present as is the case from late spring to early fall at Lake George, stream inflow would insert near the level of the thermocline, about 10 to 15 meters below the surface.

Overall, groundwater entering the lake will have the same general dynamics as a surface inflow. Groundwater enters the lake below the water surface at varying but not well documented depths and at temperatures that do not vary greatly from the annual average air temperature of about 8.3°C regardless of season (www.epa.gov/athens/learn2model/part-two/onsite/ex/jne_henrys_map.html). Assuming similar dissolved solids content, groundwater entering the lake in winter is therefore likely to be significantly warmer than the lake's ambient water, and thus rises in the water column while entraining ambient water. When a thermocline is present, deep groundwater entering the lake below the thermocline will likely have a temperature significantly warmer than the ambient, so it is also expected to initially rise while mixing with ambient water. Manley (2006) reports on measuring a gradual warming of the hypolimnion in the north basin of the lake in the summer of 2004, which may be related to the inflow of deep warm groundwater. Groundwater entering the lake above the thermocline in summer and early fall would be cooler than the ambient water, and thus descend along the bottom to insert in the vicinity of the thermocline.

OUTFLOW DYNAMICS

The major flow out of Lake George is through the LaChute River at the north end of the lake. Because Lake George is typically operated at a near constant surface elevation, the river flow is very seasonal with the highest flows usually occurring in the spring after snow melt. Outflows in the summer can be less than 10% of the maximum flow during the spring snow melt (Shuster 1994), a result of increased lake evaporation and increased ground infiltration in the watershed.

When the lake is not stratified in the spring or fall, the flow pattern within the lake generated by the LaChute River outflow can be described by potential theory of classical fluid mechanics, which requires the water to flow radially towards the outlet, equally from all vertical and horizontal directions. In the stratified period, however, the density differences in the water column are expected to strongly influence the outflow withdrawal layer and confine it to the region above the thermocline. Effectively, the outflow is expected to establish a near surface current that pulls water from the epilimnion of the entire lake. Because Lake George comprises two basins of nearly equal surface area and volume (Table 2) separated by the Narrows, approximately half of the outflow will originate in the epilimnion of the south basin, thus establishing an overall south-to-north current whose strength varies widely with season. As a result, the influence of the outflow on lake circulation can be expected to be seasonal.

LAKE BATHYMETRY

The unique bathymetry of Lake George can significantly influence water circulation. In the top 20 m of the lake, above the sill near the north end of the Narrows (Fig. 2), horizontal transport of water in the entire lake is expected to occur freely, and driven by the various effects described above (wind, inflows, outflows, etc.). Below the 20 m level, horizontal water movement between the south and north basins is inhibited by the bathymetry at the Narrows.

OVERALL CIRCULATION

The overall circulation in Lake George depends on the combined effects of stratification, wind, inflow and outflow volumes and dynamics interacting with basin bathymetry. The relative influence of these factors can change rapidly. For example, the largest effects of inflows and outflows on circulation occur during the spring snow melt season when the stream and river flows peak. In contrast, the effects of wind are negligible when the lake surface is ice covered, but otherwise are amplified during storms and high wind episodes. To best capture the detailed circulation in Lake George, a detailed three-dimensional circulation model is needed. Such a model would serve to integrate current understanding of all the processes that affect circulation in the lake, and would provide a basis for anticipating the effects of physical perturbations in the future. In the following subsections, possible flow regimes that can influence the rate of water transport and mixing in the lake during both stratified and unstratified periods are discussed.

STRATIFIED PERIODS

During stratification, water transport in the epilimnion is likely on the order of a few kilometers per day, and substantially more during high wind events, as discussed above. At the thermocline, water typically moves horizontally as multiple density currents generated by the outflow and various inflows that have been previously inserted at that level. In the hypolimnion, water movement is significantly slower than that in the surface layer, because of isolation from the physical forcing factors near the surface. Below the 20 m level, horizontal motion between the south and north basins is inhibited by the bathymetry at the Narrows (Fig. 2). Hypolimnetic water from the south basin must first be entrained into the epilimnion before transport into the north basin can occur. Similar considerations apply among the somewhat isolated deep water pools within each basin (Fig. 2).

High winds can rapidly move surface waters in the general wind direction. A strong sustained northerly wind that propels warm surface water through the Narrows towards the south basin could generate a reverse flow at a lower elevation in the epilimnion that moves water back into the north basin (Fig. 2). The resulting high water velocities will cause mixing at the thermocline, and entrain bottom water. Due to the differing bathymetry in each basin, and possible spatial differences in wind speed and other meteorological factors such as rainfall, it may be possible to temporarily generate surface water with different salinities and temperatures on either side of the Narrows. After the wind subsides or reverses, buoyancy effects resume dominance with respect to wind forcing, and rapid horizontal movement to equalize the density of the water masses between the two basins may ensue. The speed of a horizontal density-generated current (Brooke Benjamin 1968) is given by:

$$U = c\sqrt{(gL\Delta\rho/\rho)}$$

Where:

U = Speed of the density current
c = A coefficient on the order of 0.7 to 1.4
g = Gravitational acceleration
L = Initial height of the density current
$\Delta\rho$ = Density difference across the density current, and
ρ = Reference density of water

For a density difference of 0.04 % (equivalent to about 2°C), and $L = 10$ m (typical thermocline depth), the resulting density current velocity is on the order of 0.7 kilometers/hour or 17 kilometers/day. The velocities of such a front can be further accelerated by a reversing wind and/or steep bottom topography. Fronts moving at speeds of up to 1.8 kilometers per hour have been observed in the north basin after strong reversing wind episodes (Manley 2006). While fast moving waters as described herein may be infrequent and dependent on the confluence of several factors, such events can rapidly move water across large portions of the lake and across the Narrows, enhancing water exchange between the basins.

UNSTRATIFIED PERIODS

The lake turnover in both spring and fall results in a nearly uniform water column vertically on each side of the Narrows. Due to the limited density difference, vertical mixing can be relatively vigorous and rapid, especially if the wind velocity is high. Following vertical mixing in each basin during each turnover period, horizontal water transport near the surface, induced by the wind and density variations, can rapidly move water across the Narrows at a rate of a few kilometers per day or more. The lake turnover, when coupled with other physical forcing factors provides a mechanism whereby water from various lake regions and depths can be transported and mixed with other previously isolated water masses in a matter of a few weeks or less.

Based on the above discussion, two major lake-wide mixing events are expected per year during the turnover periods. In the interim, significant surface exchange between the basins is expected, with limited inter-basin exchange below the surface layer.

V. METHODS

OFFSHORE CHEMICAL MONITORING PROGRAM

The Lake George Offshore Chemical Monitoring Program monitored eight deep water sites throughout the length of the lake (Fig. 1b). Monitoring began at six sites in 1980, with two more sites (at Basin Bay and Sabbath Day Point) added in 1995. This report will focus on the results from the original six stations because the longer data sets are amenable to time-series analysis. Biweekly sampling from April, approximately two weeks after ice breakup through mid-June allowed a more precise determination of the onset of thermal stratification and episodic inflows during spring. Monthly sampling occurred thereafter through September. Biweekly sampling was then reinstated to better characterize the fall turnover. When possible an additional winter sample was collected through the ice at the Tea Island and Smith Bay stations in February or March.

During each sampling event, Secchi depth and water column profiles of temperature and dissolved oxygen were measured. Hose integrated epilimnetic water samples from 0 –10 m were collected at all sites for chemical analysis. Analytes are listed in Table 3. Most chemical concentrations are reported as milligrams (ppm) or micrograms (ppb) per liter of the respective element. For example, total phosphorus is reported as micrograms per liter of the element phosphorus (P) not the phosphate (PO_4^{-3}) ion. One exception is alkalinity, which is reported as milligrams of calcium carbonate per liter.

Holding times and preservation were dictated by the standard method associated with each analyte. All samples were analyzed at the Darrin Fresh Water Institute in Bolton Landing. Phosphate compounds were determined as total phosphorus, total soluble phosphorus and orthophosphate. Total phosphorus and total soluble phosphorus include all compounds convertible into orthophosphate by persulfate oxidation and by sulfuric acid hydrolysis, respectively, and orthophosphate is the PO_4^{-3} ion.

Analyte	Method
pH	Electrometric - Method 4500-H+ (Standard Methods 1998)
Specific Conductance	Wheatstone Bridge type meter - Method 120.1 (US EPA 1979)
Alkalinity	Titrimetric – pH 4.5 Method 310.1 (US EPA 1979)
Chloride, Nitrate & Sulfate	Ion Chromatograph - Method 300 (US EPA 1979)
Chlorophyll	Fluorometric - Method 10200 (Standard Methods 1998)
Ammonia	Phenate Method -Method 4500-NH3 F. (Standard Methods 1998)
Soluble Silica	Molybdate Reactive -Method 4500-SiO2 E. (Standard Methods 1998)
Total & Total Soluble Phosphorus	Colorimetric – Persulfate Oxidation, Method 365.2 (US EPA 1979)
Orthophosphate	Colorimetric - Method 365.2 (US EPA 1979)
Total Nitrogen	Colorimetric – Persulfate Oxidation (Langner & Hendrix, 1982)
Major Cations & Metals (Ca, Mg, Na, K, Fe)	Atomic Absorption Spectroscopy – Flame, Method 3111 (Standard Methods 1998)

TABLE 3. *Methods used for the analysis of chemical analytes sampled from Lake George, 1980 – 2009.*

Quality control focused on evaluations of precision and accuracy. Precision was monitored by reanalysis of at least 15 percent of all samples. Reanalysis was performed in two ways, by multiple analysis of a single sample or analysis of individual samples subdivided prior to analysis (split samples). The use of duplicate and split samples, spikes and external check standards ensure the analyses conducted are accurate. If any of these QA/QC protocols were not within 10% of the expected value, samples were reanalyzed. The strict analytical protocols followed throughout the 30-years of study ensure that comparing data is valid for the entire time period. To determine the percent recovery for each ion analyzed, selected samples were spiked with a standard of known concentration after initial analysis. The difference between the observed and expected concentrations of this spiked sample gives the percent recovery.

Type of QA/QC Procedure	Frequency
Analysis of replicate samples	5% (one in twenty samples)
Analysis of split samples	5% (one in twenty samples)
Analysis of blank (DI water) samples	Daily
Inter-laboratory exchange of samples	Annually
Analysis of unknown samples	Quarterly
Control Chart analyses	Daily

Replicate: sample analyzed twice = duplicate
Split: single sample divided into two samples in the field

TABLE 4. *Laboratory Chemical Analyses - QA/QC Procedure Frequencies.*

Analytical accuracy was determined by analysis of samples of known concentration purchased from a National Institute of Standards and Technology (NIST)-traceable supplier. At least one sample of known concentration was included for each ion analyzed on a daily basis. Control charts were maintained for each analyte and control limits were monitored daily. The frequencies of quality control procedures are listed in Table 4. Ionic balance provides a rigorous test of analytical accuracy, with the sum of cations agreeing with the sum of anions within an average deviation of ~2 percent.

Since 1987, the DFWI Laboratories have been certified by the New York State Environmental Laboratories Approval Program. Certified analytes include a variety of inorganic chemical and microbiological assays. The DFWI Laboratories also have been involved in interlaboratory sample exchange programs with the New York State Department of Health, the New York State Department of Environmental Conservation (NYSDEC), and a number of public and private laboratories. The low ionic strength chemistry of Adirondack lakes necessitates inter-laboratory exchanges with other laboratories specializing in this area.

CLIMATE AND ATMOSPHERIC DEPOSITION

The Cedar Lane Atmospheric Deposition Station was established by the NYSDEC in 1980 and is located near the intersection of Beach Road and Cedar Lane, in the Town of Lake George. The station was in operation until 1984, when it was discontinued until re-established by the Department in 1991. In 2006, the DFWI assumed daily operation of the station with financial support from The FUND for Lake George. The amount and composition of precipitation that falls on Lake George, including sulfate and nitrate, pH, and airborne phosphorus loading is monitored at this station. In addition to a continuous record of the amount of precipitation (wet and frozen), wetfall, dryfall, and bulk deposition samples were collected for chemical analysis during the early 1980s. Wetfall, dryfall, and precipitation volume have also been collected at this site since sampling resumed in 1991. In 2001, a bulk collector was installed, and dryfall collection discontinued in 2003.

Precipitation volume is collected with a Qualimetrics, Inc., Model 6021A, tipping bucket rain-snow gauge that tips once for each 0.01 inches of wetfall. An Aerochem-Metrics Inc. wet/dry deposition collector and bulk precipitation collector gather deposition samples. Wetfall and bulk sample analytes are listed in Table 3. The wetfall bucket is collected after every rain event of approximately four tenths of an inch or more, or after several smaller events, so there is sufficient volume to process for all analytes. Bulk samples are collected every three weeks.

STATISTICAL ANALYSIS

Most statistical comparisons employ simple Student t-tests on un-transformed data. Paired comparisons are performed for samples collected from different stations at nearly the same times, otherwise comparisons are un-paired tests assuming equal variance. Unless otherwise indicated, uncertainty of estimated means is presented as 95% confidence intervals derived from standard deviations calculated from the underlying data, along with the sample size. The standard deviations may be recovered by multiplying half the confidence interval by the square root of the sample size and dividing the result by 1.96.

Correlations were calculated as simple product-moment correlation and are summarized as coefficients of determination calculated as the square of the correlation coefficient.

Time-series analysis was performed for 0 – 10 m integrated sample records that were sufficiently long (> 25 years) and with similar sampling frequency each year. An auto-regressive integrated moving average (ARIMA) analysis was performed in R. In these cases, we also present summary statistics to compare means, medians, quartiles and extreme values for stations. Further details regarding the time-series analysis methods are presented in Appendix II.

VI. MAJOR IONS AND CONDUCTIVITY

MAJOR IONS COMMONLY FOUND IN LAKE WATERS affect physical and biological processes, and can provide insights on water circulation. Such insights are especially important for Lake George where so little work has been done to understand its hydrodynamic circulation. In most lakes and rivers, four cation species (Na^+, K^+, Ca^{+2} and Mg^{+2}) and four anion species (HCO_3^-, CO_3^{-2}, Cl^- and SO_4^{-2}), account for nearly all of the ions present (Hutchinson 1957). In this section, we summarize results for these components, deferring our summary of nutrients (*i.e.* forms of phosphorus, nitrogen and silicon) to Section VII following. We also include conductivity in this section, because it serves as an integrative measure of all the ions present.

Calcium is usually the most abundant cation in lakes owing to chemical weathering of widespread calcareous rocks. While this was formerly true of Lake George, sodium chloride has now displaced calcium carbonate as the dominant dissolved salt, resulting from application of road de-icing salt during winter. Based on the New York State Department of Transportation (NYSDOT) annual application rate of 9.36 tonnes NaCl/lane-km an estimated 10,200 tonnes of salt were applied to the 1094 km of roads within the basin annually (Godwin et al. 2003). This is consistent with broader scale application rates of 200 kg ha^{-1} within the State of New York (Jackson and Jobbagy 2005) equivalent to 12,000 tonnes apportioned with the Lake George watershed. Actual application is likely somewhat less, around 8,000 tonnes (personal communication, Dave Decker). De-icing salt is typically composed of 90% to 98% sodium chloride with minor amounts of calcium, magnesium and potassium chloride along with an anti-caking agent, ferrocyanide (Titler and Curry 2011). Other sources of dissolved solids include chemical weathering of minerals in the watershed soils and bedrock, sewage discharges and urban surface runoff, and atmospheric deposition.

Addition of de-icing salt provides a fortuitous means of tracking water masses flowing into the lake and subsequent internal circulation. We therefore begin with consideration of results for sodium, potassium and chloride in some detail, focusing on chloride because it interacts least with organics and other soil components *en route* to or within the lake, and consequently serves as a conservative tracer of water movements. These results will help to interpret the behavior of many of the other parameters monitored.

CHLORIDE, SODIUM, AND POTASSIUM

The 0 – 10 m epilimnetic concentrations of chloride almost tripled over the course of the monitoring period. The lake-wide sample mean concentrations, averaged across the six monitoring stations, increased from 5.8 mg L^{-1} in 1980 to 15.9 mg L^{-1} in 2009 (Fig. 3), with similar values measured in samples from deeper depths to 30 m. Mean annual concentrations decline from south to north, with concentrations at the Tea Island station measuring 1.49 mg L^{-1} higher on average than corresponding results at the Rogers Rock station, and more recently over 2 mg L^{-1} higher (Table 5). The differences between the Tea Island and Rogers Rock stations within each year are significant ($P < 0.001$, $df > 12$; t-test) after 1985. About half this difference in chloride concentration occurs between the Tea Island and Dome Island stations.

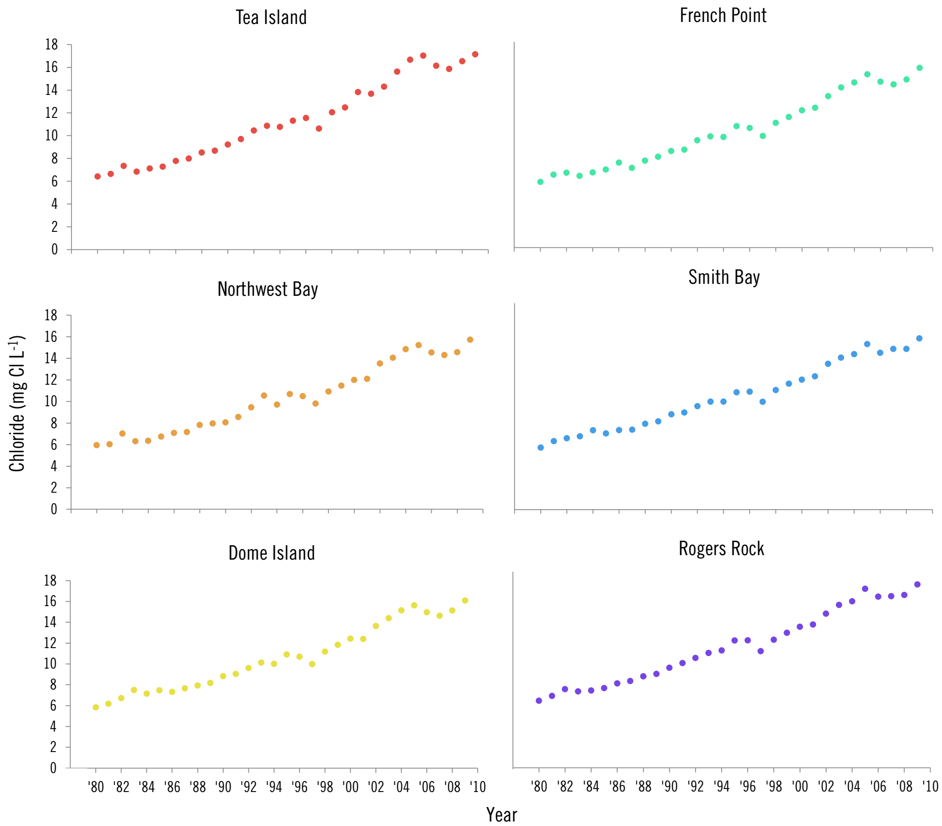

FIGURE 3. *Mean annual chloride concentration (as mg Cl L^{-1}) measured in surface waters (0 – 10 m depth) at the six Lake George monitoring stations, 1980 – 2009. 95% confidence intervals are smaller than the colored circles indicating data points.*

Mean annual chloride concentrations are significantly correlated among stations (Fig.3), with coefficients of determination (r^2) near 0.99. Time-series analysis shows similar trends at Tea Island and Rogers Rock, with slowly decaying autocorrelations significant to lags of six years and a significant partial autocorrelation at one year, suggesting a random-walk model with constant inter-annual differences of 0.33 mg L^{-1} at Rogers Rock and 0.38 mg L^{-1} at Tea Island. The random-walk model implies that annual deviations from the long-term trend reflect random processes rather than distinct or episodic shifts.

Station	Tea Island	Northwest Bay	Dome Island	French Point	Smith Bay	Rogers Rock
Min	6.42	5.96	5.83	5.76	5.46	5.40
1st Quartile	8.18	7.33	7.99	7.48	7.17	7.40
Median	10.8	10.2	10.1	9.71	9.54	9.41
Mean	11.4	10.3	10.7	10.3	9.90	9.89
3rd Quartile	14.2	13.1	13.3	13.0	12.4	12.4
Max	17.2	15.7	16.1	15.6	15.0	14.9

TABLE 5. *Table of mean, median and quartile chloride concentrations (as mg Cl L^{-1}) in surface waters (0 – 10 m depth) at the six Lake George monitoring stations, 1980 - 2009. Left to right columns correspond with generally down-current (south to north) station location in the lake, except for Tea Island and Northwest Bay, which are both up-current from Dome Island but not from each other (see Fig. 1b).*

Sodium concentrations follow nearly identical trends as chloride, when compared on a molar basis (Fig. 4). Sodium was not analyzed as consistently as chloride, precluding a comparative time series analysis. Sodium concentrations on average were around 15% less than chloride concentrations in molar terms, a deficit of about 0.066 *mM* in the late 2000s. This deficit is likely the result of cationic exchange as salt-laden water moves through watershed soils (Mason et al. 1999).

After road salt dissolves, chloride and sodium ions react differently depending on the concentrations of each and the soils with which they interact (Mason et al. 1999, Langen et al. 2006). Dissolved chloride travels through the soil at varying rates depending mainly on soil type and precipitation patterns. Chloride ions do not sorb onto mineral surfaces, enter oxidation, reduction or biochemical reactions or form quantitatively important complexes and thus are highly mobile with a migration rate identical to water (Jones et al. 1992). In contrast, the positive charge on sodium ions results in some adsorption onto the negatively-charged active sites on organic matter, minerals and secondary soil phases such as $Al(OH)_3$ (Langen et al. 2006, Kelting and Laxson 2010). The extent of adsorption depends on the cation exchange capacity, base saturation and selectivity (Environment Canada 2001). When present in excess, sodium displaces other soil cations because of simple mass action rather than a greater charge attraction, because sodium has the lowest affinity of major cations in soil (Langen et al. 2006, Kelting and Laxson 2010). The close molar correspondence of sodium and chloride ions in the lake samples leaves little doubt that the main source is sodium chloride from road de-icing in winter.

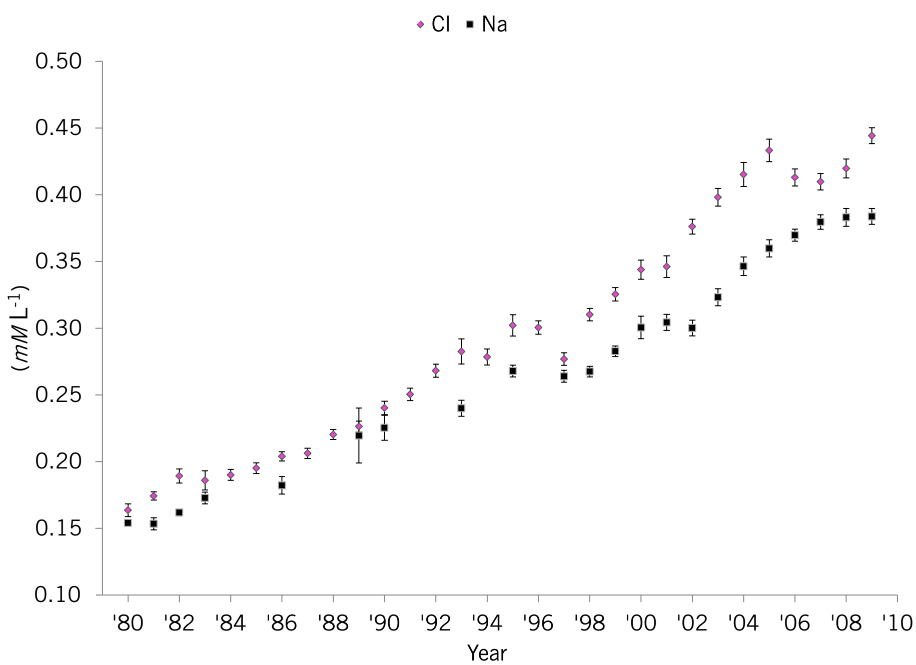

FIGURE 4. *Mean annual sodium and chloride concentrations (as millimoles per liter) measured in surface waters (0 – 10 m depth) of Lake George, 1980 – 2009.*

Routine monitoring for potassium did not begin until early 2004, although there were sporadic measurements dating back to 1980. The overall average potassium concentration is 0.516 ± 0.002 mg L^{-1}, ranging from 0.38 – 0.68 mg L^{-1}. Mean concentrations of potassium at the Tea Island and Rogers Rock stations were 0.533 and 0.506 mg L^{-1}, respectively, a small but significant difference ($P < 0.001$, $df = 359$; *t*-test), confirming a declining south-to-north concentration gradient similar to that of chloride and sodium. This may reflect contributions of potassium chloride from road de-icing salt or desorbed potassium.

Salt concentrations in Lake George had already increased substantially from pristine concentrations by 1980. Chloride concentrations in Adirondack lakes within roadless watersheds are typically below 1 mg L^{-1} (Baker et al. 1990b), compared with 5.8 mg L^{-1} in Lake George in 1980. The higher concentration in Lake George in 1980 is almost certainly the result of road salt application in the decades prior to 1980. These increases in other Adirondack lakes within developed watersheds, including Lake Champlain, where chloride concentrations are nearly as high as in Lake George (Vermont and New York State Departments of Environmental Conservation 2012), confirm widespread watershed salinization from road salt application throughout the region (Langen et al. 2006).

Most of the road de-icing salt probably enters the lake during winter periods of snow melt culminating in the spring thaw (Langen et al. 2006), and if added prior to spring turnover would tend to flow to the bottom of the receiving basin. Saline water from snowmelt that enters the lake during spring will be cooler than the lake's warming surface waters, and so it will initially descend in the lake's water column. As the saline meltwater descends to the isopycnal depth of mechanical equilibrium, it may still not be in thermal equilibrium with the surrounding water at that depth. Heat exchange with the surrounding water, which can be a slow process, will cause it to approach thermal equilibrium, but any remaining salt burden will result in the isothermal saline water becoming more dense than the surrounding water, causing the saline water to sink further. These concurrent processes will result in the saline water reaching the bottom of the receiving basin unless dilution through mixing sufficiently reduces these density differences.

Monitoring data for chloride corroborate transport of saline meltwater to the bottom of Caldwell Basin prior to the spring turnover. At the Tea Island station, the median winter chloride concentrations near the bottom exceeded the surface concentrations by 13%, and ranged up to 76%. During spring after the turnover, the median surface chloride concentration at Tea Island exceeded subsequent concentrations during the year by 7%. Neither of these increases was observed at comparable depths and times at the other stations, in part because sampling was not conducted during winter except at Smith Bay, where sampling did not extend to near the maximum depth of the basin. Moreover, in watersheds such as Northwest Bay with few roads and little other development, spring meltwater is likely to be less saline than the receiving waters of the lake, and the lower density of the meltwater would then tend to spread across the surface layer of the lake. In either case, density differences ultimately arising from addition of road de-icing salt to the lake will alter insertion of stream or groundwater inflows in comparison with its pristine state, which will also alter the initial depths at which nutrients carried by meltwater or groundwater are introduced into the lake. Expanded monitoring during winter and early spring to the deepest depths of the other sub-basins of the lake would provide helpful insights on the extent to which these processes occur beyond the Caldwell Basin.

The results for sodium, potassium and especially chloride are consistent with disproportionately greater input of salt at the more developed southern end of the lake, especially during winter, and progressive dilution by less saline streams and precipitation as water is transported horizontally after ice melt and spring turnover. The very high inter-annual correlations of chloride among stations (Fig. 3) suggests horizontal transport throughout the surface waters of the lake on a time scale of substantially less than a year. A decreasing trend of salt concentrations (and of several other water quality parameters) from south to north (*e.g.* Table 5) is a consequence of the greater watershed area and longer water residence time of the south basin, and disproportionately greater influx to the southern basin (Siegfried 1982, Dillon 1983, Siegfried and Quinn 1986). But while this scenario appears consistent with the physical forcing factors discussed in Section IV above, a detailed circulation model that accounts for distributions of salts, temperature and other measured parameters would provide more compelling evidence.

Recent concentrations of chloride in Lake George are near thresholds that may alter the composition of the phytoplankton community. A statistical analysis of diatom assemblages in sediment cores from Adirondack lakes suggests that chloride may begin to alter diatom species composition above concentrations of about 10 – 15 mg L^{-1}, or 0.3 – 0.5 *mM* (Fig. 6 in Dixit and Smol 1994; see also Environment Canada and Health Canada 1999). Chloride concentrations of this magnitude have been present in Lake George since about 1995 (Figs. 3 and 4).

Mass-balance comparisons suggest that application rates of de-icing salt are currently equivalent to loading rates on an annual basis. A residence time of 7 years means that 1/7 of the 2.1 km^3 lake volume drains out the LaChute River annually, carrying a chloride concentration of 15.9 mg L^{-1} chloride in 2009, equivalent to 26.2 mg L^{-1} sodium chloride. This is equivalent to an export of 7,900 tonnes of sodium chloride from the lake. The ~0.35 mg L^{-1} increase of chloride concentration in the lake during 2009, equivalent to 0.58 mg L^{-1} sodium chloride, implies a retained increase of another 1,200 tonnes of salt within the lake, for a combined total of 9,100 tonnes of salt, agreeing with the 8,000 tonnes estimated to have been dispersed annually within the watershed.

If the salt loading to the lake remains constant at its current rate, the long-term steady-state concentration of salt in Lake George is close to the current concentration. Addition of 9×10^6 kg of salt annually to an inflow of 1/7 of 2.1 km^3 water implies a steady-state sodium chloride concentration of 30 mg L^{-1}, only 15% above the 26.2 mg L^{-1} present during 2009. This estimate should be confirmed by more rigorous monitoring since annual loadings may change substantially based on the amount of precipitation and the severity of winter storms, which require higher road salt application rates.

CALCIUM AND MAGNESIUM

The calcium and magnesium concentrations in Lake George reflect the soft water of the lake, resulting from the relative scarcity of calcareous minerals in the lake's watershed. The bedrock minerals are dominated by silicates and aluminum silicates, some containing potassium, sodium and calcium, with more localized carbonates (McClelland and Isachsen 1986). Calcium concentrations in the lake range from about 10 – 12.5 mg L^{-1} (Fig. 5). Consistent monitoring for magnesium did not begin until early in 1997, after which mean annual concentrations ranged from about 2.1 – 2.6 mg L^{-1} (Fig. 5). The combined concentrations range from 12 – 15 mg L^{-1}, indicating relatively soft water.

In contrast with sodium and chloride ions, there is no evidence of a declining south-to-north concentration gradient for calcium ions. The mean calcium concentration averaged over 1984 – 2009 at the Tea Island station in the south and at the Rogers Rock station in the north were 11.27 ± 0.084 mg L^{-1} ($n = 555$), and 11.32 ± 0.102 mg L^{-1} ($n = 340$), respectively, an insignificant difference ($P = 0.33$, $df = 528$; paired t-test).

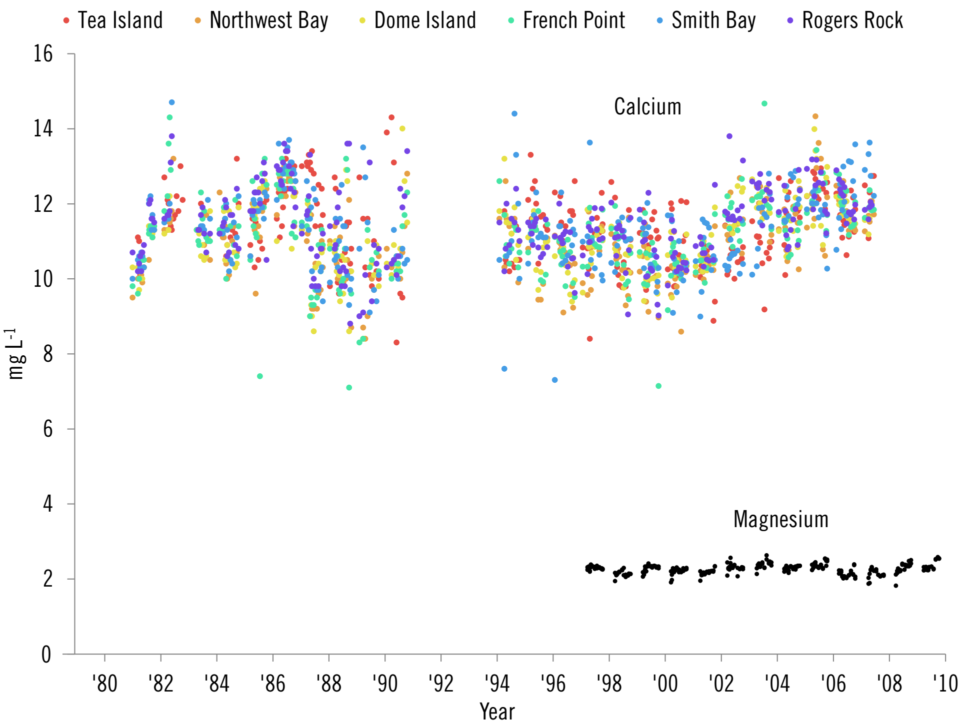

FIGURE 5. *Calcium and magnesium concentrations (as mg Ca or Mg L^{-1}) measured in surface waters (0 – 10 m depth) of Lake George, 1984 – 2009. Individual stations are identified by colored dots for calcium, but all stations are represented by black dots for magnesium because the lower variability of magnesium among stations precludes their resolution at the scale of the figure.*

There is little evidence for sustained temporal change of calcium from 1984 through 2009. There appears to have been a downward shift of perhaps 1.5 mg L^{-1} after 1987, followed by a shift back upward after 2003 (Fig. 5). Causes for these shifts are unknown.

Results for magnesium concentrations suggest a slight increasing trend since 1997 (Fig. 5), but several more years of monitoring will be needed for confirmation by time-series analysis. In any case the magnitude of the apparent increase is modest, on the order of 10%. The ratio of calcium to magnesium in Lake George is about 4.8, similar to mean river water worldwide (3.7; in Clarke 1924). As with calcium, there is little indication of a declining south-to-north concentration gradient for magnesium. Mean concentrations of magnesium at the Tea Island and Rogers Rock stations were both 2.35 ± 0.010 mg L^{-1} ($n > 281$), again consistent with numerous natural sources dispersed throughout the northern and southern basins of the lake. Finally, variation of magnesium concentrations among stations, both absolute and as a proportion of the mean, was substantially less than for calcium (Fig. 5).

ALKALINITY, SULFATE, AND PH

Alkalinity refers to the acid-neutralizing capacity of water. Although traditionally expressed in terms of the acid-neutralizing capacity of calcium carbonate ($CaCO_3$), in lakes this capacity is furnished by conjugate bases of weak acids generally, particularly bicarbonate (HCO_3^-), carbonate (CO_3^{-2}), sulfate (SO_4^{-2}), and organic acids generated in terrestrial soils. For Lake George where the pH is almost always less than 8, any carbonate dissolved from calcite or dolomite is converted to bicarbonate ion.

Alkalinity measurements in the 0 – 10 m integrated surface water samples from the six main monitoring stations began in 1990, and were consistently measured after 1995 except in Northwest Bay, where measurements commenced in 2002 (Fig. 6). A modest but clear upward trend is evident from about 27 mg $CaCO_3$ L^{-1} in 1990 to about 30 mg $CaCO_3$ L^{-1} (0.27 – 0.30 *mM*) in 2009. This concentration range for alkalinity is less than the calcium and magnesium concentrations presented above, which sum to about 0.39 *mM*.

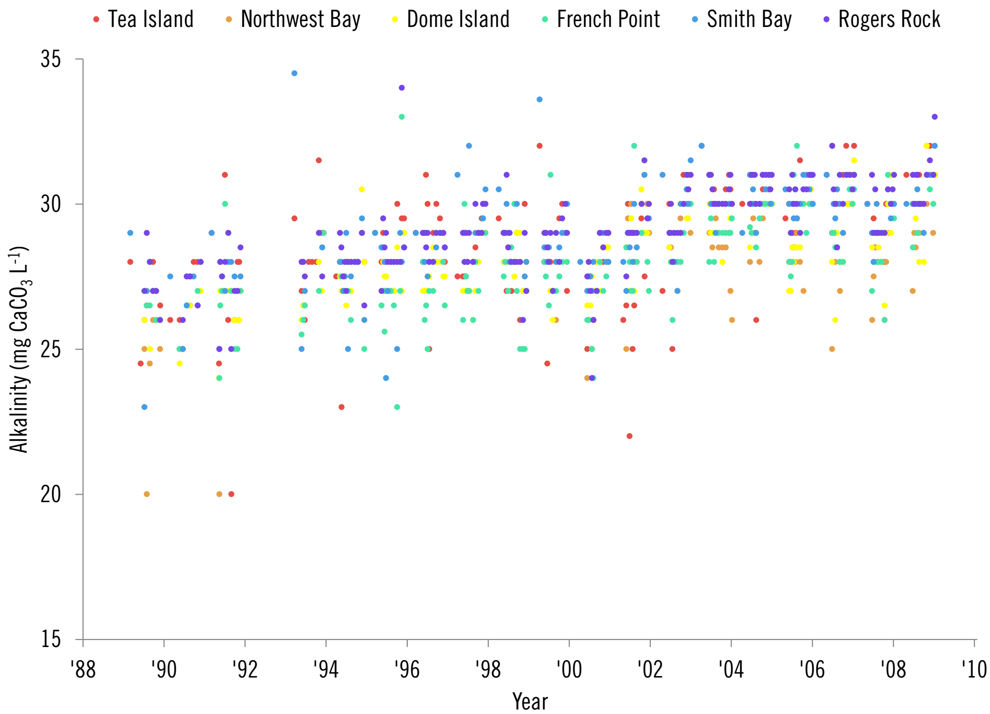

FIGURE 6. *Mean annual alkalinity (as mg $CaCO_3$ L^{-1}) in surface waters (0 – 10 m depth) of Lake George, 1990 – 2009.*

Alkalinity at the southern stations of the lake is on average slightly lower than at the northern stations. Paired comparison of contemporaneous alkalinity samples from 1995 through 2009 shows a slight but significantly greater alkalinity at Rogers Rock (29.3 ± 0.021 mg $CaCO_3$ L^{-1}, n = 168) than at Tea Island (28.8 ± 0.026 mg $CaCO_3$ L^{-1}, n = 168; P < 0.001; df = 334, paired t-test). Comparing results for the four southern-most stations from 2002 onwards to include the Northwest Bay station on a comparable basis, the overall average alkalinity of the southern and northern stations was 29.2 ± 0.16 mg $CaCO_3$ L^{-1} (n = 321) and 30.1 ± 0.14 mg $CaCO_3$ L^{-1} (n = 182), respectively, again a significant difference (P < 0.001; df = 538, t-test). Within the southern basin mean alkalinity was lowest at the Northwest Bay station (28.8 ± 0.28 mg $CaCO_3$ L^{-1}, n = 87). Overall, these results confirm that Lake George retains substantial acid-neutralizing (i.e. buffering) capacity.

Emission reductions of sulfate following the 1990 Clean Air Act re-authorization are clearly reflected in steadily declining sulfate concentrations in Lake George (Fig. 7). Lake-wide mean annual sulfate concentrations declined from 2.94 ± 0.095 mg S L^{-1} (n = 46) in 1990 to 1.94 ± 0.026 mg S L^{-1} (n = 68) in 2009, a decline in molar terms of 0.031 *mM*. Paired comparison from 1995 through 2009 shows a significant (P = 0.016; df = 364, paired t-test) but negligibly greater mean sulfate at Rogers Rock (2.50 ± 0.053 mg S L^{-1}, n = 183) than at Tea Island (2.48 ± 0.050 mg S L^{-1}, n = 183). These results are consistent with an overwhelmingly dominant atmospheric source for sulfate deposited homogenously within the watershed. Atmospheric deposition of sulfate has been declining steadily since 1992, with a parallel decline in the surface waters of Lake George (Fig. 7). The higher concentrations in the lake compared with precipitation reflects a lag in the release of sulfate from soils as atmospheric deposition declines (Rustad et al. 1994).

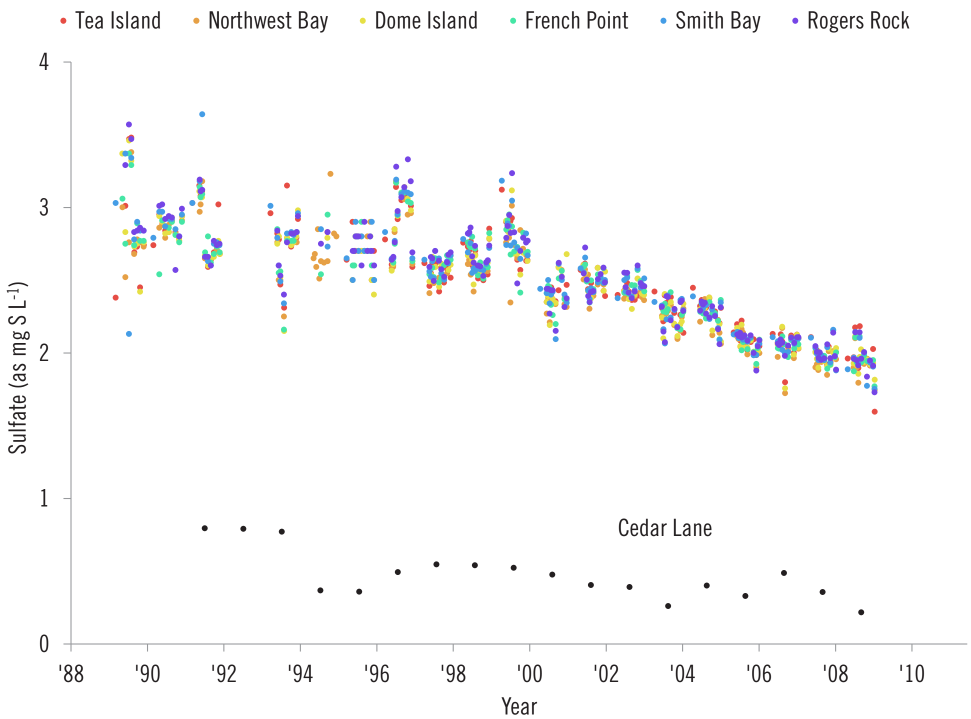

FIGURE 7. *Mean annual sulfate concentration (as mg S L⁻¹) in precipitation measured at the Cedar Lane atmospheric deposition station from 1991 to 2009, and in the surface waters (0 – 10 m depth) of Lake George, 1989 – 2009.*

The declining deposition of atmospheric sulfate (and nitrogen compounds; see Sec. VII), and increasing alkalinity are associated with declining acidity in the surface waters of Lake George (Fig. 8). The seasonal decrease of acidity during summer evident in Fig. 8 reflects the photosynthetic uptake of carbon dioxide by plants growing within the lake. As plants withdraw carbon dioxide from the water column, the lake's carbonate buffer system shifts to replace some of the carbon dioxide removed, which consumes hydrogen ions.

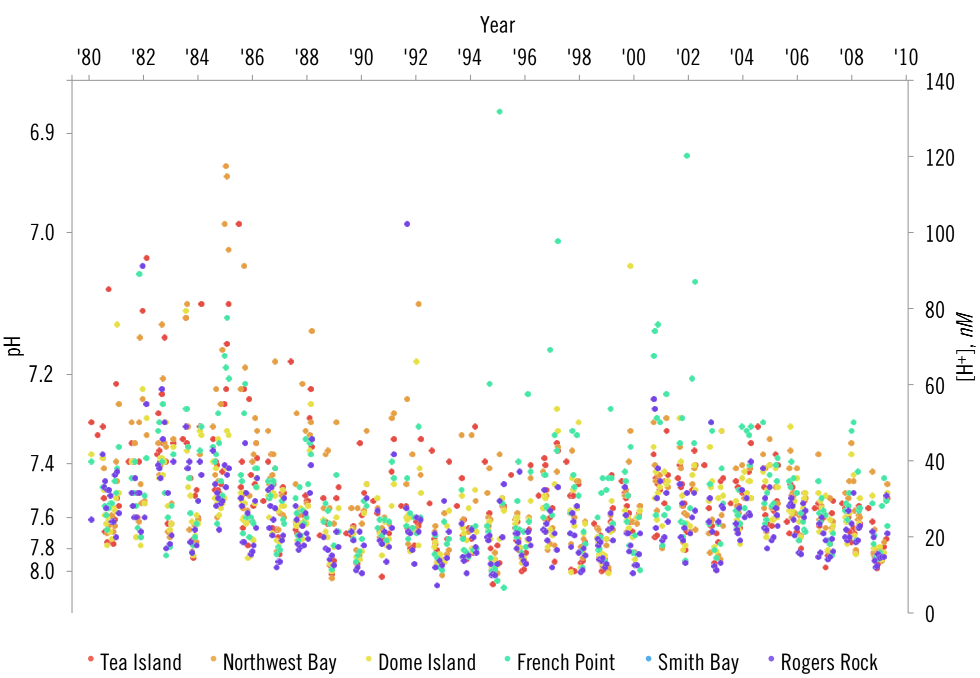

FIGURE 8. *Hydrogen ion activities (as pH, left-hand axis, and as nanomoles per liter, right-hand axis) in surface waters (0 – 10 m depth) of Lake George, 1980 – 2009.*

CONDUCTIVITY AND IONIC BALANCE

Ionic balance, or the agreement of the normal equivalents of the anions and cations measured in lake waters, provides a rigorous measure of the accuracy and completeness of the analytical results. The dominant anions and cations were consistently measured at the six main monitoring stations beginning in 2004. Since then, the sum of the positive charges of the cations agrees with the sum of the negative charges of the anions to within about 2% on average, indicating excellent agreement. The 2% discrepancy results from a small dominance by cations, consistent with an unmeasured contribution of deprotonated organic acids and possibly fluoride to the anions. This average 2% discrepancy was evident at all six monitoring stations, ranging from 1.7% at Tea Island to 2.6% at Northwest Bay. The overall standard deviation is 0.046 ($n = 344$), indicating that 95% of results lie within 93% - 111%.

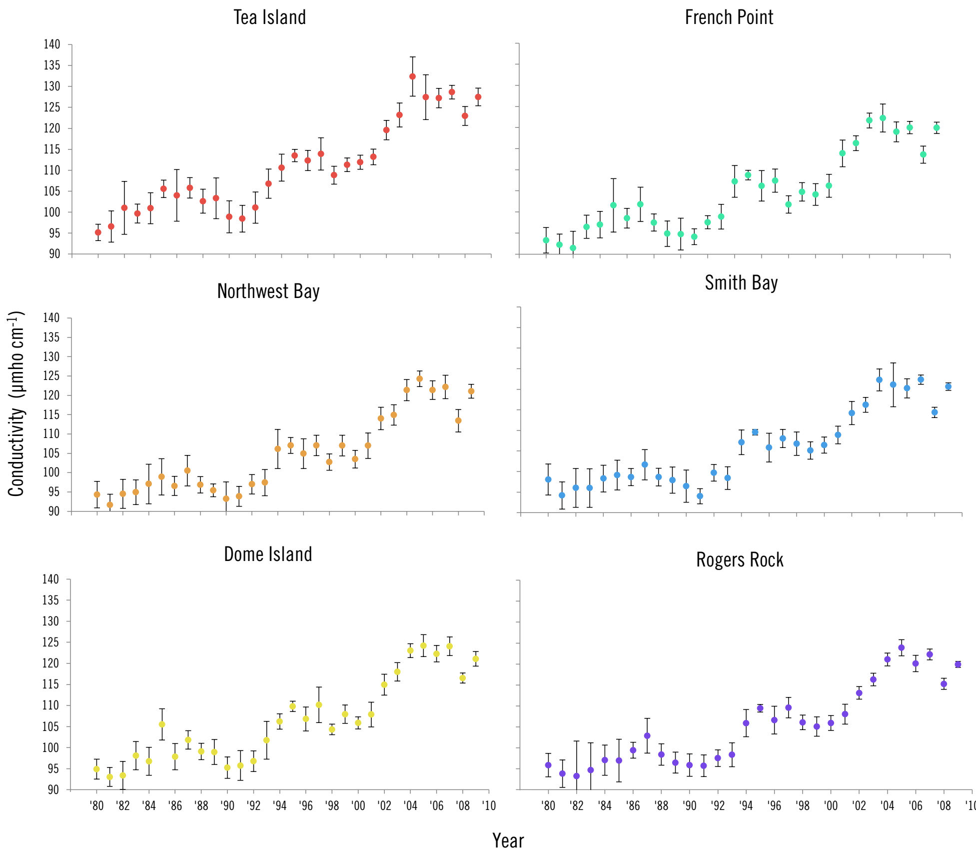

FIGURE 9. *Mean annual specific conductivities (as µmho cm^{-1}) of surface waters (0 – 10 m depth) at the six Lake George monitoring stations, 1980 – 2009.*

Temporal and spatial trends of specific conductivity in the 0 – 10 m water samples are broadly similar to those of sodium and chloride (compare Figs. 3, 4 and 9). Trends among annual mean specific conductivities at all six stations are highly correlated ($r^2 > 0.99$), with conductivities at the Tea Island station higher on average than at the other stations, consistent with comparable trends in the chloride concentrations (compare Tables 5 and 6). However, conductivity increased less consistently with time compared with sodium and chloride, at least in part because of declines of sulfate during the 1980s and 1990s when sodium and chloride were increasing.

Station	Tea Island	Northwest Bay	Dome Island	French Point	Smith Bay	Rogers Rock
Min	95.2	91.7	93.0	91.4	94.0	93.3
1st Quartile	102	96.6	97.9	97.1	98.3	97.0
Median	110	103	106	103	106	106
Mean	111	105	106	105	106	106
3rd Quartile	118	112	114	112	113	112
Max	134	124	124	122	122	124

TABLE 6. *Table of mean, median and quartile specific conductances (as μmho cm^{-1}) in surface waters (0 – 10 m depth) at the six Lake George monitoring stations, 1980 - 2009. Left to right columns correspond with generally down-current station location in the lake, except for Tea Island and Northwest Bay, which are both up-current from Dome Island but not from each other (see Fig. 1b).*

SUMMARY

Persistent addition of road de-icing salt to the Lake George watershed has converted the lake from a calcium carbonate- to a sodium chloride-dominated water body. At the beginning of the monitoring period the molar concentration proportions of cations, Ca^{+2}:Na^{+}:Mg^{+2}:K^{+} was about 0.28:0.14:0.10:0.013, and of anions CO_3^{-2}:Cl^{-}:SO_4^{-2} was about 0.27:0.16:0.11. By 2009 these ratios changed respectively to 0.28:0.41:0.10:0.013 for the cations and to 0.31:0.45:0.063 for the anions.

Analysis of the spatial and temporal trends of chloride suggest that the surface waters of the northern and southern basins of the lake mix on a time scale of weeks to months, resulting in strong correlation among stations for chloride ion and for conductivity.

Concentrations of chloride in Lake George are near or above the threshold that may alter the community composition of diatoms, and hence may be changing the biological productivity of the lake.

Calcium and magnesium appear to enter the lake from chemical weathering of calcareous minerals, mostly calcium and magnesium carbonates, and lesser weathering of silicate minerals. Along with sulfate deposition from the atmosphere, these ions have negligible spatial variation in their concentrations in the surface waters of the lake. In contrast, concentrations of sodium, potassium and chloride are consistently higher in the more densely populated southern basin of the lake, most likely because of greater application of de-icing salt there.

VII. NUTRIENTS

THE BIOLOGICAL CHARACTERISTICS OF LAKES DEPEND FUNDAMENTALLY on the supply and cycling of inorganic nutrients. Phosphorus, nitrogen, and silica constitute "macro-nutrients" because of the relative quantities required by organisms in comparison with "micro-nutrients" such as iron, manganese, cobalt, etc. Populations of fish and other fauna depend on plant productivity that is limited by the nutrient in shortest supply (Sprengel 1828). In lakes, phosphorus is usually limiting, although silica may limit growth of diatoms that usually sustain annual phytoplankton blooms during spring, and of periphytic algae that proliferate concurrently on littoral sediments and rooted vegetation also during spring (Hutchinson 1957). The ultimate source of phosphorus in pristine systems is from chemical weathering of relatively scarce phosphate minerals, especially apatite, $Ca_5(PO_4)_3(OH)$, which is usually present as < 2% of many common rocks. Nitrogen compounds can be fixed from atmospheric N_2 by some aquatic algae, and by terrestrial and aquatic microbes. Nitrogen compounds may also be deposited from the atmosphere as nitrate produced by combustion and as ammonium ion (NH_4^+) from natural and anthropogenic sources. Ubiquitous silicate minerals in the watershed provide silica from chemical weathering. Nutrients exported from lakes drained by an outlet must be replaced to maintain plant productivity.

With its relatively small watershed, Lake George has rather limited natural sources of phosphates. Phosphate is released through the weathering of rocks which generally contain 0.1 to 1.0% phosphate, although the ~5:1 ratio of land to lake area in the catchment basin limits the availability of phosphate from these sources. In addition to chemical weathering of phosphate minerals within the watershed, atmospheric deposition of phosphate-bearing particulates either directly into the lake, or indirectly following deposition within the watershed is another important source. These sources may be augmented by deposition of urine and fecal matter produced by migratory birds, mammals and other fauna, which serve to disperse nutrients among watersheds. Imports from these sources initially fertilize growth of the surrounding forest, eventually releasing organic phosphates and other nutrients from decaying plant matter into streams feeding the lake. The extreme clarity of Lake George documented by early European witnesses (Appendix I) attests to a nutrient-poor water body, probably qualifying as hyperoligotrophic, and capable of supporting only modest populations of fish and aquatic wildlife. Such lakes are especially sensitive to increased nutrient loading that typically results from human settlement, as relatively modest nutrient increases may result in substantial biological responses.

Humans supplement the nutrient supply to lakes through discharges containing excretion of bodily waste products, by application of agricultural fertilizers and by discharges of products from other industrial, domestic and recreational activities. Household detergents containing relatively high concentrations of phosphorus compounds accounted for 30% - 50% of phosphorus in wastewater streams discharging into freshwaters of the United States in the 1960s (Ferguson 1968, Hammond 1971) prior to regulatory limitation by states.

Excess nutrient loading frequently leads to eutrophication, where plant productivity becomes so great that the subsequent plant matter decay causes excessive biological demand for oxygen. The increased oxygen demand can lower oxygen concentrations dramatically, which can lead to fish kills. Furthermore, low dissolved oxygen promotes nutrient regeneration from lake-bottom sediments, stimulating a positive feedback where more plant productivity leads to more extensive hypoxia of bottom waters causing more nutrient release, which stimulates more plant productivity. Once engaged, such feedbacks are difficult to arrest or reverse, justifying the considerable attention given to evidence of increasing nutrient loading to Lake George throughout the 1960s and 1970s (Ferris et al. 1974, Wood and Fuhs 1979).

Accelerating eutrophication rates noted throughout the United States and Canada during the 1960s and 1970s led to legislation limiting phosphorus discharges, as well as to voluntary industry efforts. Thousands of public lakes were affected, perhaps most notoriously Lake Erie (ReVelle and ReVelle 1988). In New York, regulations adopted in 1976 limited phosphorus in household detergents to 8.7% (NYSDEC 1985). As a result of these concerns and to evaluate the efficacy of such regulatory efforts, most lake monitoring programs have focused on phosphorus compounds.

Evaluating the consequences of nutrient loadings was a major goal of the Lake George Offshore Chemical Monitoring Program. By the time the program started, loading had already increased substantially, and anticipated increases associated with economic and population growth prompted concerns regarding eutrophication (*e.g.* Wood and Fuhs 1979, Stross 1982). In this section we focus on evaluating trends in the concentrations of the nutrients monitored, spatially in the context of results for conserved analytes presented in Section VI above on major ions and conductivity, and temporally in the context of loading increases. These results will provide a broader context for interpreting the biological response of the lake through plant growth and oxygen availability presented in subsequent sections.

PHOSPHORUS

Epilimnetic concentrations of total phosphorus and total soluble phosphorus remained stable from 1980 – 2009 (Fig. 10). Mean total phosphorus concentrations averaged across all samples (~70) collected within a year ranged from 3.69 µg L^{-1} (in 2007) to 5.52 µg L^{-1} (in 1990), with a grand mean of 4.36 µg L^{-1} (n = 1,970) for the whole monitoring period. About half the total phosphorus was soluble phosphorus, which ranged from 1.70 µg L^{-1} to 2.72 µg L^{-1}, with a 30-year mean of 2.09 µg L^{-1}. Results for orthophosphate suggest a slightly increasing tendency based on frequency of detection. Orthophosphate concentrations were often near or below the detection limit of 1.0 µg L^{-1}. Values below the analytical detection limit were converted to one-half the detection limit for statistical analyses. The proportion of samples above the detection limit within a year has clearly increased from 1980 to 2009 (Fig. 11), with mean annual concentrations ranging between 0.88 µg L^{-1} and 1.84 µg L^{-1} since 1997. However, these orthophosphate increases are too small to be reflected in the results for total phosphorus.

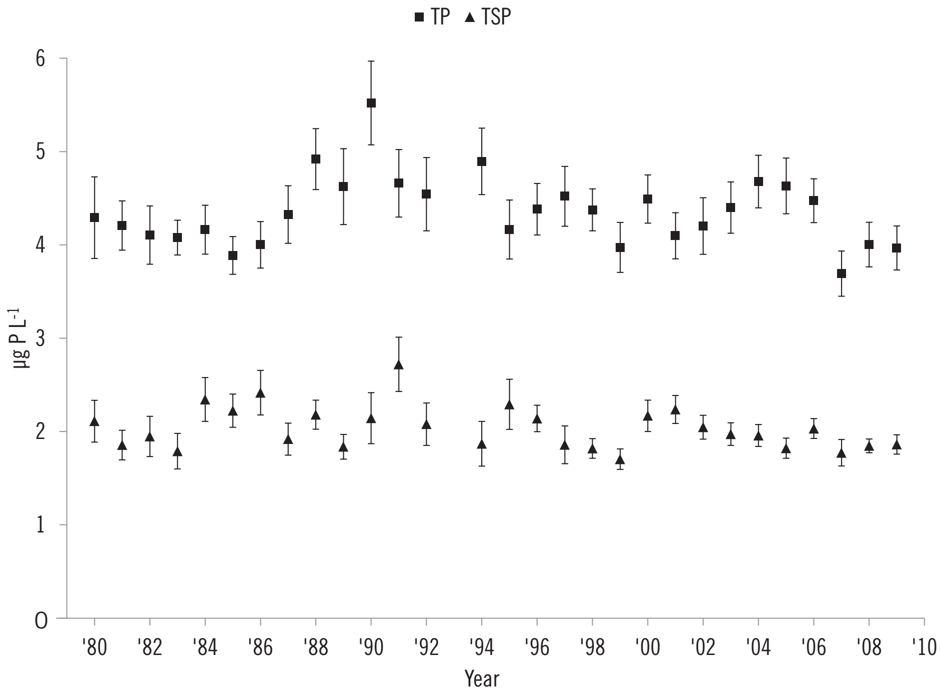

FIGURE 10. *Average annual concentrations of total phosphorus (TP) and total soluble phosphorus (TSP) as µg P L^{-1} in the surface waters (0 – 10 m depth) of Lake George, 1980 – 2009.*

Time-series analysis indicated no autocorrelation or partial autocorrelation at the Tea Island station, and was consistent with a random-walk model, implying purely random fluctuations about the long-term mean on a time scale of less than one year. In contrast, results for the Rogers Rock station indicate significant autocorrelation and partial autocorrelation with a one year lag, implying a (1,0,0) ARIMA model (*i.e.* one autoregressive parameter with no differencing and no moving average). These results suggest similar inputs of total phosphorus that mix relatively rapidly throughout the south basin, but have a delayed effect on the north basin.

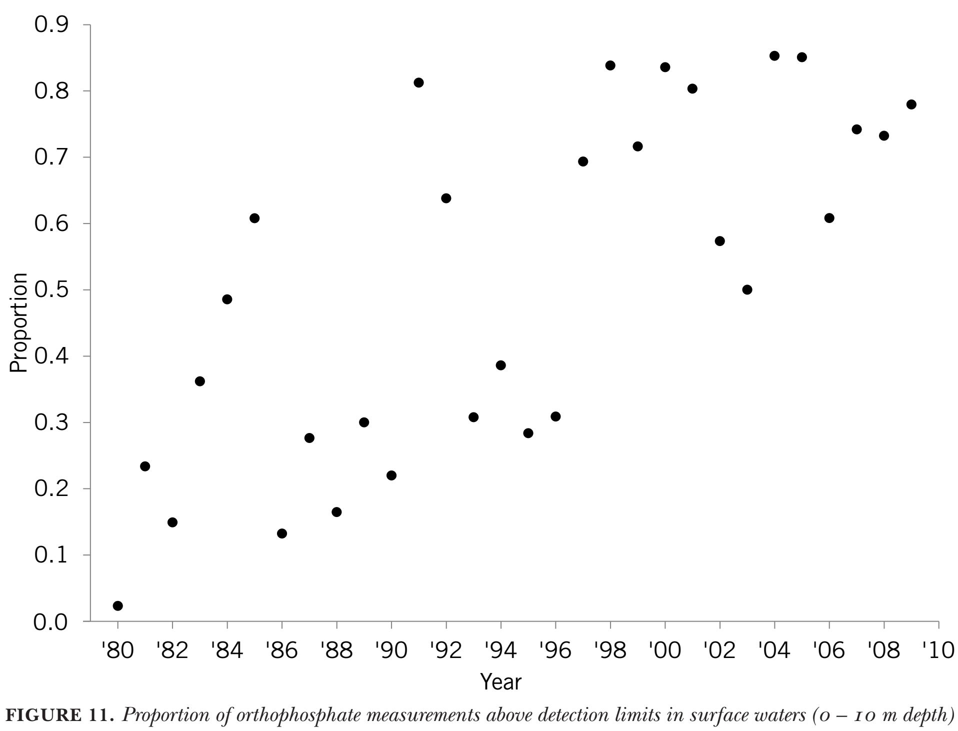

FIGURE 11. *Proportion of orthophosphate measurements above detection limits in surface waters (0 – 10 m depth) of Lake George, 1980 – 2009.*

Mean annual concentrations of epilimnetic total phosphorus were considerably less correlated among stations in comparison with chloride. Coefficients of determination (r^2) among the three stations in the southern basin ranged from 0.43 to 0.62, but all other coefficients ranged from 0.07 to 0.33. The overall mean concentration was highest at the Tea Island station (5.14 μg L^{-1}, Table 7), declining steadily northward to the lowest concentration at the Rogers Rock station (3.64 μg L^{-1}), with about a third of the decline occurring between the Tea Island and Dome Island stations. Note that the total phosphorus decline of 41% from south to north is steeper than the 15% decline for chloride (Table 5). A nearly identical south-to-north spatial gradient for total phosphorus based on independent sampling was reported for 1982 (Siegfried and Quinn 1986). Monitoring results since 1980 indicate phosphorus inputs that are disproportionally greater than chloride at the more developed southern end of the lake, consistent with trends noted by Ferris and Clesceri (1974).

Station	Tea Island	Northwest Bay	Dome Island	French Point	Smith Bay	Rogers Rock
Min	4.20	3.79	3.61	3.67	3.00	3.17
1st Quartile	4.63	4.15	4.15	4.11	3.49	3.44
Median	5.08	4.59	4.63	4.27	3.71	3.63
Mean	5.14	4.59	4.60	4.42	3.73	3.65
3rd Quartile	5.55	5.01	4.87	4.74	3.88	3.81
Max	6.89	5.50	5.88	5.50	5.44	4.21

TABLE 7. *Table of mean, median and quartile total phosphorus concentrations (as μg P L^{-1}) of 1,970 measurements in surface waters (0 – 10 m depth) at the six Lake George monitoring stations, 1980 - 2009. Left to right columns correspond with generally down-current station location in the lake, except for Tea Island and Northwest Bay, which are both up-current from Dome Island but not from each other (see Fig. 1b).*

Biologically available phosphorus imported to the lake is incorporated into the food web on time scales of days to weeks. Experiments conducted by Eiling (1992) demonstrated that even in the dark, phosphorus released by hypoxic sediments of Lake George stimulated microbial production within about a week. Even faster incorporation would be expected by phytoplankton for biologically available phosphorus imported into the lake. Such rapid incorporation is confirmed by monitoring observations of seasonal depletion of phosphorus showing declines from wintertime maxima to seasonal minima during summer, followed by increases during fall (Fig. 12). While seasonal declines in orthophosphate and total soluble phosphorus often account for most of the observed decline in total phosphorus, this is not always the case, suggesting that some of the particulate phosphorus may be made biologically available through decomposition. Conversely, biological utilization of phosphorus and subsequent loss from the water column from sedimentation only results in depletion of about half the total phosphorus available.

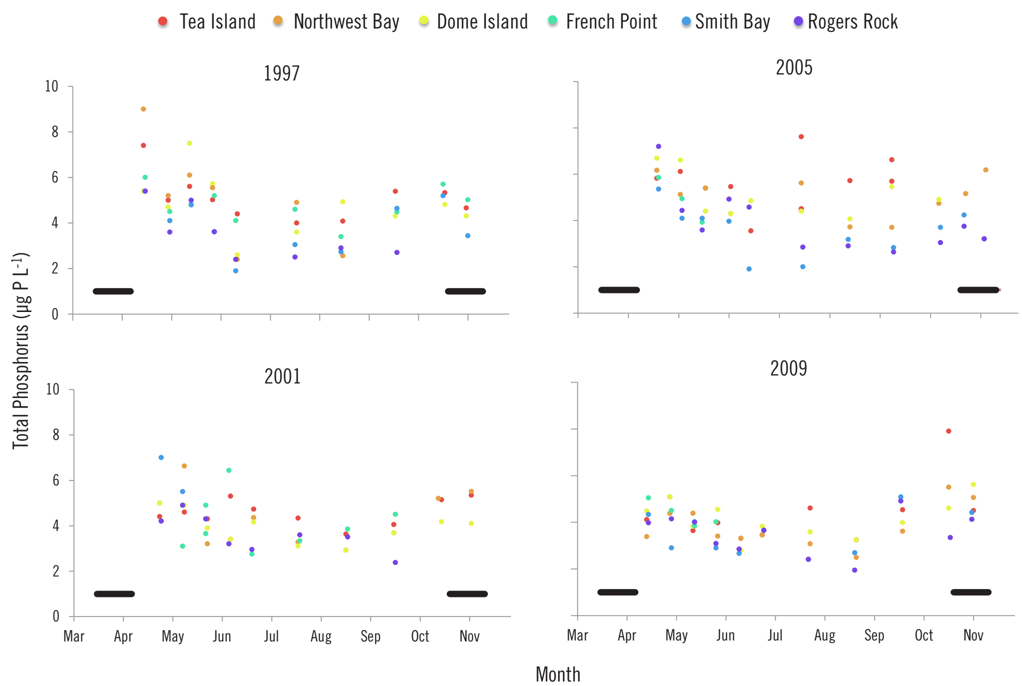

FIGURE 12. *Seasonal variation of total phosphorus (as µg P L^{-1}) in surface waters (0 – 10 m depth) of Lake George. Results from the six monitoring stations are presented for each sampling event. Spring and fall turnover periods are indicated by horizontal black bars.*

Once incorporated into the base of the food web, phosphorus will move into consumer organisms through grazing and predation, and subsequently into sediments through deposition of fecal material and carcasses, or removal from the lake through fishing, predation by mobile terrestrial predators or water discharge out the LaChute River. Rapid incorporation into the local biotic community would account for the south to north decrease in both phosphorus and the lower spatial correlation among the monitoring stations in comparison with the more conserved chloride and other major ions. When such biological cycling is rapid, monitoring phosphorus concentrations in the lake water may not provide a very sensitive reflection of increased loading, because undetectable increases in the total phosphorus measured may be associated with substantially increased flux through the total phosphorus pool to plants and then to higher-level consumers.

Although not well constrained, phosphorus budgets indicate that loading has increased since European settlement. Of 13 budgets reviewed, annual loading estimates ranged from 3,980 kg – 20,100 kg total phosphorus without a clear consensus regarding relative source contributions (Stearns & Wheler 2001). The observed spatial gradient of the monitoring results imply that area-wide non-point sources such as atmospheric deposition or forest runoff are likely less important than sources associated with human settlement. The most recent loading estimate (Stearns & Wheler 2001) corroborates this view, attributing about 36% of the estimated 10,200 kg total phosphorus loading to developed areas. An earlier study attributed 53% of an estimated total phosphorus loading of 13,100 kg to anthropogenic sources (Aulenbach 1979), despite considerable differences in source allocations. Probably the most careful budget for phosphorus was done by Sutherland et al. (1983), which indicated an anthropogenic increment of about 25% of total phosphorus loading. If phosphorus loadings were biologically inert, these annual inputs would result in total phosphorus concentrations of up to 40 µg P L^{-1}, about ten times the concentrations observed, confirming that phosphorus loading is rapidly incorporated into the food web or settles to the benthos as particulates.

Analysis of diatoms in a sediment core collected from Caldwell Basin in 1970 confirms a near doubling of water column productivity since European settlement (DelPrete and Park 1981). While current phosphorus loading to Lake George is clearly greater than loadings prior to European settlement, loadings may have peaked just prior to enforcement of regulations curtailing phosphorus in household detergents in the late 1970s. Although the accuracy and precision were not reported, studies conducted in the early 1970s reported average total phosphorus concentrations near 7 μg P L^{-1} throughout the lake (Aulenbach and Clesceri 1972, Hetling 1974), substantially above mean concentrations of 3.7 – 5.1 μg L^{-1} measured since 1980 (Table 7). This comparison strongly suggests the voluntary and regulatory efforts led to an appreciable reduction of phosphorus loading to Lake George from the late 1960s to the early 1980s.

Phosphorus released by hypoxic sediments during fall turnover can contribute to plant and microbial growth, at least in Caldwell Basin at the south end of the lake. Total phosphorus concentrations ranging to 25 μg L^{-1} occur in the 25 m and 30 m water samples from the Tea Island station during the weeks immediately preceding the fall turnover, when oxygen concentrations drop below 4 mg L^{-1} (see Sec. IX). The phosphorus released by these sediments mixes throughout the water column during the fall turnover, becoming available to stimulate phytoplankton and other plant growth. But without knowing how much phosphorus is utilized by bacteria during stratification and the total amount released into the epilimnion during turnover, the effect of the additional phosphorus loading cannot be quantified.

Elsewhere in the lake evidence for such internal phosphorus cycling is less clear. Monitoring at the other stations usually did not extend to the maximum depth of the respective basin except at Sabbath Day Point. The Sabbath Day Point station is located near the 30+ m local maximum sub-basin depth and within a few hundred meters of the 20 m minimum-depth barrier sill in the Narrows separating the South and North basins. This basin is therefore likely much better flushed by turbulence induced by wind- and density-driven currents and internal waves than is Caldwell Basin (see Section IV above), but which otherwise has bathymetry similar to that of Caldwell Basin.

Oxygen measurements from the Offshore Chemical Monitoring Program did extend to 43 m depth at the Dome Island station in 1993, where the oxygen concentration fell to 6.2 mg L^{-1} (or 52.6% saturation at 7 °C) in late October. Oxygen concentrations at the maximum basin depth of ~58 m, were not measured but were likely substantially lower. Similarly, the oxygen concentration declined to less than 5.5 mg L^{-1} (or about 50% saturation) at 35 m depth east of Dome Island by late October 1981 (Siegfried and Quinn 1986), again implying substantially lower concentrations at the local maximum sub-basin depth. These observations suggest that hypoxic conditions conducive to internal phosphorus cycling may occur in other basins of the lake (Sabbath Day Point excepted), although the volumes of hypolimnetic water affected are likely relatively small.

While the causes are not entirely clear, total phosphorus concentrations tend to increase in surface waters after August and through the fall turnover (Fig. 12). Phosphorus liberated from decaying littoral vegetation is one likely source (Hutchinson 1957), as well as phosphorus entering the hypolimnion from groundwater. Another possible source is phosphorus liberated from hypoxic sediments in Caldwell Basin and perhaps elsewhere. Hypolimnetic waters entrained into the epilimnion as a result of a weakening thermocline provide a mechanism to transport phosphorus vertically. Because understanding the full extent of internal phosphorus cycling is crucial for understanding the biological functioning of the lake, future monitoring efforts should place a high priority on this issue.

Coupled nutrient-phytoplankton-zooplankton models that account for incorporation of nutrients into the food web predict higher total phosphorus concentrations for Lake George than are observed. Assuming an annual loading of ~10,000 kg total phosphorus, four different models deemed appropriate predicted aqueous concentrations that were 32% – 104% greater than observed (Stearns & Wheler 2001). The discrepancy was attributed in part to nutrient uptake by rooted vegetation and epiphytic algae near shore, although given the uncertainty of the loading estimates, lower loadings than estimated may also be a factor. However, another possibility is nutrient uptake by the *Nitella* spp. meadows in the lake. *Nitella* are attached macroalga that absorb inorganic nutrients directly from the water column. These plants form extensive meadows at depths ranging from 7 – 16 m (Needham et al. 1922, Stross et al. 1988) and cover an estimated 20% of the south basin and 14% of the north basin (Stross 1981), accounting for about two-thirds of all the chlorophyll in the lake (Stross et al. 1988). The biomass of *Nitella* in the south basin was estimated at ~2 to 3 times that of the north basin (Stross et al. 1988), consistent with its suspected role as a major consumer of nutrient loading into the south basin.

Another possibility for the apparently lower phosphorus levels may be related to the sampling methodology whereby sampling is concentrated in the photic zone during the warmer months, when phosphorus uptake is greatest, thus possibly biasing the findings on a yearly basis. Furthermore, the phosphorus concentrations in a thin band at the thermocline may be higher than concentrations elsewhere due to the insertion of various inflows at the thermocline. Since the thermocline is not routinely targeted in the sampling, these higher concentrations may not have been measured.

NITROGEN

Monitoring for total nitrogen, which includes organic and inorganic forms, did not begin until 1997, but concentrations have declined steadily in surface waters since then (Fig. 13). Mean annual epilimnetic total nitrogen concentrations declined 23% (0.155 mg N L^{-1} in 1997 to 0.126 mg N L^{-1} in 2009), a significant trend ($P <$ 0.001, df = 1,754, linear regression). Total nitrogen concentrations in surface waters near the Tea Island station were about 17% greater than those near the Rogers Rock station, a significant difference ($P < 0.001$, $df = 139$, paired t-test). This is similar to the gradient for chloride but substantially less than the gradient for phosphorus. Coefficients of determination (r^2) among annual means for the three stations in the south basin ranged from 0.33 to 0.49, but all other coefficients ranged from 0.13 to 0.34.

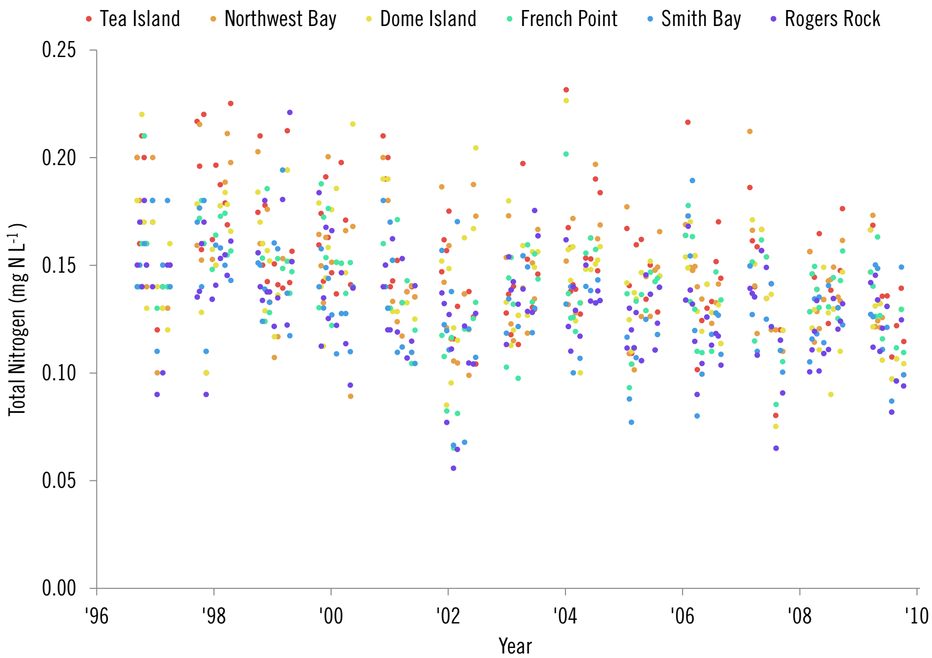

FIGURE 13. *Concentrations of total nitrogen (as mg N L⁻¹) in surface waters (0 – 10 m depth) of Lake George, 1997 – 2009.*

Total nitrogen increased significantly with depth, with the greatest increase at the Tea Island station where the mean concentration at 30 m depth, just above the lake bed was 0.193 mg N L^{-1}, 25% greater on average than the mean surface concentration of 0.154 mg N L^{-1} ($P < 0.001$, $df = 149$, paired *t*-test). Smaller but still significantly greater concentrations in the deeper samples occurred at other stations, indicating generally greater concentrations in the hypolimnion throughout the lake.

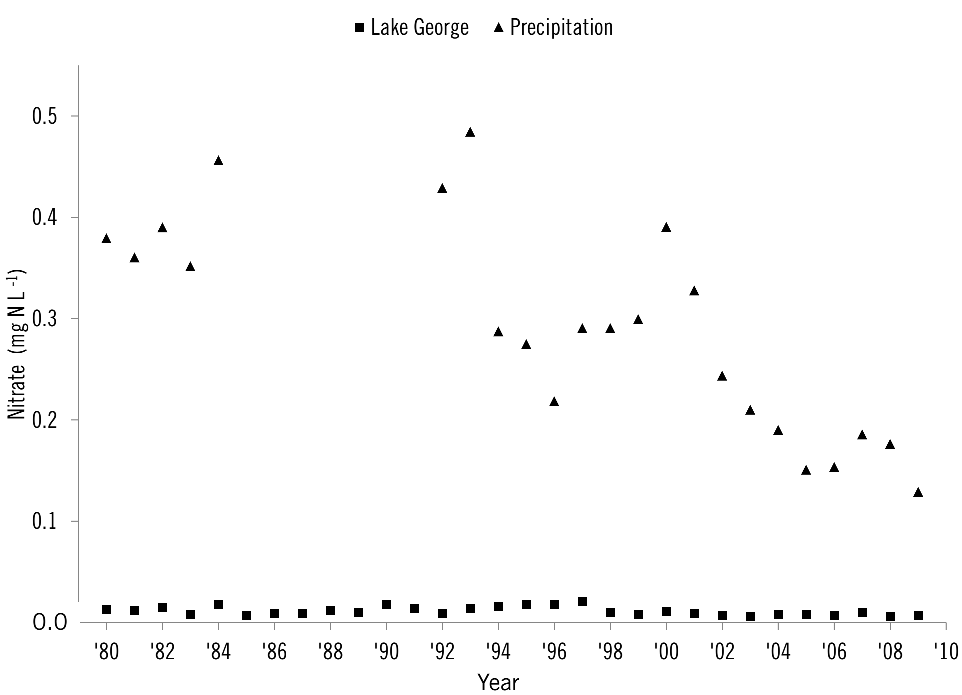

FIGURE 14. *Average annual concentrations of nitrate in precipitation (black triangles) and in surface waters (0 – 10 m depth) of Lake George (black squares), 1980 – 2009. Precipitation also contains nitrogen in the form of ammonium ion (NH_4^+) at concentrations of about half that of nitrate on a mg N L^{-1} basis (Eichler et al. 2008).*

Monitoring for nitrate and ammonia began in 1980, but concentrations were often near or below detection limits (10 μg N L^{-1}), especially after 1997, hampering statistical analysis for spatial and temporal trends. However, the lower concentrations evident after 1997 corroborate the declining trend of total nitrogen during this period.

Comparison of monitoring results for total nitrogen with total phosphorus confirms that phosphorus is the primary nutrient limiting plant growth. Converted to a molar basis, total nitrogen is about 70-times more abundant than total phosphorus, exceeding the ratio of N:P in most freshwater algal species (Hecky and Kilham 1988). Nitrogen loading to the lake has been estimated as 197,000 kg yr^{-1}, or 34-times phosphorus loading on a molar basis (Aulenbach 1979), with atmospheric deposition accounting for 59% of nitrogen loading. If the nitrogen were biologically inert and completely soluble, the steady state concentration in the lake would be the ratio of the annual loading and the lake volume divided by the turnover time, or about 0.860 mg L^{-1}. This is about 7 times greater than the measured concentrations for total nitrogen, indicating that as with phosphorus most of the nitrogen entering the lake soon becomes incorporated into the food web. Additional corroboration of phosphorus limitation in the Caldwell Basin is presented by Siegfried and Quinn (1986). It is also noted that since historical measurements were concentrated in the surface layer, the ratio of N:P for the entire lake may be different than calculated above. As a result, more monitoring of the thermocline and hypolimnion zones is recommended.

The reasons for the decline of nitrogen from 1997 to 2009 in the surface waters are not clear. Factors that might contribute include decreased atmospheric deposition of nitrate and ammonia compounds onto the lake or watershed, slight increases in biological productivity, changes in weather patterns, changes in the phytoplankton community composition that potentially alters nitrogen fixing capacity and a decrease from anthropogenic sources. The declining concentration gradient northward indicates disproportionately greater nitrogen inputs at the south, similar to phosphate and chloride. In general, atmospheric deposition of nitrogen compounds declined along with sulfate as a result of implementation of the 1990 Clean Air Act amendments, but the decreases of nitrate measured in the lake are an order of magnitude smaller than the decreases of total nitrogen (compare Figs. 13 and 14). These comparisons corroborate the importance of biological factors that mediate nitrogen cycling within the lake.

SILICA

Soluble silica concentrations varied little either spatially or inter-annually in surface waters of Lake George. The overall mean soluble silica concentration across all six stations over the 30-year monitoring period was 0.91 ± 0.011 mg Si L^{-1} (n = 1,920). The overall mean concentration at Tea Island was 0.96 ± 0.033 mg Si L^{-1} (n = 308), slightly greater than at Rogers Rock where the concentration was 0.88 ± 0.025 mg Si L^{-1} (n = 308), a small but significant difference ($P < 0.001$, $df = 614$, paired t-test). Averaged within each year, mean concentrations exceeded 1.10 mg Si L^{-1} from 1981 – 1983, but ranged from 0.66 – 1.04 mg Si L^{-1} thereafter (Fig. 15), with a period of relatively low concentrations from 1998 – 2003. Mean annual concentrations among stations were highly correlated ($0.64 < r^2 < 0.95$).

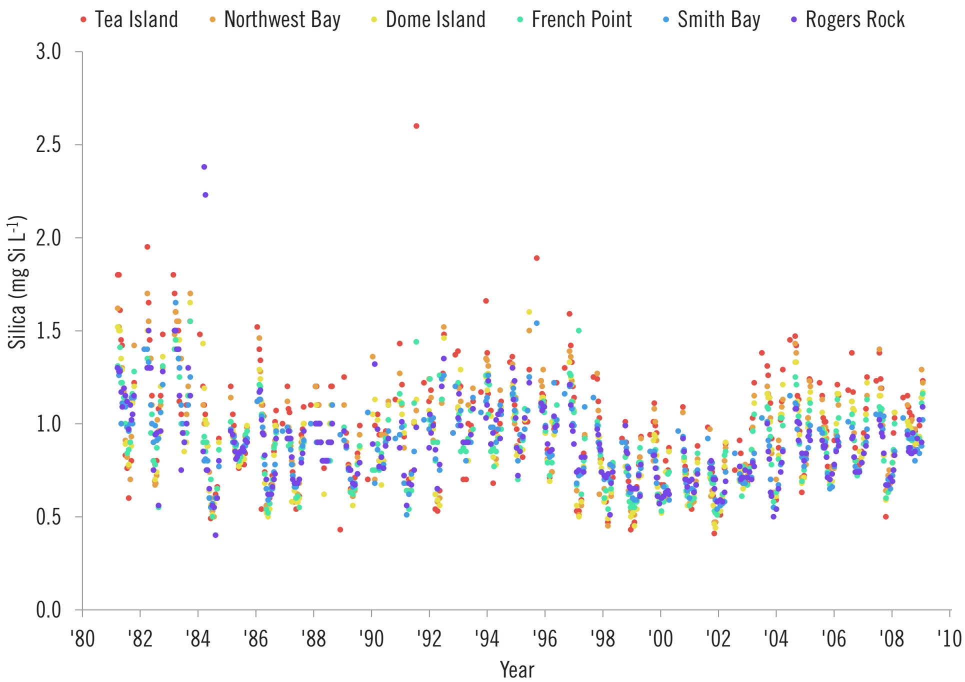

FIGURE 15. *Concentrations of soluble silica (as mg Si L^{-1}) in surface waters (0 – 10 m depth) of Lake George, 1981 – 2009.*

The pattern of seasonal variation of soluble silica resembles that of phosphorus, usually declining from seasonally high concentrations during winter, mid-summer minima, and increases during late summer and early fall (Fig. 16). This depletion results from silica uptake by rapidly increasing diatom populations and by periphytic algal growth during spring and early summer, followed by ingestion by first-order consumers, as noted in an earlier study (Long et al. 1982b).

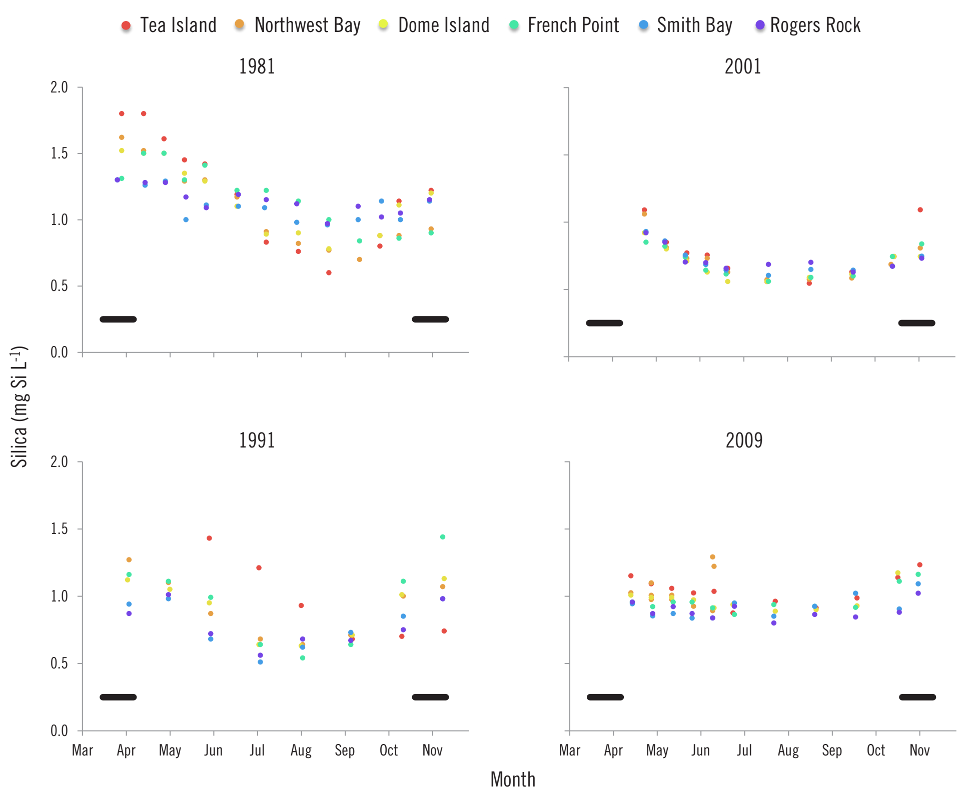

FIGURE 16. *Seasonal variation of soluble silica (as mg Si L^{-1}) in surface waters (0 – 10 m depth) of Lake George. Results from the six monitoring stations are presented for each sampling event. Spring and fall turnover periods are indicated by horizontal black bars.*

Soluble silica availability is typically controlled by the weathering of silicate minerals and the dissolution of skeletal remains of diatoms, both in the lake and watershed. Silica concentrations within the hypolimnion increase during the latter part of stratification until fall turnover when complete vertical mixing occurs. Concentrations in 25 m and 30 m water samples from the Tea Island station may increase by a factor of three during the weeks immediately preceding the fall turnover, when oxygen concentrations may drop below 4 mg L^{-1} (see Section IX). The fall increase of soluble silica (Fig. 16) indicates internal cycling. It is unclear if hypoxia has a direct or indirect effect on this release. A plausible mechanism involves diatoms that sink to the deeper parts of the lake's basin and become coated by concurrent precipitation of $Fe(OH)_3$ while hypolimnetic oxygen remains above hypoxic concentrations, but then dissolve under more hypoxic conditions that are conducive to dissolution of the $Fe(OH)_3$. This may also represent the slow release of soluble silica from the dissolution of sedimented diatom frustrules over time that increase in concentration while isolated from the epilimnion.

SUMMARY

Concentrations of phosphorus and silica show no trend from 1980 through 2009, whereas nitrogen has declined modestly. Part of the decline of nitrogen is likely a result of Clean Air Act regulations that have reduced atmospheric emissions of nitrogen oxides from combustion sources. Comparison of total nitrogen and total phosphorus concentrations in the lake, and of the seasonal variation of these strongly indicates that phosphorus is the limiting nutrient in Lake George, as is typically the case for oligotrophic lakes.

Based on results for phosphorus concentrations, Lake George remains oligotrophic overall. Oligotrophic lakes typically have concentrations of total phosphorus below 12 µg P L^{-1} (Carlson 1977), and total phosphorus in Lake George is about one third of that. However, considering that primary productivity is mainly limited by phosphorus (corroborated in Eiling 1992) and that biologically available phosphorus is quickly incorporated into the food web, the most sensitive indicator of increased phosphorus loading in the lake would be increasing algal production and its effect on water clarity, covered in the following section.

VIII. CHLOROPHYLL AND CLARITY

A LARGE PART OF THE BIOLOGICAL PRODUCTIVITY OF AQUATIC ECOSYSTEMS arises from phytoplankton growth. Populations of these microscopic algae typically rise dramatically during spring in response to increased insolation that provides light to sustain growth and heat to stabilize the water column. These conditions allow phytoplankton to remain in surface waters continuously where inorganic nutrients have been replenished following spring turnover. Phytoplankton populations are eventually limited by declining phosphorus availability, and are further diminished by ingestion by first-order consumers that eventually die off and may settle to the bottom. While standing stocks of phytoplankton are conveniently monitored by measuring chlorophyll extracted from water samples, these chlorophyll concentrations do not necessarily reflect phytoplankton productivity. Standing phytoplankton stocks represent a balance between phytoplankton production and consumption by zooplankton and suspension-feeding organisms, and hence productivity may be considerably greater than what is implied by simple increases of standing stock biomass.

Increasing the supply of phosphorus usually stimulates phytoplankton growth in oligotrophic lakes such as Lake George, which may ultimately lead to eutrophication if not curtailed. One of the consequences of eutrophication is diminished water clarity that results from increased phytoplankton in the water column that absorb and scatter more light. The monitoring results summarized in this section reflect the non-specific biological response of Lake George phytoplankton to the changes of major ions and nutrients since 1980.

CHLOROPHYLL

Annual mean concentrations of chlorophyll, measured as chlorophyll a (Chl a), have been increasing at all six monitoring stations during the monitoring period (Fig. 17). The lake-wide mean concentration increased significantly at a rate of 0.0112 µg Chl a L^{-1} yr^{-1} ($P < 0.01$, $df = 161$; linear regression), or about a 33% increase over the monitoring period. Mean annual concentrations decline from the south to north, with concentrations at the Tea Island station 0.81 µg Chl a L^{-1} higher on average overall than corresponding results at the Rogers Rock station (Table 8). The differences between the Tea Island and Rogers Rock stations within each year are significant ($P < 0.05$, $df > 12$; paired *t*-test) after 1982, except 1991 ($df = 10$) and 2001 ($df = 18$). About a third of this difference occurs between the Tea Island and Dome Island stations (Table 8). This decline reflects the same decreasing south-to-north gradient of concentrations as do many other water quality parameters measured. A spatial concentration gradient for chlorophyll was recognized by the time this monitoring program began (Wood 1981).

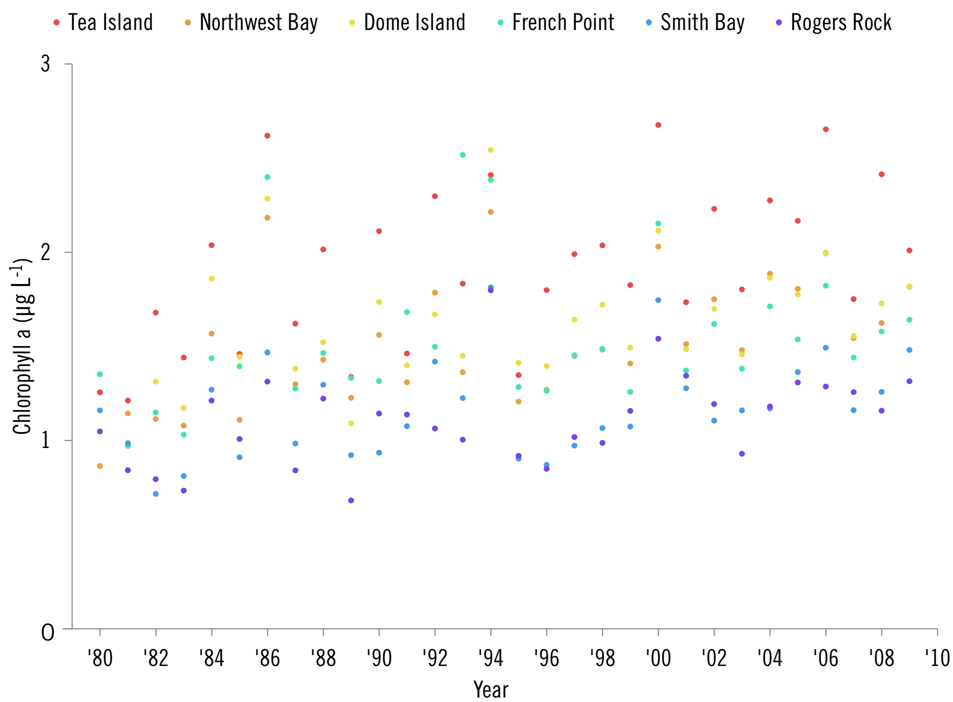

FIGURE 17. *Average annual concentrations of chlorophyll (as µg Chl a L^{-1}) in surface waters (0 – 10 m depth) of Lake George, 1980 – 2009.*

Station	Tea Island	Northwest Bay	Dome Island	French Point	Smith Bay	Rogers Rock
Min	1.21	0.86	0.97	0.97	0.72	0.68
1st Quartile	1.64	1.28	1.40	1.32	0.97	0.95
Median	1.91	1.48	1.54	1.45	1.16	1.14
Mean	1.92	1.52	1.61	1.54	1.17	1.11
3rd Quartile	2.22	1.77	1.77	1.64	1.30	1.25
Max	2.68	2.54	2.54	2.52	1.81	1.80

TABLE 8. *Table of mean, median and quartile chlorophyll concentrations (as µg Chl a L^{-1}) for the six Lake George monitoring stations. Left to right columns correspond with generally down-current station location in the lake, except for Tea Island and Northwest Bay, which are both up-current from Dome Island but not from each other (see Fig. 1b).*

Mean annual chlorophyll concentrations are highly correlated among the stations (Fig. 18), with coefficients of determination within the interval $0.35 < r^2 < 0.85$. A comparison of time-series analyses for the Tea Island and Rogers Rock stations show considerable autocorrelation. This is most pronounced at Tea Island, where autocorrelations at successive lags had opposite signs, reflecting annual oscillations. A deterministic trend at both stations was significant in several simple models. The best fitting model considered was the moving average model (0,1,1) after differencing the data to remove the deterministic trend. The moving average parameter estimate is −0.8194 (standard error of 0.0947) at Tea Island and −0.8556 (standard error of 0.0902) at Rogers Rock. This reflects the pattern of above-average years for chlorophyll alternating with below-average years at Tea Island (Fig. 18). These results confirm the significance of the increasing trend in chlorophyll concentrations based on linear regression, despite the tendency towards alternation of mean chlorophyll concentrations during successive years.

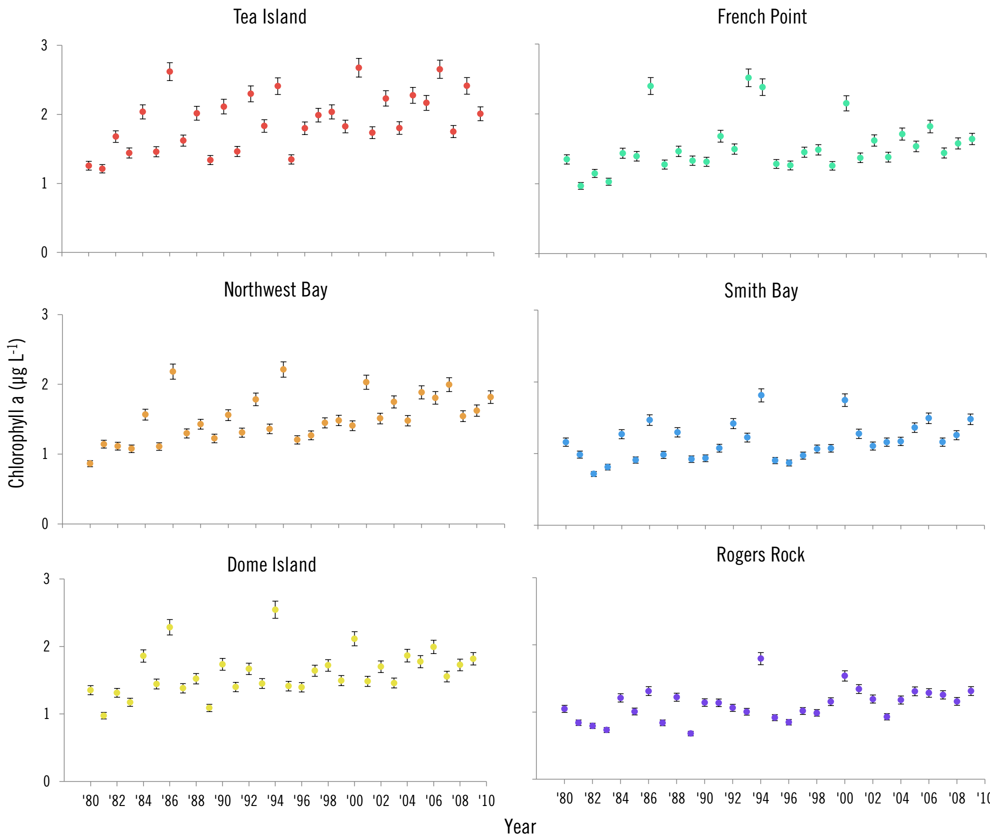

FIGURE 18. *Mean annual chlorophyll concentration (µg Chl a L^{-1}) in surface water (0 – 10 m depth) of Lake George, 1980 – 2009.*

In concert with nutrients (see Sec. VII), chlorophyll concentrations show a pronounced seasonal pattern of relatively high concentrations during spring that decline to a mid-summer minimum followed by an increase during fall (Fig. 19). The high concentrations during spring reflect utilization of nutrients from terrestrial snowmelt, augmented by regenerated nutrients sequestered through winter in the hypolimnion until the spring turnover. These conditions produce a spring phytoplankton bloom, followed by a zooplankton bloom which reduces phytoplankton populations through late spring and early summer. The increased zooplankton populations are in turn cropped by second-order consumers including zooplanktivorous fishes, sessile suspension feeders, insect larvae, etc. As the thermocline deepens into early fall, nutrients previously present in the hypolimnion or released by decaying sessile vegetation become available to phytoplankton in the photic zone (Section IX), supporting the observed fall phytoplankton bloom. As clearly shown in Fig. 19, this seasonal cycle has deepened in recent years, superimposed on the long-term trend of increasing chlorophyll concentrations in the lake.

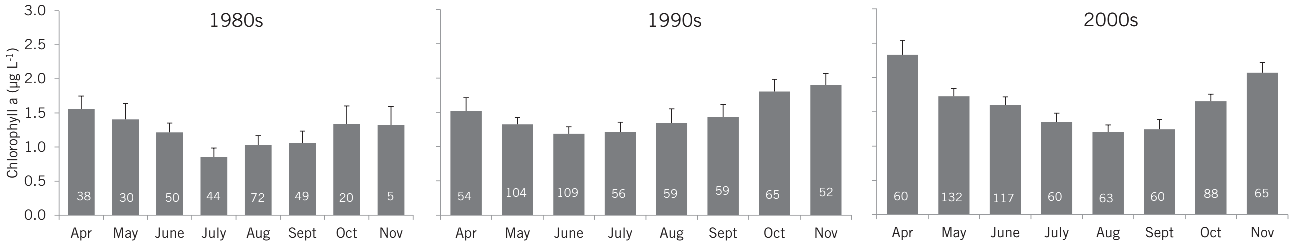

FIGURE 19. *Seasonal variation of chlorophyll concentrations (as µg Chl a L⁻¹) in surface waters (0 – 10 m depth) of Lake George, by decade, from 1980 – 1989; 1990 – 1999; and 2000 - 2009. The number of samples is indicated within each bar.*

WATER CLARITY

The increasing chlorophyll concentrations in Lake George are associated with significantly decreased water clarity. Mean Secchi disk depths have decreased by 0.52 m from 1980 to 2009 ($P < 0.001$, $df = 1{,}548$, linear regression; Figs. 20 and 21). Paired measurements of chlorophyll concentrations and Secchi depths are significantly but weakly correlated ($r^2 = 0.22$, $P < 0.001$, $n = 1{,}470$, linear regression after log-log transformation and omission of 9 outlying chlorophyll a measurements < 0.25 µg L^{-1}). Based on the resulting regression equation ln Secchi depth = 2.21 – 0.189 ln chlorophyll a, an increase of chlorophyll concentration from 1.0 µg L^{-1} to 1.33 µg L^{-1} corresponds to a decrease of Secchi depth from 9.15 m to 8.67 m, or about 0.48 m on average (Fig. 21), consistent with the observed decrease of 0.52 m. However, comparison with a regression equation based on other mainly North American lakes (eq 5 in Carlson 1977: ln Secchi depth = 2.04 – 0.68 ln chlorophyll a) indicates that the surface waters of Lake George are unusually clear for their chlorophyll a content, and that increasing chlorophyll a concentrations in Lake George have less effect on decreasing water clarity compared with the lakes considered by Carlson (1977).

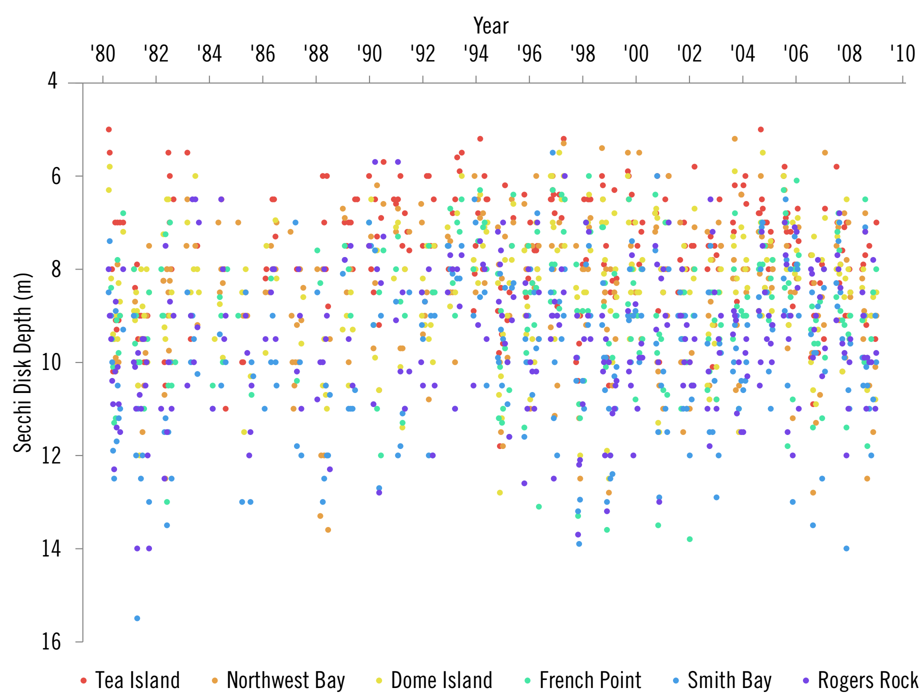

FIGURE 20. *Secchi disk depths at the six monitoring stations in Lake George, 1980 – 2009.*

Secchi depths progressively increased from the south to the north of the lake. At the Tea Island station, the mean Secchi depth was 7.7 ± 0.16 m (*n* = 266), shallower by 1.8 m in comparison with the mean depth of 9.5 ± 0.18 m at Rogers Rock (*n* = 266). About half of this difference occurs between the Tea Island and Dome Island stations. The lower clarity at Tea Island may be the result of higher chlorophyll concentrations along with greater erosion associated with the more developed watershed around the Caldwell Basin, allowing greater stormwater transport of fine-grained particulates into the lake, along with greater re-suspension of benthic sediments within the relatively shallow basin itself during these precipitation events. This south-to-north increase in Secchi depth has been evident since the late1970s (Wood 1981).

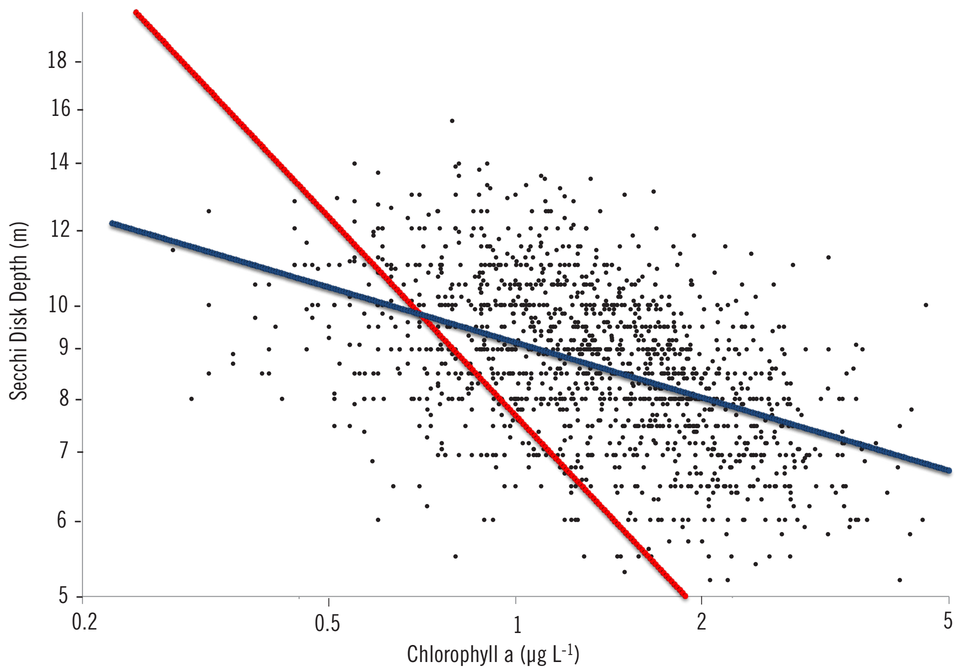

FIGURE 21. *Paired measurements of chlorophyll concentrations and Secchi depths in Lake George, 1980 – 2009. The blue line represents the regression equation ln Secchi depth = 2.21 – 0.189 ln chlorophyll a for Lake George measurements, and the red line represents ln Secchi depth = 2.04 – 0.68 ln chlorophyll a for typical North American lakes (i.e. eq 5 in Carlson 1977).*

If the ~33% increase in the standing stock of phytoplankton since 1980 implied by the chlorophyll a measurements is a response to increased nutrient loading to the lake, a concurrent expansion of the lake's *Nitella* meadows would provide corroborating evidence, especially given the much greater standing stock biomass associated with *Nitella* compared with phytoplankton (Stross et al. 1988). A quantitative comparison of standing stocks with estimates from the 1980s may therefore provide an especially sensitive indication of increased nutrient loading since then. Conversely, changes in the fish community may exert trophic cascade effects that result in less zooplankton grazing on the phytoplankton community, which could plausibly account for the increased chlorophyll concentrations observed (Siegfried and Quinn 1986). Short-term turbidity and nutrient loading due to stormwater runoff might also affect the sessile *Nitella* population differently than phytoplankton, for example through differential responses to shading and nutrient uptake.

Changes in the phytoplankton community structure may also affect long-term changes in water clarity. Establishment of a substantial cyanobacteria community by the early 1980s (Siegfried and Quinn 1986) during late summer/early fall may be a consequence from a bottom-up or top-down trophic cascade. Predation of zooplankton by rainbow smelt could be responsible for the observed increase, as well as a shift in nutrient loading or a change in the proportion of nitrogen to phosphorus, which could have led to increasing abundance of cyanobacteria in later years. Cyanobacteria remained an important component of the phytoplankton during the late 1990s, especially during late summer and early fall (Richardson 1999). Given the overall nutrient limitation by phosphorus, it is unlikely that the increased abundance of cyanobacteria reflect limitations imposed by nitrogen, because in such conditions the nitrogen-fixing ability of cyanobacteria would confer scant competitive advantage.

SUMMARY

As expected for an oligotrophic lake, the long-term increase of chlorophyll concentrations is broadly commensurate with possible increases of phosphorus loading since 1980. Development has been closely linked to increased phosphorus loading (Sutherland et al. 1983) and with a 62% increase in buildings between 1980 and 2009 a commensurate increase in anthropogenic phosphorus loading is likely. Alternatively, part of the chlorophyll increase may have resulted from a trophic cascade effect caused by increased abundance of planktivorous fish that reduce zooplankton grazing on phytoplankton. Whatever the cause, phytoplankton increases have as yet only modestly affected water clarity throughout most of the lake. Phosphorus loading may also have stimulated increases in the larger carbon pools comprising nuisance algal and plant growth associated with nutrient sources near shorelines, and the *Nitella* meadows farther offshore. However phytoplankton populations in the Caldwell Basin have clearly responded to the increases of phosphorus loading there inferred from the higher concentrations of phosphorus, the density of the human population in the drainage and the considerably higher standing stock biomass of *Nitella* and littoral plants in the south basin compared with the north (Stross et al. 1988). If not altered, present trends of increasing phytoplankton abundance could significantly degrade water clarity in the future, and any such continued degradation will most likely first become evident in the Caldwell Basin.

IX. TEMPERATURE AND OXYGEN

TEMPERATURE PLAYS A FUNDAMENTAL ROLE modulating physical, chemical and biological processes in lakes. Along with salt, temperature is crucial for elucidating water mass movement within lakes, providing a direct means of monitoring thermal stratification and destabilization of the water column. Chemically, temperature determines the solubility of oxygen and carbon dioxide in water, which directly affects biological processes. Moreover, biological processes themselves are highly temperature dependent, especially in plants and cold-blooded organisms. For all of these reasons and many more, temperature is a fundamental parameter of any lake monitoring program.

The distribution of dissolved oxygen is also crucial for evaluating the ecological functioning of a lake. While temperature determines oxygen solubility, biological processes can deplete the available oxygen, with potentially drastic consequences for resident biota. In this section we summarize the monitoring results for temperature and oxygen, focusing especially on the development of hypolimnetic hypoxia during summer stratification.

TEMPERATURE

As a temperate lake, Lake George surface water temperatures typically range from 0 °C directly beneath ice cover during winter to nearly 25 °C during late summer (Fig. 22). Surface water temperatures of the integrated 0 – 10 m depth samples at the six monitoring stations were highly correlated, with $r^2 > 0.83$ ($df = 594$), reflecting a largely coherent thermal response of surface water to heating by insolation and cooling by evaporation and radiative heat loss. Despite this coherence, mean temperatures increase from the south to the north end of the lake, with the mean temperature at the Rogers Rock station about 1.8 °C warmer than at the Tea Island station ($P < 0.001$, $df = 594$; paired t-test; Table 9).

Station	Tea Island	Northwest Bay	Dome Island	French Point	Smith Bay	Rogers Rock
Min	9.00	10.8	10.7	10.7	10.8	12.1
1st Quartile	12.2	12.7	13.0	12.8	12.7	13.5
Median	12.4	13.1	13.4	13.5	13.3	13.9
Mean	12.5	13.4	13.4	13.4	13.4	14.3
3rd Quartile	13.0	13.6	13.8	13.9	14.0	15.0
Max	14.8	17.8	16.1	15.8	17.6	17.1

TABLE 9. *Table of mean, median and quartile temperature measurements (as °C) in surface waters (0 – 10 m depth) at the six Lake George monitoring stations. Left to right columns correspond with generally down-current station location in the lake, except for Tea Island and Northwest Bay, which are both up-current from Dome Island but not from each other (see Fig. 1b).*

Overall surface water temperatures integrated over the uppermost 10 m have increased significantly by about 1.8 °C from 1980 to 2009 ($P < 0.001$, $n = 1{,}764$; linear regression: Fig. 22). Time-series analysis revealed little autocorrelation or partial autocorrelation, and the data did not fit simple time-series models well. The models that fit most closely were random-walk and moving average models, implying random inter-annual fluctuations of mean temperature superimposed on a consistently increasing trend.

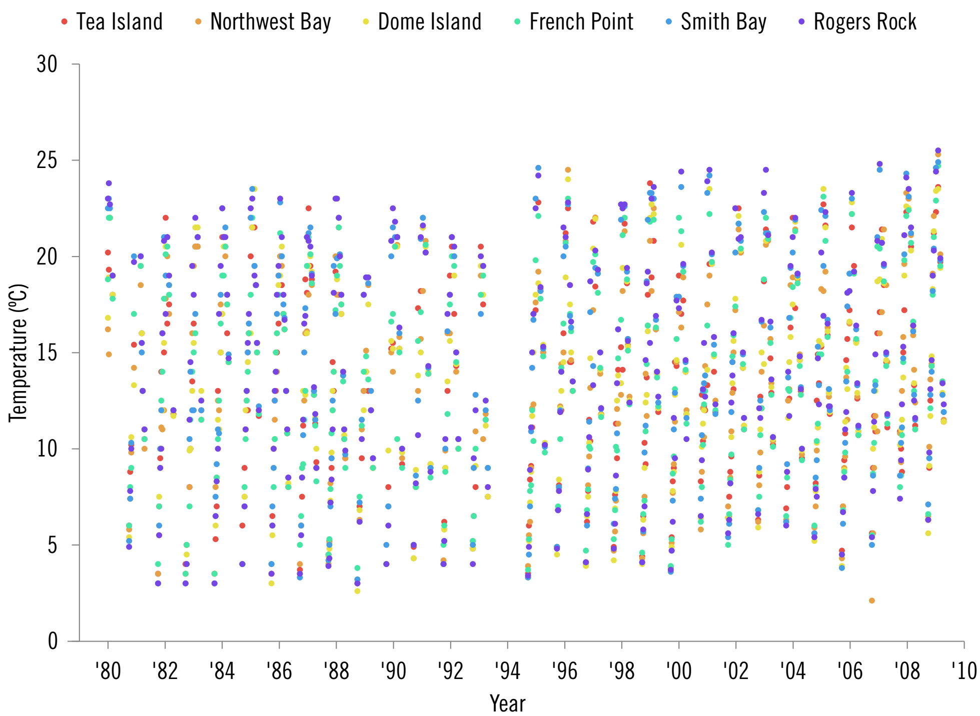

FIGURE 22. *Temperature (°C) of surface waters (0 – 10 m depth) of Lake George, 1980 – 2009.*

Insolation during early spring usually establishes a weak, shallow thermocline by mid-May, which grows stronger and deeper as the season progresses. Thermal profiles with depth at the Tea Island station (Fig. 23) illustrate concurrent patterns at the other stations. Maximum surface temperatures are usually attained during late July - early August, associated with a mixed layer extending to 10 m depth above a sharp thermocline. Thereafter the thermocline weakens and the mixed layer deepens to ~20 m just prior to the fall turnover. This seasonal thermal response of the lake is typical for a temperate, mid-latitude lake of moderate depth.

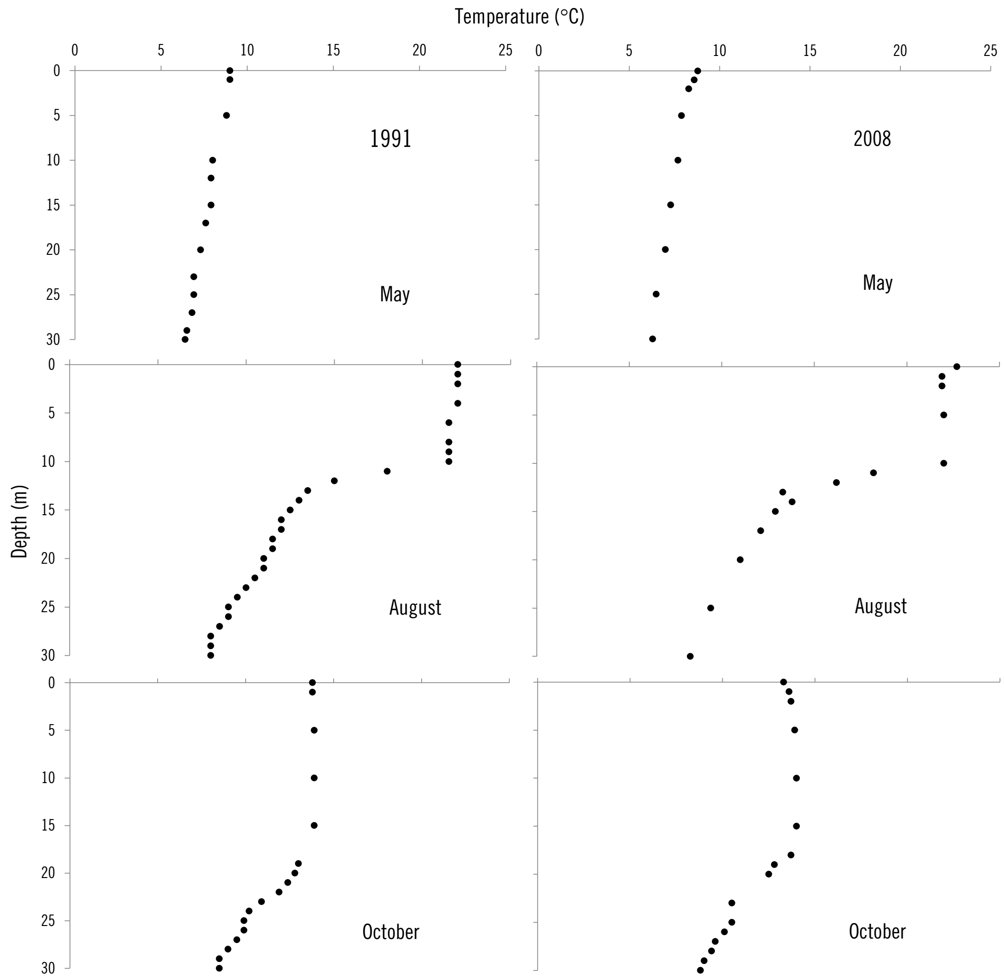

FIGURE 23. *Comparison of temperature profiles with depth at the Tea Island station, May, August and October in 1991 with 2008 in Lake George.*

OXYGEN

The solubility of oxygen in water is strongly affected by temperature, declining from 14.2 mg L^{-1} at 0 °C to 8.1 mg L^{-1} at 25 °C (Truesdale et al. 1955). Oxygen concentrations throughout the lake thus naturally decline as it warms during spring and summer. Oxygen released by photosynthesis into warming waters may lead to supersaturation in parts of the water column such as the metalimnion where phytoplankton biomass is high. An increase in the phytoplankton biomass may be accompanied by an increase in productivity which would result in the elevated oxygen concentration observed. Much of this metalimnetic standing crop during late summer and early fall is associated with cyanobacteria (Richardson 1999).

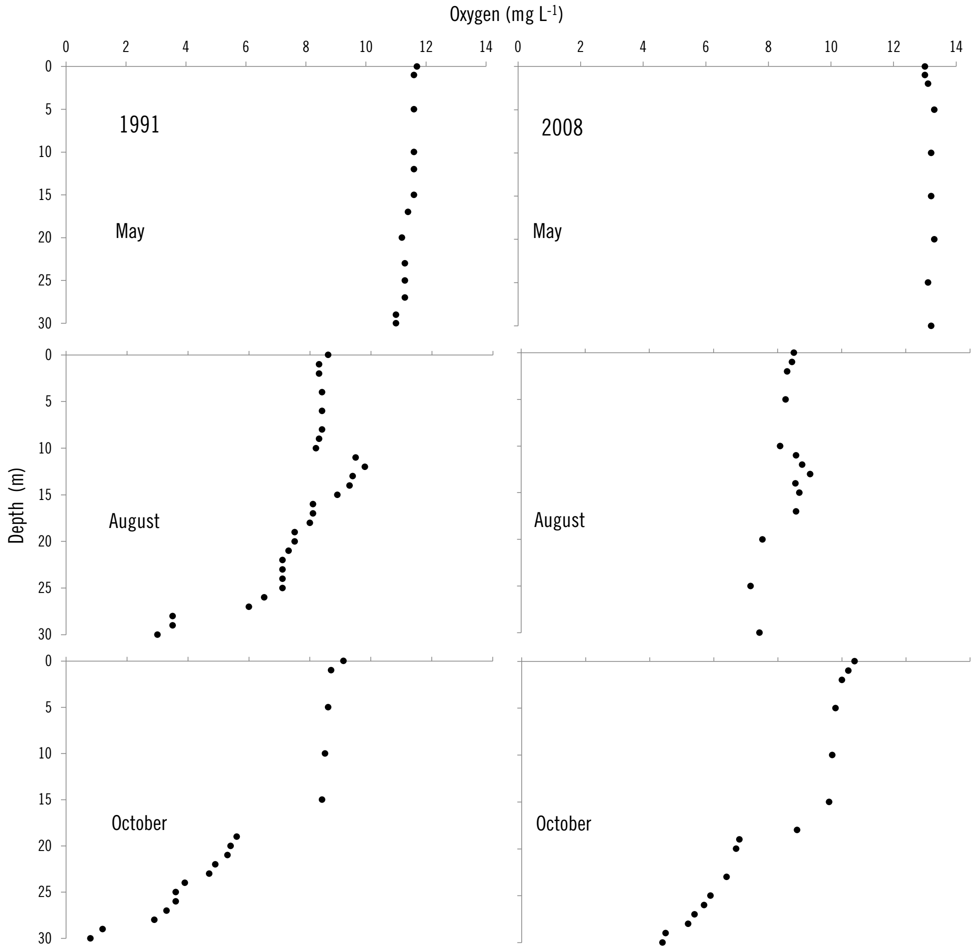

FIGURE 24. *Comparison of dissolved oxygen profile with depth in Lake George at the Tea Island station, May, August and October during years with strong (1991) and weak (2008) hypoxia in bottom waters.*

As with temperature, oxygen profiles with depth at the Tea Island station (Fig. 24) illustrate concurrent patterns at the other stations, although the similarities are not as close owing to the deeper basins elsewhere. Oxygen concentrations are slightly supersaturated in the cool waters of the well-mixed water column during spring, as the spring phytoplankton bloom adds oxygen to the warming waters. By mid-August, oxygen concentrations decline in the well-mixed epilimnion, but an increase relative to the epilimnion is evident within the thermocline, reflecting phytoplankton utilization of nutrients there, especially by cyanobacteria (Richardson 1999). The oxygen increase within the thermocline persists for about 5 m at the Tea Island station, but extends below 25 m at stations in the deep basins. Near the bottom, oxygen depletion below 25 m by mid-August is evident at the Tea Island station (Fig. 24). By mid-October just before the fall turnover, oxygen depletion is evident throughout the hypolimnion at Tea Island, for example with concentrations declining to 0.8 mg L^{-1} at 30 m depth in 1991 when strong hypoxia developed during fall, compared to 4.0 mg L^{-1} in 2008 when fall hypoxia was weak (Fig. 24, see also Fig. 25). The average minimum dissolved oxygen concentration over the 30 years of monitoring was 2.6 with a range from 0.8 to 5.6 mg L^{-1}.

Hypoxic bottom waters support the chemical release of phosphorus (Eiling 1992). This additional input of nutrients supplies an added increment to a phosphorus-limited system and may increase the productivity of the lake. The appearance of hypoxic bottom water within the Caldwell Basin during fall prompts concerns that such processes are in progress. Extensive surveys of oxygen concentrations in Caldwell Basin were conducted in 1991 and in 2010. These surveys found oxygen concentrations below 3 mg L^{-1} above 281 ha of the deepest part of the basin in 1991 (Eiling 1992), compared with 235 ha in 2010 (Eichler and Boylen 2011).

The oxygen-depth profiles for the Tea Island station provide additional insight regarding the intensity of the seasonal hypoxic zone in Caldwell Basin. The profiles illustrated in Fig. 24 can be used to estimate the depth corresponding to the 4 mg L^{-1} horizon for each year of the monitoring program after 1980. These estimated horizons are presented in Fig. 25. The year 1991 turned out to be the year the hypoxic zone was greatest during the entire 29-year record, with oxygen concentrations less than 4 mg L^{-1} at depths below 23.9 m. During four other years, 1983, 1989, 2003 and 2008, the < 4 mg L^{-1} hypoxic zone was deeper than 30 m, confining it to the deepest 1.8 m of the basin if it developed at all. These results indicate the intensity of the seasonal hypoxic zone in Caldwell Basin is quite variable, ranging from deeper than 30 m to as shallow as 24 m.

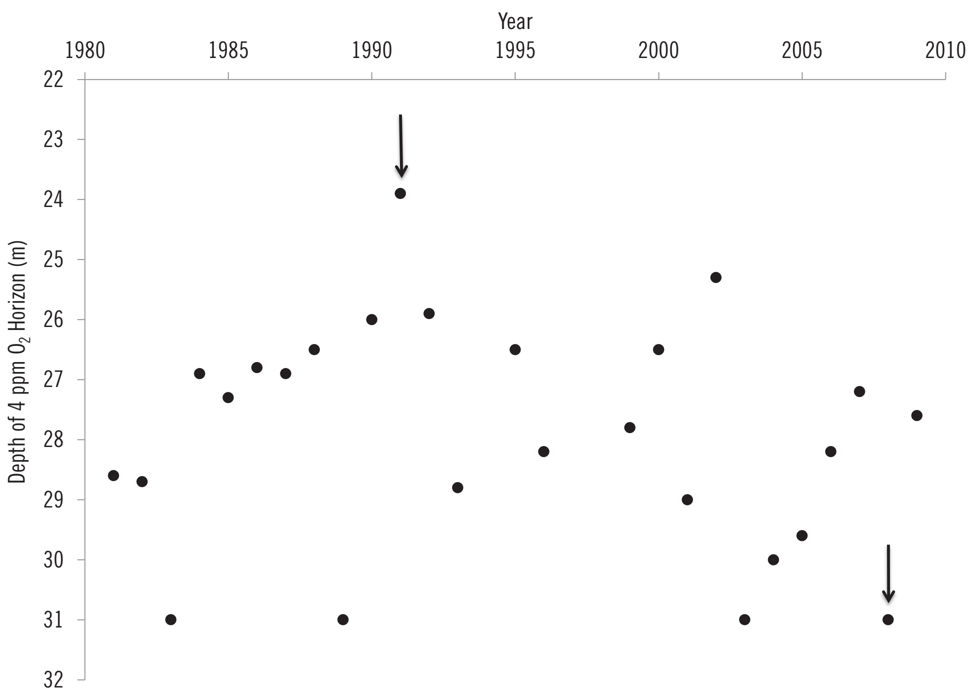

FIGURE 25. *Estimated minimum depth of 4 µg L^{-1} oxygen horizon during October at the Tea Island station, Lake George, from 1981 – 2009. Missing data from 1994, 1997 and 1998 precluded calculation of the estimate for those years.*

Comparison with earlier studies suggests the seasonal hypoxic zone in Caldwell Basin may have been larger prior to the period of the monitoring program. Clesceri and Williams (1972) report mid-October oxygen concentrations 4 mg L^{-1} or less at depths of 21 m or shallower for the three successive years covered by their measurements (1968, 1969 and 1970). Hypoxia at this depth implies considerably greater affected lake bottom area and volume of water compared with the years after 1980. This in turn suggests that measures undertaken after 1970 to reduce nutrient loading into Lake George, especially regulations limiting the phosphorus content of household detergents, have likely averted more substantial deterioration of water quality.

Development of hypoxia during late summer and early fall in the hypolimnion of Lake George sub-basins depends on the amount of organic loading in relation to sub-basin shape and depth. Caldwell Basin is especially susceptible to hypoxia because organic loading is likely greater there, and the sub-basin is relatively shallow, concentrating the decaying organic matter in a relatively limited amount of stagnant bottom water during late summer and early fall. However, hypoxia may also develop when similar conditions are found in the deepest parts of other sub-basins in the lake such as off Dome Island and near Roger's Rock (Fig. 1b; see also Sec. VII). A more rigorous comparison of eutrophication stress among the basins of Lake George would rely on estimates of relative areal oxygen deficits. Water column monitoring that extends to the deepest part of each basin would provide the necessary temperature and oxygen data needed to calculate these deficits. Until such comparisons are available, conclusions regarding eutrophication tendencies within the different basins must be made with caution. But nonetheless, as one of the shallower basins and situated at the headwater of the lake, and receiving the most nutrient loading per unit area, Caldwell Basin is clearly the most vulnerable to eutrophication. This basin therefore warrants close monitoring in the future, because adverse changes in the lake's water quality are likely to appear there first.

SUMMARY

The annual progression of spring turnover, summer stratification, early fall stagnation followed by another turnover is clearly evident in Lake George, as expected for a temperate, mid-latitude lake of moderate depth. Surface water temperatures have increased by about 1.8 °C during the 30-year monitoring period, and are strongly correlated among the monitoring stations. The thermocline is usually established during June, is strongest in mid-August with a 10-m mixed layer depth, and then weakens and deepens to 20 m as the fall turnover is approached in early November.

Supersaturated oxygen concentrations appear in the upper water column in conjunction with the spring phytoplankton bloom, and once nutrients are exhausted in the mixed layer, most algal production is concentrated in the thermocline for the remainder of the summer, leading to a positive heterograde oxygen curve with depth. Much of this metalimnetic production during summer and fall is associated with cyanobacteria that became established at least in part because of chemical and biological changes in the lake. At the height of stratification in mid-October, a hypoxic zone of variable size usually develops in the deepest portions of Caldwell Basin, and possibly elsewhere in the lake. Understanding the extent, persistence and intensity of hypoxic zones in Lake George is a high research priority because it can have a substantial effect on the extent of internal nutrient cycling, and hence of lake productivity.

X. CONCLUSIONS

While some of the threats to Lake George water quality have receded since 1980, others are worsening. Fortunately, the water quality of the lake remains exceptional, largely as a result of prudent management efforts in the past, which have preserved management options for coping with current stresses. The following represent the chemical and physical changes to Lake George observed over the past 30 years:

1. Salt concentrations within Lake George have nearly tripled between 1980 and 2009. Although it is not yet clear to what extent the increase in salt is affecting the functioning of the lake, circulation is impacted and data from other Adirondack lakes suggest phytoplankton community composition may be influenced at current salt concentrations.
2. Chemical, physical and biological gradients exist in the lake, including salt (NaCl), phosphorus, and chlorophyll which decrease northward and clarity which increases along the same gradient, an observation confirming earlier work. These gradients are linked to heavier development and chemical loadings in the south end of the lake, which serves as the lake's headwater.
3. The amount of phytoplankton in the lake, measured in terms of chlorophyll, has increased by 33% between 1980 and 2009. The cause(s) of this increase may include greater nutrient loading, changes in the zooplankton community due to the introduction of rainbow smelt or a decline in competition for nutrients by other primary producers.
4. Clarity within the lake, based on Secchi disk measurements, has declined by 6%, with Tea Island water clarity averaging 1.6 m shallower than Rogers Rock over the 30 years. The south-to-north gradient within the lake has not changed substantially since 1980.
5. Hypolimnetic hypoxia usually develops in the Caldwell basin during late summer/early fall due to microbial decomposition depleting dissolved oxygen.
6. As a result of the Clean Air Act amendments, sulfate concentrations within the lake have decreased substantially with buffering capacity and pH increasing as a result. While only a minor decrease in nitrate was detected, total nitrogen declined more substantively.
7. Surface water temperature has increased ~1.8 °C between 1980 and 2009, possibly leading to a longer growing season and potentially extending the period of lake stratification.
8. The monitoring results corroborate phosphorus limitation of primary productivity as found by earlier studies.

As recognized at the outset of the Offshore Chemical Monitoring Program, the greatest vulnerability of the lake's water quality remains its sensitivity to nutrient loading, especially phosphorus. The interacting responses of phytoplankton and *Nitella* to increased nutrient loading could potentially lead to serious consequences for the ecological functioning of Lake George. The lake may already be deceptively productive despite its highly transparent waters owing to competition between *Nitella* and phytoplankton for phosphorus (Stross 1981). Inhabiting water depths of 7 – 16 m, *Nitella* depends on high water clarity for sufficient illumination to sustain photosynthesis. Continued increases of phytoplankton abundance with concomitant decreases of water clarity could eventually trigger a positive feedback wherein loss of *Nitella* from increased shading increases phosphorus availability to phytoplankton, thereby increasing shading and further loss of *Nitella* (Stross 1981). Once engaged, this feedback could rapidly reduce the lake's water clarity and dramatically alter its food web, generally improving conditions for cyanobacteria and invasive plants and animals. Proper appreciation for this vulnerability prompts evaluation of the current size and health of the *Nitella* meadows in Lake George, as the last survey was done in the early 1980s (Stross et al. 1988).

At present, the cause of the observed 33% increase of chlorophyll associated with phytoplankton over the last three decades remains unclear. Gradually increasing phosphorus loading associated with population growth in the watershed and increasing recreational use of the lake since 1980 is likely at least a contributing factor. However, trophic cascade effects associated with the rainbow smelt stocking is another potentially important factor. Monitoring the size and health of the *Nitella* meadows may help to constrain contributions from these two causes. If sustained increases of phosphorus loading are primarily responsible for the observed increase of phytoplankton chlorophyll, then a comparable increase in the *Nitella* standing stock biomass would be expected since both compete for phosphorus. To properly assess the current state of *Nitella* in Lake George a comprehensive lake-wide assessment is needed and a commitment to a long-term study to better understand its impact and life cycle in Lake George is required. The spatial and temporal variability of macroalga pose more restraints than rooted macrophytes because macroalga can reproduce both vegetatively and through oospores. Conversely, increasing phytoplankton, but unchanged or decreased *Nitella* biomass, would suggest trophic cascade effects as the primary cause of the observed phytoplankton chlorophyll increases, but would require corroboration through fishery studies to identify the extent of predation and if rainbow smelt are responsible. Distinguishing between these two causes is crucial because they prompt different management responses. If increasing phosphorus loading is the implied cause of the increasing phytoplankton chlorophyll, then the appropriate management response would be aimed at reducing these loadings. But if trophic cascade effects are implicated, then management responses should focus on the ecological effects of fish stocking programs, beginning with rainbow smelt.

Sustained increases of salt loading to Lake George pose a gathering threat to the lake's ecological health. Already at a threshold that may alter the structure of the lake's diatom community, further increases in salt loading may cause other biological responses to the natural community assemblages of the lake which are not well understood. Increasing salt loading likely affects the lake's physical circulation by increasing vertical density gradients that are more difficult to mix. Moreover, approximately 26,000 tonnes of sand are applied along with salt for roadway ice management during winter, and eventually wash into the lake to build near-shore deltas. These deltas provide rearing habitat for invasive species such as Asian clams and may introduce more phosphorus to the lake in the form of small amounts of apatite contributing to the mineral composition of the sand. This may eventually affect both the water quality and the ecological health of Lake George.

The trend of increasing water temperature at about 0.06 °C yr^{-1} imposes unavoidable stresses on Lake George. Warming water temperatures may increase primary production in the lake as well as more undesirable phytoplankton species including cyanobacteria. Warmer water promotes a longer growing season, and extends the duration of the stratification. Extending the length of time hypolimnetic water is isolated from the epilimnion can lead to a worsening of the hypoxia in the Caldwell Basin and perhaps in other basins of the lake.

Overall, Caldwell Basin in the most vulnerable part of the lake to anthropogenic stress because its relatively small water volume and shallow basin depth receives a disproportionate share of the lake's salt and nutrient loading from the extensive upland settlement along the Basin's shoreline. Consequently, Caldwell Basin warrants especially close monitoring because adverse trends are likely to be detectable there first.

The concerns that led to the Offshore Chemical Monitoring Program in 1980 also led to extensive management efforts to address them, in Lake George and in other lakes throughout the United States and Canada. These efforts, both regulatory and voluntary, were largely successful, and averted much more serious damage to these ecosystems from excessive nutrient loading and from acidification caused by atmospheric deposition of nitric and sulfuric acids. Results from the Offshore Chemical Monitoring Program suggest that Lake George had begun recovering from excessive nutrient loading during the decades prior to inception of the program in 1980. Monitoring results supporting this conclusion include the lower concentrations of phosphorus during the monitoring program compared with those typical of the 1960s and 1970s, the N:P ratio indicating clear phosphorus limitation of primary productivity throughout the lake, and the decreased extent of hypoxic bottom waters prior to fall turnover in Caldwell Basin during the period of the monitoring program compared with the measurements in the late 1960s. Despite worsening modestly since the 1980s, these earlier improving trends with respect to eutrophication reflect the success of regulatory limitations on the phosphorus content of household detergents enacted in 1976. Subsequent Clean Air Act restrictions on emissions of sulfur and nitrogen oxides have reduced the acidification threat to the lake and have materially reduced nitrogen loading from precipitation. The clear success of these efforts confirms their utility in maintaining the high water quality and ecological stability of Lake George and other lakes like it.

Despite some past successes, additional efforts are needed to maintain the outstanding water quality of Lake George. The lake will require sustained vigilance to act on every opportunity to reduce nutrient loadings, especially of phosphorus. Similarly, steps taken now to reduce salt loading in the Lake George watershed will help preserve the lake's resilience to other environmental stresses imposed by rising water temperatures and invasive species. A better understanding of the cascading trophic effects of introduced and stocked fishes, as well as of other invasive species, will serve to focus management responses on more efficient and practical solutions. Alternatively, failure to adequately manage the threats currently confronting the lake would have clear and predictable consequences for the water quality of the lake, and consequently for the region's economy and property values within the watershed.

Recognition of the environmental threats currently facing Lake George has inspired a redoubled commitment to protecting the lake that features The Jefferson Project at Lake George. Informed by the results of the Offshore Chemical Monitoring Program, The Jefferson Project is a collaboration of Rensselaer Polytechnic Institute, The Fund for Lake George and IBM Corporation to establish a more extensive and sophisticated environmental monitoring program for the lake aimed at informing management efforts to reverse negative environmental trends and protect the lake for the continued benefit of generations to come.

XI. RECOMMENDATIONS

The results of the 30 years of the Offshore Chemical Monitoring Program, evaluated in the context of previous and concurrent studies in the Lake George region, serve to inform those responsible for management of the lake and to identify research questions where additional information is most needed. Recommended monitoring and research needs enumerated below are intended to help guide future monitoring undertaken by The Jefferson Project at Lake George, and to link those efforts effectively to immediate as well as longer-term lake management.

1. The long-term monitoring program of the lake should be modernized and, when necessary, expanded to address specific management concerns. At a minimum, sample collection locations, frequency of collection and analytical methods should be comparable with the monitoring reported herein to preserve the unbroken time series already established for at least some of the offshore sampling stations. Sample collection in the metalimnion should be implemented, and current sampling locations should be supplemented or moved to the deepest point within each basin so that the maximum depth within the major basins of the lake is sampled. Furthermore, increasing sampling frequency and the number of locations during winter would be beneficial to support characterization of annual cycles throughout the lake.

2. A thorough comparison of phytoplankton and zooplankton abundances by season in the south and north basins should be undertaken and compared with results from similar efforts in the early 1980s as reported by Siegfried and Quinn (1986). In particular, such surveys should focus on whether the differences in abundances of large-bodied zooplankton between the south and north basins persist, and whether cyanobacteria still dominate the plankton community from mid-summer through the fall turnover. This effort will provide additional context for evaluating whether trophic cascade effects account for observed increases in phytoplankton chlorophyll, along with an assessment of the population size of the spiny waterflea (*Bythotrephes longimanus*), a recent invasive zooplankton species. The current productivity of phytoplankton should be accurately measured, with separate measurements for the cyanobacteria to permit direct comparisons among these major sources of primary productivity to the lake. This should be a continuing effort at a minimum of one or two sampling sites in each basin. These measurements will also allow estimation of an upper limit to the lake's carrying capacity for fish.

3. A detailed circulation study is needed for Lake George to better understand how nutrients, phytoplankton, salt and other contaminants move within the lake, as well as to determine their effective residence times in the epilimnion and hypolimnion of the major basins. In addition, even a first-generation circulation

model would provide crucial guidance for developing a more efficient monitoring program, and for identifying unusual but important episodic events that modulate the ecological functioning of the lake. Detailed mapping of the lake's bathymetry, undertaken as the first major study of The Jefferson Project, provides the necessary foundation for developing a circulation model.

4. Detailed hydrologic and chemical budgets should be prepared for Lake George by incorporating atmospheric, groundwater and surface water inputs, along with storage and exports to identify sources and magnitudes of nutrient loadings, where they enter the lake, and how they vary seasonally. This information will improve estimates of threats to the lake's water quality, especially from eutrophication. Monitoring flow inputs, temperature and concentrations of cations, anions, alkalinity, dissolved organic matter and acidity will be needed to develop a hydrologic model, and to better understand how these inputs affect the lake's circulation and productivity. This should include detailed nutrient and salt budgets, and proportions of phosphorus, nitrogen and silica supplied from allochthonous versus autochthonous sources, especially during spring and fall overturn events. A detailed salt budget characterizing specific sources and pathways of salt to Lake George would provide a framework for remediation. A one-time study of roadside runoff would be useful to better understand any phosphorus loading from sand.

5. Estimates of standing stock biomass of the major functional biotic groups, and of the flow of carbon among these trophic groups, should be organized within the framework of a quantitative food web model. This food web model will provide a summary of current understanding of the lake's ecological functioning, an indication of associated uncertainties, and a consistent basis for anticipating how the lake's ecosystem would respond to perturbations. Coupled with the circulation and hydrologic models, the food web model can also provide useful constraints on public interest questions such as the carrying capacity of the lake for game fish, or the scope for perturbations caused by invasive species. As part of this effort, nutrient utilization and flux should be evaluated experimentally for primary producers, including phytoplankton, benthic algae (with a focus on *Nitella*), cyanobacteria and macrophytes. Benthic and pelagic bacterial communities and productivity should also be investigated because of the importance of the decomposer component of the food web in the turnover of nutrients. Primary productivity changes in the lake should be evaluated in light of increased nutrient loading, warmer water, invasive species and/or speciation changes. Other trophic levels up to fish and other apex predators should be included so that both top-down and bottom-up processes can be evaluated simultaneously and on an equal footing. In order to validate a food web model, detailed nutrient budgets, phytoplankton, zooplankton and biota distributions are required over a minimum of a two-year period (model validation period). This will necessitate the implementation of a significantly larger monitoring program compared to Item 1 above. It is anticipated that the temporal and spatial (especially vertical) resolution of the sampling will be greatly increased during the model validation period.

6. The biological responses to increased salinization of Lake George, its tributaries, soils and groundwater should be explored through field studies, laboratory, and mesocosm experiments to evaluate the effects of different salt concentrations to the ecological functioning of the lake, especially its photosynthetic communities. Simultaneously applying relevant research from other locations, including Lake Champlain, efforts to reduce salt loading through the use of best management

practices should be given high priority, employing the improved monitoring facility of The Jefferson Project to document results.

7. Given the important ecological role of the *Nitella* meadows, the littoral vegetation and the periphyton communities, the area and standing stock biomass of these plant communities should be determined quantitatively for comparison with previous estimates in the 1980s and 1920s (Needham et al. 1922, Stross et al. 1988). Changes in the standing stock biomass and productivity of *Nitella* meadows provide crucial context for interpreting changes in phytoplankton chlorophyll and associated decline in water clarity. Also, the primary productivity of these plant communities should be estimated to more accurately determine their contributions to the overall annual productivity of the lake.
8. A quantitative assessment of the stock of rainbow smelt and of their indigenous trophic competitors (*e.g.* Ciscoes) should be conducted to evaluate whether they may still be affecting phytoplankton chlorophyll through trophic cascade effects.
9. Sediment coring studies across a lake bottom gradient from littoral zones to areas of maximum depth should be undertaken to characterize the types of phosphorus storage in the sediments. The form of phosphorus storage determines its availability for re-distribution to the water column.
10. Some formal means for explicitly comparing results of water quality monitoring and other related studies for Lakes George and Champlain, as well as the lakes monitored by the Adirondack Lakes Survey Corporation should be identified and pursued at the earliest opportunity. An annual workshop, symposium or other venue that allows researchers in the region to present and interpret their findings relevant to these lakes would have considerable value, scientifically and for improving lake management efforts. Ultimately, future research should be pursued in the context of the larger Lake Champlain watershed to gain a complete picture of ecological conditions and management implications.
11. Expanded engagement of the limnological research community, in the U.S. and beyond, should be actively developed, drawing on expertise and pathways of inquiry to produce enhanced understanding of the complex process defining the present and projected health of Lake George, while also establishing the Lake as a world-leading research destination. As the “Queen of American Lakes,” this would present a natural and necessary progression of growing importance.

REFERENCES

Aulenbach, D. 1979. Nutrient budgets and the effects of development on trophic conditions in lakes. Presented at Lake Symposium III, Practical Lake Management, Boston College, February 21, 1979. Fresh Water Institute Report 79-2.

Aulenbach, D.B. and N.L. Clesceri. 1972. Sources and Sinks of Nitrogen and Phosphorus: Water Quality Management of Lake George (N.Y.). Rensselaer Freshwater Institute at Lake George. Fresh Water Institute Report 72-32.

Aulenbach, D.B., Clesceri, N.L. and J.R. Mitchell. 1981. The impact of sewers on the nutrient budget of Lake George, NY. *in*: C.W. Boylen (ed.). The Lake George Ecosystem, Vol. I. Lake George Assn., Lake George, NY.

Baker, J.P., D.P. Bernard, S.W. Christensen, M.J. Sale, J. Freda, K. Heltcher, D. Marmorek, L. Rowe, P. Scanlon, G. Suter, W. Warren-Hicks and P. Welbourn. 1990a. Biological effects of changes in surface water acid-base chemistry. NAPAP Report 13, vol. II. In: National Acid Precipitation Assessment Program, Acid Deposition: State of Science and Technology, Washington, DC.

Baker, J.P. and S.A. Gherini, eds. 1990b. Adirondack Lakes Survey, an Interpretive Analysis of Fish Communities and Water Chemistry, 1984 – 1987. Adirondack Lakes Survey Corporation, Ray Brook, New York.

Boylen, C. W. and A. Kuliopulos. 1981. Further studies on the bathymetry of Lake George. in: C.W. Boylen (ed.). The Lake George Ecosystem, Volume I. The Lake George Association, Lake George, New York.

Brooke Benjamin, T. 1968. Gravity currents and related phenomena. Journal of Fluid Mechanics 31 (Part 2):209-248.

Carlson, R.E. 1977. A trophic state index for lakes. Limnology and Oceanography 22(2):361-369.

Clarke, F.W. 1924. The Data of Geochemistry. Fifth ed. Bull. U.S. Geol. Surv. 770. U.S. Government Printing Office, Washington, D.C. 841 pp.

Clesceri, N.L and S.L. Williams. 1972. Diatom Population Changes in Lake George, NY. Final Report: Office of Water Resources Research – Dept. of the Interior Contract 14-31-001-3387. 212 pp.

Colon, E.M. 1972. Hydrologic study of Lake George, New York. Ph.D. Thesis. Rensselaer Polytechnic Institute, Troy, New York. 81 pp. + Appendices.

Decker, D. 2011. Salt application estimates in the Lake George watershed. Personal communication. Executive Director, Lake George Watershed Coalition (http://www.lakegeorge2000.org/).

DelPrete, A. and R.A. Park. 1981. Postglacial history of Lake George based on diatoms in sediment cores. *in*: C.W. Boylen (ed.). The Lake George Ecosystem, Vol. I. Lake George Association, Lake George, NY.

Dillon, P.J. 1983. Nutrient budgets for Lake George, New York. in: C.D. Collins (ed.). The Lake George Ecosystem, Vol. III. Lake George Association, Lake George, NY.

Dixit, S.S. and J.P. Smol. 1994. Diatoms as indicators in the environmental monitoring and assessment program-surface waters (EMAP-SW). Environmental Monitoring and Assessment 31:275-307.

Eichler, L.W. and C.W. Boylen. 2011. The Extent of Oxygen Depletion in the Southern Basin of Lake George, New York. DFWI Technical Report 2011-12. Prepared for the FUND for Lake George. Darrin Fresh Water Institute, Bolton Landing, New York.

Eichler, L.W., R.T. Bombard, T.M. Burlingame and J.W. Sutherland. 2008. Cedar Lane Atmospheric Deposition Station – Town of Lake George, Warren County, New York. DFWI Technical Report 2008-03. Darrin Fresh Water Institute, Bolton Landing, New York.

Eiling, A. 1992. The phosphorus release from deep water sediments of an oligotrophic lake and its relation to the oxygen depletion in the overlying waters. Master of Science Thesis, Rensselaer Polytechnic Institute.

Environment Canada 2001. Priority Substances List Assessment Report - Road Salts. ISBN 0-662-31018-7.

Environment Canada and Health Canada 1999. Environmental Protection Act 1999. Priority Substances List Assessment Report – Road Salt.

Ferguson, F.A. 1968. A nonmyopic approach to the problem of excess algal growths. Environmental Science and Technology 2(3):188-193.

Ferris, J.J. and N.L. Clesceri. 1974. A description of the trophic status and nutrient loading for Lake George, New York – Preliminary Report. PO# 03J1PO1245, Office of Research and Monitoring, U.S. Environmental Protection Agency. 127 pp.

Ferris, J.J., N.L. Clesceri and S.I. Auerbach. 1974. Freshwater ecosystem research in water quality management. Environment Science and Technology 8(8):706-710.

Fischer, H.B., E.J. List, R.C.Y. Koh, J. Imberger and N.H. Brooks. 1979. Mixing in Inland and Coastal Waters. Academic Press, New York.

George, C.J. 1981. The fishes of the Lake George watershed. *in*: C.W. Boylen (ed.). The Lake George Ecosystem, Volume 1. Proceedings of the Lake George Research Symposium and Contributed Papers. Lake George Assn., Lake George, NY.

Godwin, K.S., S.D. Hafner and M.F. Buff. 2003. Long-term trends in sodium and chloride in the Mohawk River, New York: The effect of fifty years of road-salt application. Environmental Pollution 124:273-281.

Haines, D.A. and R. A. Bryson. 1961. An empirical study of wind factor in Lake Mendota. Limnology and Oceanography 6(3):356.

Hammond, A.L. 1971. Phosphate replacements: Problems with the washday miracle. Science 172:361-363.

Hannoun, I.A. and E.J. List. 1988. Turbulent mixing at a shear-free density interface. Journal of Fluid Mechanics 189:211.

Hannoun, I.A., K. Bowman-Kavanagh, E.J. List, W. Chiang, L. Ding, D. Karafa and I. Rackley. 2005. Use of ELCOM and CAEDYM for Water Quality Simulation in Boulder Basin, in the Proceedings of the Water Environment Federation Total Maximum Daily Load (WEF TMDL) 2005 Conference, Philadelphia, Pennsylvania.

Hecky, R.E. and P. Kilham. 1988. Nutrient limitation of phytoplankton in freshwater and marine environments: A review of recent evidence on the effects of enrichment. Limnology and Oceanography 33:796-822.

Hetling, L.J. 1974. Observations on the Rate of Phosphorus Input into Lake George and its Relationship to the Lake's Trophic State. Technical Report No. 36. NYS Dept of Environmental Conservation, Albany, NY 20 pp.

Hutchinson, G.E. 1957. A Treatise on Limnology, Vol. I Part 2. Chemistry of Lakes. John Wiley & Sons, New York, NY.

Hutchinson, D. R., W. M. Ferrebee, H. J. Knebel, R. J. Wold and Y. W. Isachsen. 1981. The sedimentary framework of the southern basin of Lake George, New York. Quaternary Research 15:44-61.

Jackson, R.B. and E.G. Jobbagy. 2005. From icy roads to salty streams. Proceedings of the National Academy of Sciences 102:14487-14488.

Jones, P.H., B.A. Jeffrey, P.K. Watler and H. Hutchon. 1992. Environmental impact of road salting. In D'Itri, F.M. (ed.) Chemical Deicers and the Environment. Lewis Publishing, MI. page 1-107.

Kelting, D.L. and C.L. Laxson. 2010. Review of Effects and Costs of Road De-icing with Recommendations for Winter Road Management in the Adirondack Park. Adirondack Watershed Institute Report #AWI2010-01.

Langen, T.A., M. Twiss, T. Young, K. Janoyan, J.C. Stager, J. Jr. Osso, H. Prutzman and B. Green. 2006. Environmental Impacts of Winter Road Management at the Cascade Lakes and Chapel Pond. Clarkson Center for the Environment Report #1. New York State Department of Transportation.

Langner, C.L. and P.F. Hendrix. 1982. Evaluation of a persulfate digestion method for particulate nitrogen and phosphorus. Water Research 16: 1451-1454.

Langmuir, I. 1938. Surface motion of water induced by wind. Science 87:119-123.

Lake George Association. 2012. Watershed maps and areal estimates for the major sub-watersheds in the Lake George basin. Compiled by Randy Rath, GIS Coordinator, Lake George Association, Lake George, NY.

Long, D.R., L.W. Eichler, R.F. Clifford and D. H. Pope. 1982. The Lake George monitoring program. *in*: M.H. Schadler (ed.). The Lake George Ecosystem, Vol. II. Lake George Association, Lake George, NY.

Long, D.R., L.W. Eichler, D. Roberts, R. Armstrong and D. H. Pope. 1982. Interaction of soluble silica and other major nutrients in Lake George. *in*: M.H.

Schadler (ed.). The Lake George Ecosystem, Vol. II. Lake George Association, Lake George, NY.

Madsen, J.D., J.W. Sutherland, J.A. Bloomfield, K.M. Roy, L.W. Eichler and C.W. Boylen. 1989. Lake George Aquatic Plant Survey Final Report. NYS Department of Environmental Conservation, Albany, NY. 350 pp.

Manley, T.O. 2006. Preliminary Hydrodynamic Investigation of Lake George's North Arm from June to October, 2004. Prepared for the Lake George Watershed Coalition, Lake George, NY. March 2006.

Mason, C.F., Norton, S.A., Fernandez, I.J. and L.E. Katz. 1999. Deconstruction of the chemical effects of road salt on stream water chemistry. Journal of Environmental Quality 28:82-91.

McClelland, I.M. and Y.W. Isachsen. 1986. Geological Synthesis of the Adirondack Mountains and their Tectonic Setting within the Southwestern Grenville Province, in Moore J., Davidson A, and Baer A (eds.) The Grenville Province: New Perspectives. Geological Association of Canada, Special Paper 31, p. 75-95.

Needham, J.G., C. Juday, E. Moore, C.K. Sibley and J.W. Titcomb. 1922. A Biological Survey of Lake George, New York. State of New York Conservation Commission. Albany, NY 78pp.

Newland, D.H. and H. Vaughan. 1942. Guide to the Geology of the Lake George Region: NY State Museum Handbook 19, 234 pp.

New York State Department of Environmental Conservation. 1985. Regulations, Chapter X, Part 659: Household Cleansing Products.

Nicholson, S.A., J.T. Scott and A.R. Breisch. 1979. Structure and succession in the tree stratum at Lake George, New York. Ecology 60:1240-1254.

Ogden, E.C., J.K. Dean, C.W. Boylen and R.B. Sheldon. 1976. Field Guide to the Aquatic Plants of Lake George, New York. Bull. No. 426. NYS Museum, Education Dept., Albany NY. 65 pp.

ReVelle, P. and C. ReVelle. 1988. The Environment: Issues and Choices for Society. 3rd ed. Jones and Bartlett Publishers. Boston. 749 pp.

Richardson, L.E. 1999. Physical and chemical environmental factors associated with the temporal and spatial distribution of cyanobacteria in Lake George, New York. MS Thesis. Rensselaer Polytechnic Institute, Troy, New York. 107 pp. + Appendix.

Rustad, L.E., J. S. Kahl, S..A. Norton and I.J. Fernandez. 1994. Multi-year estimates of dry deposition at the Bear Brook Watershed in Eastern Maine, USA. Journal of Hydrology 162:319-336.

Schindler, D.W., and E.J. Fee. 1974. Experimental lakes area: whole-lake experiments in eutrophication. Journal of the Fisheries Research Board of Canada 31:937-953.

Shuster, E. L. 1994. The hydrogeology of the Lake George drainage basin, southeastern Adirondack Mountains, New York. PhD Dissertation. Rensselaer Polytechnic Institute, Troy, New York. 246 pp. + Appendices.

Siegfried, C.A. 1982. Water Quality and Phytoplankton of Lake George, New York: Urban Storm Water Runoff and Water Quality Gradients. New York State Department of Environmental Conservation Technical Report No. 66, 64 pp.

Siegfried, C.A. and S.O. Quinn. 1986. Lake George Clean Lakes 314 Study (diagnostic/feasibility study), Final Report. Biological Survey, State Science Service, New York State Museum, Albany, New York, and Bureau of Technical Services, Division of Water, New York State Department of Environmental Conservation, Albany, New York, 430 pp.

Sklenar, K. S. 1996. Results of the Lake George 1995 Hydrodynamic Study, Bureau of Public Water Supply Protection, Division of Environmental Protection, Center for Environmental Health, NY Dept. of Health, 2 University Place, Albany, NY 12203-3313, 25 pp, 1996.

Sprengel, C. 1828. Von den Substanzen der Ackerkrume und des Untergrundes. J Tecnische Okonomisch Chemie 2: 423-474; 3:42-99, 313-352 and 397-421.

Standard Methods for the Examination of Water and Wastewater, 20th ed. 1998. APHA, AWWA, & WEF. Washington, DC.

Stearns and Wheler. 2001. Total Phosphorus Budget Analysis, Lake George Watershed, New York. Stearns & Wheler, Cazenovia, NY. Prepared for the Lake George Park Commission. October, 2001.

Stross, R.G. 1981. The *Nitella* meadow and the eutrophication of Lake George. in: C.W. Boylen (ed.). The Lake George Ecosystem, Vol. I. Lake George Association, Lake George, NY.

Stross, R.G. 1982. The significance of phosphorus on the eutrophication of Lake George. in: M.H. Schadler (ed.). The Lake George Ecosystem, Vol. II. Lake George Association, Lake George, NY.

Stross, R.G., J. Huvane and R.C. Sokol. 1988. Internal structure of deep-dwelling *Nitella* meadows. Aquatic Botany 29: 329-345.

Sutherland, J.W., J.A. Bloomfield and J.M. Swart. 1983. Final Report: Lake George Urban Runoff Study, National Urban Runoff Program. Bureau of Water Research, New York State Department of Environmental Conservation, Albany, New York. 84 pp. + Appendices.

Sutherland, J.W., J.A. Bloomfield, R.T. Bombard and T.A. West. 2001. Ambient Levels of Calcium and Chloride in the Streams and Stormsewers that Flow into Lake George (Warren County), New York. Division of Water, NYS Dept. of Environmental Conservation, Albany, NY.

Titler, R.V. and P. Curry. 2011. Chemical Analysis of Major Constituents and Trace Contaminants of Rock Salt. Pennsylvania Department of Environmental Protection. Bureau of Water Standards and Facility Regulation.

Truesdale, G.A., A.L. Downing and G.F. Lowden. 1955. The solubility of oxygen in pure water and sea-water. Journal of Applied Chemistry (London) 5:53-63.

US EPA. 1979. Methods for Chemical Analysis of Water and Wastes, EPA-600/4-79-020, Cincinnati, OH.

Vermont and New York State Departments of Environmental Conservation. 2012. Lake Champlain Long-term Water Quality and Biological Monitoring Program, Program Description. Vermont Department of Environmental Conservation, Water Quality Division, 100 South Main St., Waterbury, VT 05671-0408, and New York State Department of Environmental Conservation, Region 5, P. O. Box 296, Ray Brook, New York 12977-0296.

Wood, L.W. and G.W. Fuhs. 1979. An Evaluation of the Eutrophication Process in Lake George Based on Historical and 1978 Limnological Data. New York State Dept. of Health, Albany, NY. Environmental Health Report No. 5. 73 pp.

Wood, L.W. 1981. Gradients in transparency, chlorophyll and phosphorus. *in*: C.W. Boylen (ed.). The Lake George Ecosystem, Vol. I. Lake George Association, Lake George, NY.

Young, S.M. 2010. New York Rare Plant Status Lists June 2010. New York Natural Heritage Program, Latham, NY 116pp + Appendices.

APPENDIX I. HISTORICAL OVERVIEW OF LAKE GEORGE

"It was awe-inspiring when we rowed at the foot of the mountain and looked up, for it seemed as if the mountain hung right over our heads as we proceeded. The lake at the shore was very deep... The American elm grew in abundance. Chestnut, walnut, red juniper, oak, ... The mountains were everywhere overgrown with forests." – Peter Kalm, 1749

Lake George is one of New York State's most beloved scenic areas. It is famous for its spectacular scenery, its role in colonial military history and as a recreational resource for much of the northeastern United States. Since the late 1800s, there has been a renewed interest in the lake's natural history, the ecological changes brought about by human activity and the development of scientific investigations aimed at protecting the lake and its environs. This chapter, based largely on Boylen 1981a, reviews these studies and influences on the ecology and natural history of Lake George over the last 350 years.

In the summer of 1609, Samuel de Champlain sailed into Lake Champlain and claimed the region for France. Later that summer Henry Hudson, sailed up the Hudson River to present day Albany and claimed the watershed for the Dutch. The English laid claim to much of eastern North America in 1610 and the fight for European dominance of the region was underway. Named Lac du Saint-Sacrament in 1646 by the French and Lake George in honor of King George after the French and Indian War by the English, control of the lake became the key for political mastery of the region because of its strategic location between the Hudson and St. Lawrence watersheds.

For centuries the area around Lake George remained mostly a wilderness due to the hostility between the two major Indian cultures of the region - the Iroquois and the Algonquians. With European colonization contested between the French in Canada and the British in New England, the conquest of the colonial "northern frontier" centered on control of Lake George. Substantial settlement around the lake began only with the end of the American Revolution.

EARLY DESCRIPTIONS OF LAKE GEORGE

Several prominent naturalists visited Lake George prior to the 1850s. Their early historical observations provide a snapshot of the ecology of the lake and its watershed during early colonial settlement. Among the most notable were the observations of Peter Kalm, a Swedish botanist and one of the forerunners of modern ecology. His descriptive account of Lake George, in particular, is detailed and surprisingly accurate considering his journey by canoe was almost 250 years ago (Benson 1937). To make brief note of a few observations:

"We passed the nights in the midst of the forest, plagued with mosquitoes, gnats and wood-lice, and in fear of all kinds of snakes... An incredible quantity of gnats fill the woods. The gnats are very minute. They are ten times worse than the larger ones or mosquitoes for the size renders them next to imperceptible."

"Almost every night we heard trees crack and fall while we lay here in the woods, though the air was so calm that not a leaf stirred... It may be that wild pigeons settle in such quantities on one tree as to weigh it down. When the wind blows hard it is dangerous to sleep or walk in the woods... and even when it is very calm there is some danger in passing under very large and old trees."

PERMANENT SETTLEMENT AT LAKE GEORGE

The Lake George watershed was used only for seasonal foraging by Native Americans prior to European settlement in 1710. Trails led to it from all directions. Forest vegetation in the basin would occasionally be burned by the Iroquois for hunting and clearing (Day 1953). Only after the American Revolution were settlements made north along the shores of Lake George and to the west and northwest. In 1790, the population in present day Warren County was only 1,080.

European settlement accelerated exploitation of natural resources in the region. As noted by Kahn in 1750:

"All the old Swedes and Englishmen born in America whom I ever questioned asserted that there were not nearly so many edible birds at present as there used to be when they were children, and that their decrease was visible. About sixty or seventy years ago, a single person could kill eighty ducks in a morning; but at present you frequently waited in vain for a single one. The wild turkeys, partridges and hazelhens, were [once] seen in large flocks in the woods."

"The cause of this diminution is not difficult to find. Before the arrival of the Europeans, the country was uncultivated and full of great forests. Now the woods are cut down. The people, increasing in this country, have by hunting and shooting in part extirpated the birds, in part frightened them away. In spring the people still steal eggs, mothers and young indifferently, because no regulations are made to the contrary. And if any had been made, the spirit of freedom which prevails in the country would not suffer them to be obeyed."

"Aged people had experienced with the fish the same conditions which I have just mentioned in regard to birds. In their youth, the bays, rivers and brooks, had such quantities of fish that at one draught in the morning they caught as many as a horse was able to carry home. But at present things are greatly altered, and they often work in vain all night long with their fishing tackle. The causes of this decrease of fish are partly the same as those of the diminution in the number of birds. They are of late caught by a greater variety of contrivances, and in different manners than before. Many old people said that the difference in the quantity of fish in their youth in comparison with that of today was as great as between day and night."

At Lake George in 1848 Charles Lanham wrote:

"The days of trout-fishing in Lake Horicon are nearly at an end. A few years ago it bounded in salmon-trout, which were frequently caught weighing twenty pounds, but their average weight at the present is not more than a pound and a half, and they are scarce even at that. The cause of the great decrease in the large trout of this lake is this - in the autumn, when they have sought the shores for the purpose of spawning, the neighboring barbarians have been accustomed to spear them by torch-light; and if the heartless business does not soon cease, the result will be, that in a few years they will be extinct."

Certainly wildlife was a major staple in the diets of the settlers. Timothy Dwight observed in 1802 that a hunter's take of deer at Lake George would amount to 20-30 per year (Dwight 1822). Without moderation, regulation and concern, fish and wildlife populations declined dramatically. Initially forests were cut to provide space as well as wood for houses and barns. Subsistence crops were raised on cleared land. Pastures and shrub land grazed by cattle and sheep often occupied a much larger area than croplands and consumed vegetation on some of the steeper shoreline such as the slopes of Tongue Mountain. Most of the residents living off the lake in the 1800s and early 1900s maintained privies, livestock yards and pigpens that discharged nutrients to the lake.

THE IMPACT OF INDUSTRY ON LAKE GEORGE

Prior to permanent settlement the Lake George watershed supported an extensive fur trade that reduced populations of exploited wildlife substantially. By the early 1800s, logging, fishing and mining were well established commercial enterprises throughout the basin. Initially hemlock was selectively cut to provide bark for tanneries. Later white pine and spruce and a variety of hardwoods were logged extensively for lumber. Still later pulp wood was cut from the slopes around the lake to supply a growing paper industry first at Glens Falls and then at Ticonderoga. Through the early 1800s lumbering was the main commercial activity. Ultimately the Adirondack forests including those of Lake George were exhausted and often reduced to bare eroded areas. Mining activity - iron, titanium and graphite - occurred in the Lake George basin, but probably had relatively minor impact upon the ecology of the lake. A commercial fishery was located at Lake George Village by the beginning of the 20th century.

Tourism began soon after early European settlement. The natural beauty of the region attracted visitors by stagecoach traveling mainly by the military trails. Hospitality was provided by the numerous inns and boarding houses. It has been said the natives of this region lived upon fish and strangers (Van de Water 1946). The influx of vacationers increased with the coming of the railroad to Glens Falls in 1869 and to Lake George in 1882, and tourism and recreation continued as major industries throughout the 20th century. These activities likely had significant effects on the lake. Even into the early part of the 20th century, the steamboats routinely dumped ash, sewage and garbage overboard.

ENVIRONMENTAL AWARENESS AND PRESERVATION

With the decline of the lumber and mining industries in the Adirondacks in the mid- to late-1800s, the State of New York began to make major land purchases which included much of the watershed land around Lake George and many of the islands. According to Section 8 of the Adirondack Forest Preserve in 1885, "The lands now or hereafter constituting the forest preserve shall be forever kept as wild forest lands. They shall not be sold, nor shall they be leased or taken by any corporation, public or private." Concurrently the Lake George Association (LGA) representing property owners was organized in 1885 and has lobbied for state and local legislation to promote conservation of the lake and its watershed (http://www.lakegeorgeassociation.org/).

THE STATE BIOLOGICAL SURVEYS

During the early 20th century, increased fishing pressure continued to reduce fish populations. Largely in response to these reductions, 13 species of fish, including at least 8 non-indigenous species, were stocked in the lake (George 1981). In 1920 the State Legislature authorized the New York Conservation Commission to conduct a biological survey of Lake George to determine the most practical method of increasing fish production. The survey was initially led by James Needham at

Cornell University, assisted by Chancey Juday, a pioneering limnologist from the University of Wisconsin. They conducted a comprehensive biological survey that was one of the first applications of a whole ecosystem approach to freshwater studies (Needham et al. 1922). This investigation led to the establishment of the State Conservation Fund in 1926 and to a biological survey of the Lake Champlain watershed in1929.

DISCOVERY OF LANGMUIR CIRCULATION

Irving Langmuir's discovery of the water circulation mechanism that now bears his name while at his summer home on Crown Island, arose from his observations of air bubbles on the water forming windrows parallel to the direction of the wind. Langmuir made thousands of discrete measurements using oriented umbrellas and lamp-bulb floats from which he was able to show that the windrows were the result of wind-induced circulation of the surface water perpendicular to the wind direction (Langmuir 1938). This discovery was a major contribution to physical limnology and oceanography.

INTRODUCTIONS OF INVASIVE SPECIES

In addition to the deliberate introductions of fish species, a number of invasive pest species have been spread into Lake George, and have the potential to dramatically alter the ecological functioning of the lake. Five species of particular concern today include curly-leaf pondweed and Eurasian watermilfoil, both rooted aquatic plant species; spiny waterflea, a zooplankton species; and zebra mussels and asian clams, both mollusks. These species are the focus of campaigns to control and if possible eradicate them from Lake George.

CURRENT DIRECTIONS OF SCIENTIFIC RESEARCH AT LAKE GEORGE

The New York State (NYS) Department of Environmental Conservation (DEC) and the NYS Bureau of Fisheries have maintained a continued research interest in Lake George since the 1920s. Their efforts were integral to the State Legislature's passage in 1945 of the Lake George Law to maintain the Class AA Special water quality standard for the lake (NYS Legislative Document 1945). In 1947, a NYS Supreme Court decision gave control of lake levels to the State. In the 1950s, studies supported by DEC at Lake George provided the first scientific data showing the accumulation of DDT in fish and its adverse effect on fish reproduction.

In 1967 the Lake George Water Research Center was organized at Rensselaer Polytechnic Institute. Now known as the Darrin Fresh Water Institute (DFWI), it originally provided a field station on the northeast side of the lake not only for investigators directly associated with RPI but also for those from other regional colleges and universities, numbering some six to eight organizations. During the late 1960s and early 1970s, the U.S. International Biological Program (IBP) funded by the National Science Foundation identified Lake George as a control site within the Eastern Deciduous Forest Biome (EDFB). Lake George was the "clean" lake counterpart to Lake Wingra in Wisconsin, heavily polluted from decades of nutrient runoff. The early focus of the DFWI was to generate a comprehensive study of the lake, both in terms of its chemistry and biota, as well as the land processes that play a role in water quality such as stormwater, septic and sewer inputs.

Under the IBP mandate, the DFWI investigated biological and chemical relationships at the land-water interface. Five deep water stations were selected through the midrange of the lake representing two stations in both the South and North basins and one in the Narrows. Nutrient and other analyte concentrations were profiled through the water column seasonally through 1978. Numerous research projects on the bacterioplankton, phytoplankton, zooplankton, fish and aquatic vegetation were conducted. By 1980 the body of research on the lake prompted the

LGA, under the leadership and assistance of Mary-Arthur Beebe and Dr. Margaret Schadler, to organize a symposium for researchers, regional municipal representatives, state agencies and members of the community to share information on the lake. The meeting culminated in the publication of the first volume of the Lake George Ecosystem in 1981 (Boylen 1981b). The success led to two additional symposia organized by the LGA in 1982 (Schadler 1982) and 1983 (Collins 1983).

In 1977, The FUND for Lake George was formed within the LGA to assure support for water quality preservation and for prevention of further pollution of Lake George (http://www.fundforlakegeorge.org). In 1980, with financial assistance from the FUND, the DFWI initiated water chemistry monitoring based on programs established under the IBP program to serve as a barometer of the “health” of the lake.

REFERENCES

Benson, A.B., ed. 1937. Peter Kalm's Travels in North America. Wilson-Erickson, Inc., New York.

Boylen, C.W. 1981a. Four centuries of human influence upon the ecology of the Lake George basin. *in*: C.W. Boylen (ed.).The Lake George Ecosystem, Vol. I. The Lake George Association, Lake George, New York.

Boylen, C.W., ed. 1981b. The Lake George Ecosystem. The Lake George Association, Lake George, New York.

Collins, C.D., ed. 1983. The Lake George Ecosystem. Volume III. The Lake George Association, Lake George, New York.

Day, G.M. 1953. The Indian as an ecological factor in the northeastern forests. Ecology 34: 329-346.

Dwight, T. 1822. Travels in New England and New York. Vol. 111, The Belnap Press of Harvard University Press, Cambridge, MA.

Langmuir, I. 1938. Surface motion of water induced by wind. Science 87: 119-123.

Lanham, C. 1858. Adventures of an Angler in Canada, Nova Scotia and the United States. R. Bentley, London.

Needham, J.G., C. Juday, E. Moore, C.K. Sibley and J.W. Titcomb. 1922. A Biological Survey of Lake George, New York. State of New York Conservation Commission. Albany, NY 78pp.

New York State Joint Legislative Committee on Lake George Water Conditions. 1945. Report, Lake George. Legislative Document (1945) No. 67.

Schadler, M.H., ed. 1982. The Lake George Ecosystem. Volume II. The Lake George Association, Lake George, New York.

Van de Water, F.F. 1946. Lake Champlain and Lake George, The Bobbs-Merrill Co., Indianapolis.

APPENDIX II. TIME-SERIES ANALYSIS

The primary objective of the statistical analysis of measurements through time was to determine the existence and magnitude of trends. Simple regression can be useful to characterize overall trend in a time series. However, because the random errors around the regression line may be autocorrelated, statistics for the regression parameters will often be inappropriate or misleading when applied to a time series.

One way to defensibly estimate trend with appropriate precision measures in a time series is based on Box-Jenkins time-series modeling as follows. Suppose that the series can be written as $z_1, z_2, \ldots z_T$. Also, let the notation $e_1, e_2, \ldots$ up to e_T describe a series of independent normally distributed random numbers with a mean of 0 and a standard deviation of σ. This latter series will be assumed to make up the random component of the z_T series. The Box-Jenkins approach consists of breaking the original time series, z_T, down into a few estimated parameters together with a residual series of normally distributed random numbers. This residual series will then be the estimate of the e_T series (*e.g.*, Wei 1990).

Data series may require transformation so that the mean and variance are the same, or at least similar, throughout the transformed series. This may involve variance-stabilizing transformations, such as the logarithm or a power transformation. The transformation used in the Box-Jenkins analysis is to take differences. Given an initial series of data $z_1, z_2, \ldots z_T$, the transformed, differenced, series is given by $y_1 = z_2 - z_1$, $y_2 = z_3 - z_2, \ldots$ $y_{T-1} = z_T - z_{T-1}$. This transformation could also involve taking a second (difference of the y_t series) or third difference until the final series appears to have the same mean (no trend) throughout the series.

Importantly, if there was a linear trend in the series before the last differencing, then that trend will be reflected as a non-zero constant that the new y_t series will fluctuate around. That is, a simple series with only a trend and a random component can be written $z_{i+1} - z_i = c + e_t$, and c will represent the trend (*i.e.*, the series increases or decreases on average by c at each time step, depending on whether c is positive or negative).

The transformed series are next evaluated for autocorrelation, that is, the measure of the series' correlation with itself as a function of temporal differences. This involves comparison of z_i with z_{i+1} and z_{i+2} and so on up to z_{i+l}, for some value of l and then calculation of the correlations for successive lags denoted by l (*e.g.* Fig. A-II-1, lower left). These values of 1, 2, up to l are called *lags*.

A trend will induce a specific kind of autocorrelation, as each value in the series would tend to differ by a specific amount, and the magnitude of the correlation would get progressively smaller at larger lags (*e.g.* Fig. A-II-1). This trend would induce a specific kind of dependence among all of the values in the series, and that dependence would produce a specific kind of pattern in the autocorrelation display. For present purposes, discussion of the partial autocorrelation function will be omitted, apart from noting that this is an essential tool for identifying a Box-Jenkins model.

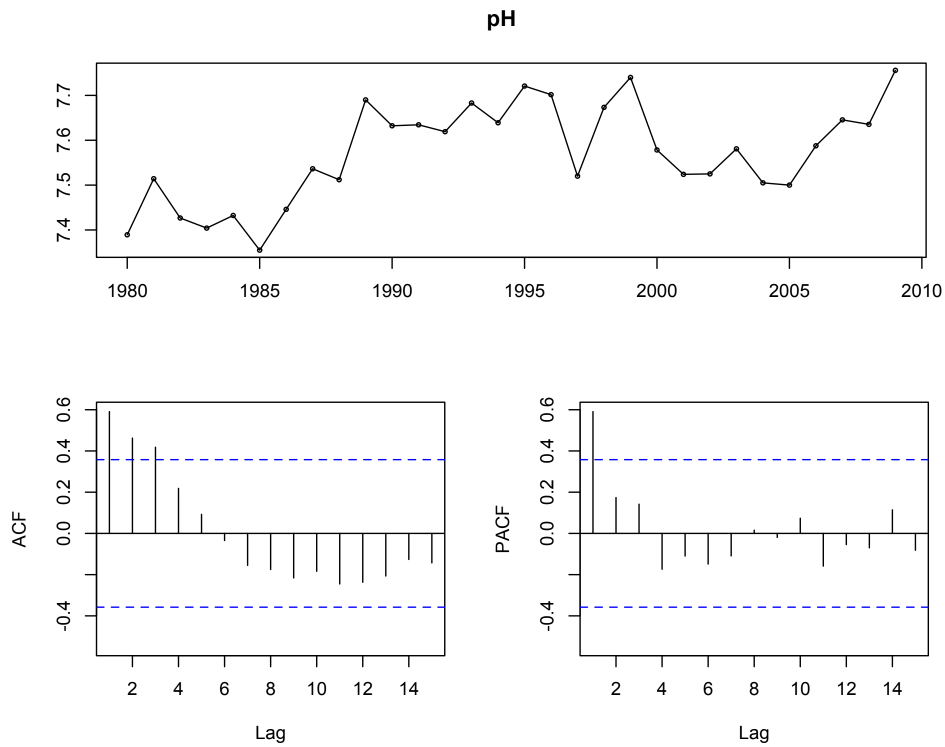

FIGURE A-II-1. *A time-series display of a series of annual mean acidity in Lake George (represented as pH) with a trend (top) together with the autocorrelation function (bottom left) and the partial autocorrelation function (bottom right). Note the general upward trend in the original series, which has induced a slowly decaying autocorrelation (lower left). Values of the series have a correlation of almost 0.6 with values one time step (lag) away, but that correlation decays to approximately 0 six time steps away before becoming slightly negative.*

After differencing the series shown in Figure A-II-1, the resulting differenced series appears to have a similar mean and variance throughout the series, and there are no significant autocorrelations or partial autocorrelations remaining at any lag (Fig. A-II-2). If the series consists of purely random normally distributed values, the series can be written $y_t = e_t$, and the underlying correlation is zero at each lag, as each value of e_i is assumed independent of e_j for any i and j. The model-fitting tactic is to continue to difference or add parameters until the residual series appears to be purely random normally distributed noise, with an autocorrelation and partial autocorrelation indistinguishable from zero.

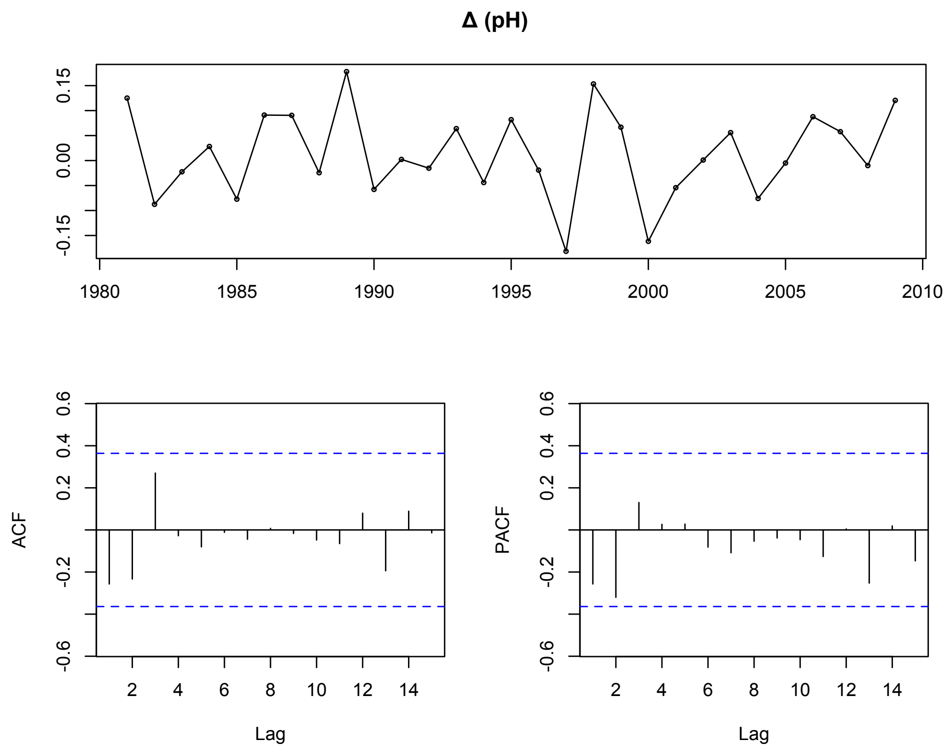

FIGURE A-II-2. *A time-series display of the differenced series from Figure A-II-1 (top) showing no remaining trend, together with the autocorrelation function (bottom left) and the partial autocorrelation function (bottom right). Note the dotted horizontal lines in the autocorrelation plots show the limits of statistical significance for the correlations.*

It is possible that the values of z_t will be autocorrelated in a more complicated way that could be explained by an *autoregressive process*. Suppose the last two values of the series affect each new value of the series. An autoregressive model with two autoregressive parameters and with no trend could be written as $z_t = a_1 z_{t-1} + a_2 z_{t-2} + e_t$, for parameters a_1 and a_2, $-1 < a_1, a_2 < 1$. In this case, values of the series are influenced by the past in a very specific way other than trend.

After accounting for the autoregressive parameters, there may still be a *moving average component* to the model, which will allow past random events to persist and affect the present. Then a series without trend but with autoregressive and moving average components could be written as $z_t = c + a_1 z_{t-1} + e_t + b_1 e_{t-1}$: c would represent some offset from zero, and a_1 would be the autoregressive parameter and b_1 would be the moving average parameter (between -1 and 1).

Box-Jenkins models are usually described with the notation (p,d,q), with p denoting the number of autoregressive parameters, d denoting the number of differences, and q denoting the number of moving average parameters. For example, (2,1,0) would indicate a model with two autoregressive parameters, one difference, and no moving average parameters.

The estimated trend together with the time series plots and correlation plots are used for comparing the time series. If two series are fundamentally describing the same basic process, and if that process is clearly reflected in the data, both series should fit similar models and estimated model parameters would have similar magnitudes. However, exactly the same fundamental process could result in different models for a variety of reasons. Consequently graphical summaries and summary statistics to compare means, medians, quartiles and extreme values for stations are also presented with the results of the time series analyses performed.

REFERENCES

Wei, W. W. 1990. Time Series Analysis; Univariate and Multivariate Methods. Addison-Wesley Publishing Co. Redwood City, California. 478pp.