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Environmental change in Lake Champlain revealed by long-term monitoring

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ABSTRACT

Long-term monitoring data on Lake Champlain spanning the past two to five decades were analyzed to document water quality and biological changes in the lake. August mean surface water temperatures increased during 1964–2009 in most Lake Champlain regions at rates (0.035–0.085 °C/year) similar to what has been observed in the Laurentian Great Lakes and elsewhere. Secchi disk transparency increased by over a meter during 1964–2009 in regions along the main stem of the lake, with much of the increase occurring after the 1993 zebra mussel invasion. Transparency declined in northeastern regions where zebra mussel densities were lower. No trends in hypolimnetic dissolved oxygen concentrations or depletion rates were found in any of the deep lake regions during 1990–2009. Sodium concentrations tripled in the Main Lake region since the 1960s. Chloride increased in the Main Lake by 30% since 1992, but declined in northeastern regions of the lake during recent years, coincident with reductions in road salt use in Vermont. Total phosphorus concentrations decreased during 1979–2009 in southern and northwestern lake regions, but increased by 72% in Missisquoi Bay where chlorophyll-a concentrations doubled over the period. There was a general lakewide trend of decreasing total nitrogen levels during 1992–2009 that may have been due in part to reductions in atmospheric nitrogen loading to the watershed. Cyanobacteria increased their dominance within the phytoplankton community in northeastern regions of the lake since the 1970s.

Introduction

Awareness of environmental change, and an understanding of the response of ecosystems to air and water pollution and land-use changes, are essential to designing appropriate management interventions (Lovett et al., 2007; Watzin, 2007). Because the rate of most ecological changes is very slow, usually occurring over decades to centuries, long-term environmental monitoring is essential for detecting trends in ecological variables.

Lakes are often the subject of long-term monitoring because representative samples can be readily obtained that integrate the influence of watershed and atmospheric disturbances (Schindler, 2009). Important knowledge has been gained from long-term monitoring of large lakes, including insights about lake ecosystem response to nutrient loadings, invasions by nonnative species, and climate change (Eimers et al., 2005; Rockwell et al., 2005; Jankowski et al., 2006; Dobiesz and Lester, 2009; Fahnenstiel et al., 2010; Mida et al., 2010). In some cases, long-term lake monitoring data were used for purposes that were unforeseeable at the time the monitoring program was initiated (Hampton et al., 2008).

Lake Champlain is one of the largest lakes in North America, with a 1127 km^2 surface area, a mean depth of 22 m, and a 21,326 km^2 drainage basin that are shared by the States of Vermont and New York and the Province of Quebec (Cohn et al., 2007). The lake has a complex morphology with numerous shallow bays and arms that are partially isolated from the deep main stem of the lake by natural land forms or causeways. As a result, a wide variety of limnological conditions exists in Lake Champlain with respect to phosphorus loadings and trophic state (Medalie and Smeltzer, 2004), ionic composition (Potash et al., 1969), thermal and hydrodynamic features (Manley et al., 1999), optical properties (Effler et al., 1991), and plankton communities (Shambaugh et al., 1999).

Like many large lakes worldwide, Lake Champlain faces a number of environmental stressors. Global climate change, land use changes, agricultural and industrial contaminants in water runoff, and increased opportunities for transport of exotic species all have the potential to substantially alter lake ecosystems. A substantial proportion of the Lake Champlain drainage was deforested in the 1800s and converted to farmland, leading to increased erosion and anthropogenic inputs of fertilizers. In comparison with the Great Lakes, the Lake Champlain Basin has a relatively low human population density and few major industrial discharges. Similarly, the lake does not receive substantial shipping traffic, which is a major vector of exotic species introductions in the Great Lakes and elsewhere. However, there are several stressors affecting the Lake Champlain Basin that would be expected to produce environmental changes within the lake.

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Water temperature increases in large lakes have provided evidence of a warming global climate (Dobiesz and Lester, 2009; Austin and Colman, 2007; Hampton et al., 2008). Summer air temperatures have increased in the Lake Champlain region over the past several decades (Stager and Thill, 2010, Fig. 1a), and we would expect summer lake water temperatures to have increased in Lake Champlain as a result.

Increasing chloride concentrations have been found in lakes (Chapra et al., 2009; Novotny and Stefan, 2009), rivers (Robinson et al., 2003; Kauschal et al., 2005), and groundwater (Mullaney et al., 2009; Eyles and Meriano, 2010) across the northern hemisphere, particularly as a result of the application of deicing salts for winter road maintenance (Chapra et al., 2009; Daley et al., 2009; Trowbridge et al., 2010). Road salt application rates in Vermont state highway districts within the Lake Champlain Basin increased during the 1990s but then declined in more recent years (Fig. 1b), either as part of a conscious management effort or as a result of less severe winter driving weather. With urban land uses representing 8% of the watershed and increasing over time (Troy et al., 2007), we would expect to find changes in sodium and chloride concentrations in Lake Champlain linked to road salt usage.

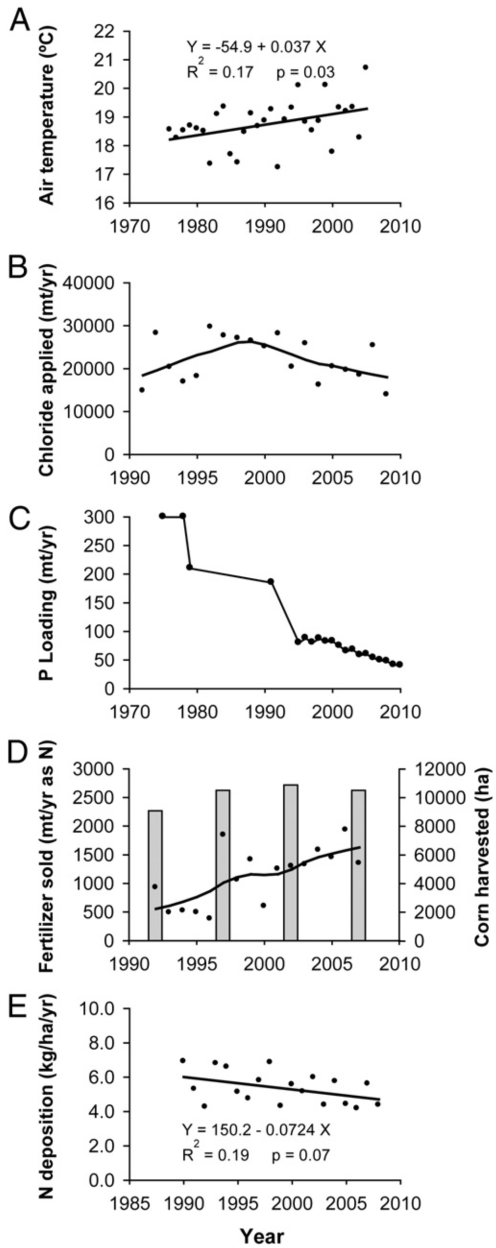

Fig. 1. Trends in Lake Champlain environmental stressors. A. Summer mean air temperature in the Lake Champlain Basin, 1976–2005 (modified from Stager and Thill, 2010). B. Winter road salt application rates (metric tons per year as chloride) for Vermont state highway maintenance districts within the Lake Champlain Basin, 1990–2009 (Vermont Agency of Transportation data). C. Total phosphorus loading to Lake Champlain from Vermont and New York wastewater treatment facilities, 1975–2010 (Smeltzer et al., 2009; Bogdan, 1978). D. Nitrogen fertilizer sold (dots and LOWESS trend line) and area of corn harvested (vertical bars) in Franklin County, VT, 1992–2007. Fertilizer sales data are from the Vermont Agency of Agriculture, Food, and Markets and do not include manure. Corn data are from the U.S. Department of Agriculture Census of Agriculture. E. Annual mean atmospheric wet deposition rates of inorganic nitrogen at Underhill, VT, 1990–2008 (National Atmospheric Deposition Program data).

Lake Champlain receives phosphorus loadings from multiple point and nonpoint sources in excess of its assimilative capacity (Smeltzer and Quinn, 1996). Control of eutrophication in Lake Champlain through phosphorus reduction has been a priority for resource management agencies since the 1970s. Phosphorus detergent laws were in place basinwide by 1978. These laws, and requirements for phosphorus removal from wastewater effluent at large treatment facilities, have reduced wastewater phosphorus loads to Lake Champlain by 86% since the 1970s (Fig. 1c). Efforts to reduce nonpoint source phosphorus loading to the lake accelerated in recent years with over $120 million being committed since 2004 to support enhanced stormwater management, implementation of agricultural best management practices through regulatory and incentive-based programs, and river corridor protection measures (Vermont Agency of Natural Resources and Vermont Agency of Agriculture, Food, and Markets, 2010). Consequently, reductions in lake variables associated with eutrophication, such as phosphorus, chlorophyll-a, and cyanobacteria concentrations, and increases in water clarity and hypolimnetic dissolved oxygen, would be expected over this time period.

Nitrogen loading to lakes can be influenced by factors including crop production on agricultural land and atmospheric deposition (Elser et al., 2009). Nitrogen fertilizer sales and the amount of corn land harvested within the most heavily agricultural sub-watersheds within the Lake Champlain Basin increased since 1990 (Fig. 1d). However, there has been a 19% decrease in atmospheric deposition of total (wet + dry) nitrogen in the eastern U.S. during 1990–2008 (MACTEC, 2010), and a marginally significant decline of similar magnitude in the wet deposition rate of inorganic nitrogen at a monitoring station located within the Lake Champlain Basin (Fig. 1e). The net effect of these and other factors on nitrogen concentrations in Lake Champlain is difficult to predict.

Hydrologic connections between Lake Champlain, the Hudson River, and the Great Lakes via the Champlain Canal and the Richelieu River, and other vectors, have created pathways for invasion of Lake Champlain by 48 exotic species (Marsden and Hauser, 2009). Of these species, zebra mussels, in particular, can have profound effects on temperate lake ecosystems as a consequence of filtration activity, resulting in significant water-column decreases in suspended solids, phosphorus, and chlorophyll, with corresponding increases in water clarity and alterations in the phytoplankton and benthic communities (Barbiero and Tuchman, 2004; Raikow et al., 2004; Higgins and Vander Zanden, 2010). Based on experiences in other lakes, increases in Secchi disk transparency and proliferation of cyanobacteria species such as *Microcystis aeruginosa* could be expected in Lake Champlain since the arrival of the mussels in 1993.

Environmental changes that result from anthropogenic activities tend to initially be small, and masked by naturally high inter-annual variability. In order to detect and monitor lake-wide changes, and be able to evaluate efficacy of management efforts to remediate environmental damage, collection and examination of long-term data are critically needed. Lake Champlain water quality managers and researchers had the foresight, decades ago, to establish long-term monitoring programs to detect changes in water quality that may result from human activities, and that could affect ecological processes and human uses of the lake. Long-term records are available

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for lake variables including temperature, water transparency, hypolimnetic dissolved oxygen, inorganic ions, nutrients, chlorophyll-a, larval zebra mussel densities, and phytoplankton community composition. Our objective in this paper is to integrate data from three such monitoring programs in order to assess the extent to which the expected water quality and biological changes in Lake Champlain have occurred over the past several decades.

Methods

Data sources

Data for this analysis were obtained from three monitoring programs including early limnological surveys on Lake Champlain by University of Vermont limnologists E.B. Henson and M. Potash (H–P), citizen monitoring supported by the Vermont Lay Monitoring Program (LMP), and a Long-Term Water Quality and Biological Monitoring Program on Lake Champlain (LTMP) supported by the Lake Champlain Basin Program. The H–P surveys were conducted during 1964–1974 (Henson and Potash, 1966; Potash and Henson, 1975) and the data from these surveys were later compiled electronically and documented by Henson and Potash (1987). The LMP began in 1979 and is supported by the Vermont Department of Environmental Conservation (DEC). The citizen volunteers are trained by professional staff and adhere to approved procedures that ensure data quality (Picotte and Pomeroy, 2000; Canfield et al., 2002). The LTMP was initiated in 1992 and is operated by state agency staff (Vermont DEC and New York State DEC, 2010).

The monitoring variables selected for this analysis included those sampled by at least two of these programs consistently over a multiple year period, as well as additional measures of interest available only from the LTMP dataset (Table 1). The H–P surveys included data on Secchi disk transparency (SDT), sodium ion (Na^{+}), calcium ion (Ca^{++}), and water temperature. The LMP and the LTMP datasets included SDT, total phosphorus (TP), and chlorophyll-a (Chl-a) results. Additional measurements included only in the LTMP dataset were total nitrogen (TN), chloride ion (Cl^{-}), hypolimnetic dissolved oxygen (DO), net phytoplankton cell densities and biovolume, and zebra mussel (*Dreissena polymorpha*) larval densities. Other variables measured by these monitoring programs but not included in this analysis are also listed in Table 1. All of the LTMP data, including those summarized in this paper, are available online at www.anr.state.vt.us/dec/waterq/lakes/htm/lp_longterm.htm.

Sampling methods and locations

The three monitoring programs differed with respect to sampling season, frequency of sampling, and sample depths (Table 1), although there was broad overlap in the sampling seasons and all programs obtained samples from the upper mixed layer of the water column in offshore locations. Data obtained during the winter months (December–March) by the H–P surveys were excluded from the analysis for better comparability with the results from the LTMP and LMP programs which operated during the growing season only.

Sampling locations were selected for this analysis to include ten sites distributed throughout the lake that were common to all three programs and where the sampling effort was sustained across the years (Fig. 2). Sampling stations for the LTMP were precisely located in the field using LORAN or GPS receivers. Each LMP station listed in Fig. 2 was co-located with a corresponding LTMP station, but the volunteer monitors generally used visual landmarks to find their stations. Sampling locations for the H–P survey were not precisely recorded, as would be possible with modern electronic navigation aids. Instead, Henson and Potash (1987) divided the lake into 69 lake areas and identified each of their sampling location as being within one of those lake areas. H–P data from the lake areas corresponding to the station locations shown in Fig. 2 were used in this analysis.

Table 1
Sampling methods for long-term monitoring programs on Lake Champlain, including the Henson and Potash surveys (H–P), the Vermont Lay Monitoring Program (LMP), and the Long-Term Water Quality and Biological Monitoring Program on Lake Champlain (LTMP). Analytical methods are documented in Vermont DEC and New York State DEC (2010).

	Monitoring program		
	H–P	LMP	LTMP
Period of record	1964–1974	1979–2009	1992–2009
Sampling season	April–Nov[a]	May–Sept	April–Nov
Sampling frequency	Variable	Weekly	Bi-weekly
Total number of monitoring sites	69	39	15
Monitored variables used in this analysis	SDT, Na^{+}[b], Ca^{++}[b], temperature	SDT, TP[c], Chl-a[c]	SDT, TP[d], Chl-a[c], TN[d], Cl[d], Na^{++}[d], Ca^{++}[d], DO[e], temperature[e], net phytoplankton[f], zebra mussel veligers[f]
Additional variables available in the dataset	pH, alkalinity, conductivity, manganese, potassium, DO		Dissolved phosphorus[g], soluble reactive phosphorus, dissolved reactive silica[g], total Kjeldahl nitrogen, total nitrate–nitrite nitrogen, total ammonia nitrogen, alkalinity[g], conductivity[g], manganese[g], potassium[g], total iron, total lead, total organic carbon, dissolved organic carbon, total inorganic carbon, total suspended solids, net zooplankton[g]

[a] Winter (December–March) data were removed from the data set prior to analysis.
[b] Surface grab samples.
[c] Vertically-integrated hose samples to twice the Secchi depth.
[d] Upper mixed layer discrete-depth composites.
[e] Vertical water column discrete-depth profiles.
[f] Vertical 63 μm net tows.
[g] Sampling of these additional variables is on-going.

Chemical and physical analytical methods

All chemical analyses for the LMP and LTMP programs were conducted by state environmental laboratories in Vermont or New York using standard methods under Quality Assurance Project Plans approved by the U.S. Environmental Protection Agency (Vermont DEC and New York State DEC, 2010). Methods used for the H–P surveys were comparable, though not identical, to these methods. Lake surface temperature was measured during the H–P surveys using a calibrated bucket thermometer (Henson and Potash, 1987), while the LTMP employed calibrated thermistor probes. Ca^{++} and Na^{+} were analyzed during the H–P surveys by atomic absorption spectrophotometry (Potash and Henson, 1975), whereas the LTMP used varying methods for these elements during the monitoring period including inductively coupled plasma (ICP) atomic emission spectrometry (1992–2001), atomic absorption (2002), and ICP mass spectroscopy (2003–2005).

Temperature

Water temperatures were measured in situ throughout the water column at each LTMP lake station using thermistors on cables or multiprobe devices. However, comparable depth profile data were not obtained during the H–P surveys, and temperature data were consistently available only for the summer months. The analysis of temperature trends was therefore limited to surface measurements recorded during the month of August, which was the month typically

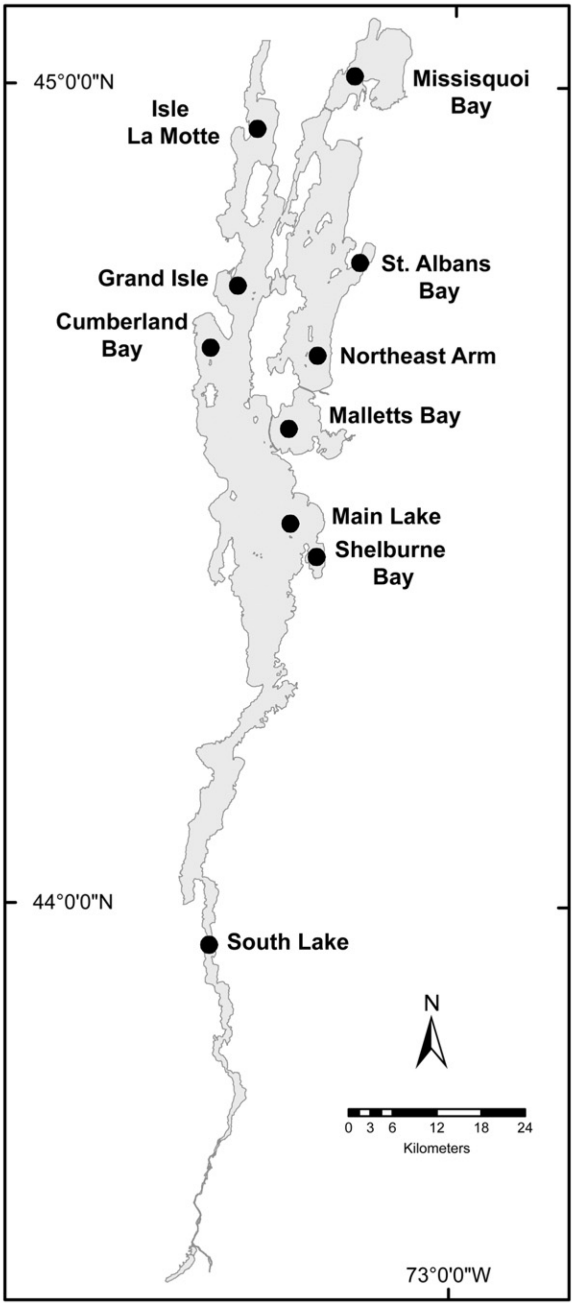

Fig. 2. Location of sampling stations in Lake Champlain. Stations sampled by the H–P surveys were not precisely located, but corresponded to the general lake region indicated.

having the largest number of measurements. All temperatures recorded at 1 m depth during August (N = 1–5/year) were averaged by year to calculate an August mean surface temperature in each lake region for years where data were available.

Hypolimnetic dissolved oxygen

Hypolimnetic DO was measured by the LTMP using both Winkler titration and in situ electrode methods (Vermont DEC and New York State DEC, 2010). However, several different instruments were employed across the years for the electrode measurements and the Winkler method provided more consistently calibrated data over the entire monitoring period. Therefore, only the Winkler titration results were used for long-term trends analysis.

Conventional measures of hypolimnetic hypoxia such as the areal or volumetric hypolimnetic oxygen depletion rate (Burns et al., 2005; Matthews and Effler, 2006) were difficult to apply to Lake Champlain because the complex morphometry and sometimes indistinct thermocline created uncertainty about the spatial extent of the hypolimnion at some sampling stations. Trends in hypolimnetic hypoxia were assessed instead using measurements of late-summer DO concentrations recorded by the LTMP in the near-bottom waters of three deep lake regions, including the Main Lake (90 m), Malletts Bay (25 m), and the Northeast Arm (45 m). In order to standardize the comparison of late-summer DO conditions across years, DO concentrations were interpolated between sampling dates to provide an estimate of the DO concentration at these depths on September 1 of each year. Additionally, summer-long hypolimnetic DO depletion rates were calculated from the differences in bottom-water DO concentrations between June 1 and September 1 each year. The depth locations of the hypolimnetic DO samples used for this analysis were the same across all years within each lake region. DO data obtained during 1990–1991 by a preceding study using comparable methods (Vermont DEC and New York State DEC, 1997) were used to supplement the LTMP dataset for this analysis.

Zebra mussel veligers

Zebra mussel adults were first discovered in the South Lake region of Lake Champlain in 1993, and their planktonic larvae (veligers) were monitored by the LTMP starting in 1994 to provide an indirect measure of population densities as the mussels spread to other regions of the lake. Zebra mussel veligers were sampled by vertical net tows concurrently with the water quality monitoring efforts (Stangel and Shambaugh, 2005). Tow depths varied between 3 and 10 m depending on the depth of the sampling station. Enumeration procedures followed Marsden (1992). The seasonal timing of veliger production varied from site to site and year to year. In order to provide a standardized basis for comparison, veliger densities at each station were reported as a time-weighted season mean calculated by numerically integrating the measured densities over 150-day periods within each May–October sampling season, starting and ending with zero density observations (Stangel and Shambaugh, 2005).

Phytoplankton

Large phytoplankton were sampled by the LTMP beginning in 2006 using a 63 μm mesh Wisconsin net. Samples were collected by vertical net tows from twice the Secchi depth and preserved with acid Lugol's solution for later analysis. Individuals with at least one linear dimension >50 μm were identified to the lowest taxonomic level practical and enumerated. Ten randomly selected individuals from each taxon were measured and the median values of these dimensions were used with standard geometric formulae to determine a representative biovolume per cell (Wetzel and Likens, 2000).

Statistical analysis

All sampling results were averaged for each date to reduce field replicates to a single value per sampling date. Locally weighted scatterplot smoothing (LOWESS) was used to visualize temporal trends in the data including any non-linearity, while illustrating the variability in the data. LOWESS identifies the centerline of the time series plots, illustrating the underlying trends amidst the considerable variability present in the data (Helsel and Hirsch, 2005). Regression window widths for weighting were controlled using moderate smoothness values of 0.4–0.6 for most variables in this analysis.

One of the concerns about using data from monitoring programs with different sampling methods operating over different time periods (Table 1) is the potential for an apparent temporal trend to be an artifact of methodological differences. To check for such differences and minimize the influence of method artifacts, annual mean values for STD, TP, and Chl-a were calculated from data that were log-transformed for normality for all lake stations and years that were sampled concurrently by the LMP and LTMP programs. A paired *t*-test ($p<0.05$) was used to test the statistical significance of any differences between these two sampling programs in the distributions of the annual means for each lake station and water quality variable. Where significant differences were found, separate LOWESS curves were fit to data from the LMP and LTMP programs and shown in parallel. Since the H–P surveys did not overlap in time with the other two monitoring programs, it was not possible to test for bias in the H–P results relative to the LMP or LTMP data.

The statistical significance of temporal trends suggested by the LOWESS plots was tested by linear regression of concentration vs. time in decimal years over the entire monitoring period. Means of the annual mean values were compared between the LMP and LTMP for each variable and lake station. For any variable and lake station found to be significantly different between the two programs during concurrent time periods, indicating possible programmatic bias, data from the LMP only were retained for the regressions because of the longer period of record from this program, and LTMP results were excluded. Unless noted otherwise, trends reported in the results as "increasing" or "decreasing" had slopes that were significantly different from zero ($p<0.05$), based on the linear regression analysis.

Intervals between sampling events within a particular lake region were typically a week or more, but the potential for temporal autocorrelation and overstated statistical significance of the regression results exists. While not tested, the influence of possible temporal autocorrelation on the general findings of these analyses is likely to be small because most regressions noted as statistically significant had p values well below the 0.05 criterion.

Results and discussion

Temperature

August mean surface water temperatures increased during the period of 1964–2009 in all lake regions, with statistically significant increases occurring at eight of ten stations (Fig. 3, Table 2). Linear regression lines were shown for temperature in Fig. 3 instead of LOWESS plots because of the discontinuity in the time series. August mean surface temperatures increased by 1.6–3.8 °C (0.035–0.085 °C/year) in these eight lake regions over this 46-year period.

The increasing trends in August surface water temperatures in Lake Champlain during 1964–2009 illustrate an effect of a warming regional climate over this period. The observed rates of summer water temperature increase in Lake Champlain were in a similar range as rates observed in Lake Ontario (0.048 °C/year), Lake Huron (0.084 °C/year), Lake Superior (0.11 °C/year), and Lake Baikal (0.038 °C/year) (Dobiesz and Lester, 2009; Austin and Colman, 2007; Hampton et al., 2008).

Summer air temperatures increased at an average rate of 0.037 °C/year in the Lake Champlain Basin during 1976–2005 (Stager and Thill, 2010, Fig. 1a). The observation that summer water surface temperatures in Lake Champlain increased faster than summer regional air temperatures is consistent with findings from the Great Lakes (Austin and Colman, 2007; Dobiesz and Lester, 2009). Winter ice cover has been declining in Lake Champlain (Stager and Thill, 2010), and the increase in summer water temperatures may be enhanced by greater heat absorption in the absence of ice and the resulting earlier onset of thermal stratification in the spring, as suggested by Austin and Colman (2007) and Stager and Thill (2010).

Secchi disk transparency

SDT is the water quality variable with the longest and most nearly continuous monitoring record in Lake Champlain, with data beginning in 1964 (Fig. 3). There has been a general trend of increasing SDT in lake regions along the main axis of the lake over the past four decades (Table 2). SDT increases ranged between 26 and 48% in the Main Lake, Shelburne Bay, Cumberland Bay, Grand Isle, and Isle LaMotte regions over the monitoring period. Water transparency in the South Lake more than doubled, with most of this increase occurring since the early 1990s. These trends were in contrast to observations in the northeastern regions of the lake where no significant trends were seen in the Northeast Arm, Malletts Bay, or St. Albans Bay over the period of 1964–2009. SDT decreased in Missisquoi Bay by about 25%, primarily since 1980.

Hypolimnetic dissolved oxygen

Three lake regions were chosen for evaluation of trends in hypolimnetic anoxia, including two regions (Northeast Arm and Malletts Bay) with historically known hypolimnetic DO deficits (Myer and Gruendling, 1979) and the Main Lake region where orthograde DO profiles with a metalimnetic minimum exist during the summer. There were no significant trends in late-summer hypolimnetic DO concentrations or June–September depletion rates in any of these lake regions during the monitoring period of 1990–2009 (Table 2). Examination of late summer depth profiles in the Main Lake did not indicate any change in the extent of the metalimnetic DO minimum during this period.

Sodium and chloride

The trend of increasing Na^+ in the Main Lake first noted by Potash and Henson (1975) continued lakewide through 2005 (Fig. 4, Table 2). Na^+ levels in the Main Lake region tripled since the 1960s. Linear regression lines were shown for Na^+ in Fig. 4 instead of LOWESS plots because of the discontinuity in the time series.

The Cl^- record (Fig. 4, Table 2) showed that the trend of increasing salt concentrations continued in the Main Lake and northern regions through 2009. Cl^- increased in the Main Lake by about 30% since 1992, although concentrations leveled off in recent years. In contrast, Cl^- concentrations declined in the northeastern regions of the lake since 1992 (Missisquoi Bay, St. Albans Bay, Northeast Arm, Malletts Bay), especially during recent years. Cl^- levels in the South Lake, which are affected by a paper mill discharge (Vermont DEC and New York State DEC, 1997), were elevated above concentrations measured elsewhere in the lake but remained stable over the 1992–2009 monitoring period.

Winter road salt application rates on Vermont state highways in the basin ranged between 14,000–30,000 mt/year as Cl^- during 1991–2009 (Fig. 1b). These quantities, when combined with the additional Cl^- used for deicing or summer dust suppression on local town roads in Vermont and on roads in the New York and Quebec portions of the watershed, represent a significant portion of the 125,000 mt/year total Cl^- load to Lake Champlain from all sources estimated for 1990–1992 (Smeltzer and Quinn, 1996). Thus, changes in the rate of road salt application could plausibly account for the Cl^- trends seen in the lake. The 30% drop in road salt use since 1999 in Vermont state highway districts within the Lake Champlain Basin (Fig. 1b) appears to have produced a Cl^- decline in eastern (Vermont-side) regions of Lake Champlain after a lag time of about 5 years (Fig. 4).

Despite the long-term Cl^- increases observed in central and western areas of Lake Champlain, current levels (<25 mg/L) are well below the USEPA (1988) criterion of 230–860 mg/L established to protect ambient aquatic life. However, just as Chapra et al. (2009) noted that Cl^- trends in the Great Lakes serve as "canaries in the coal mine,"

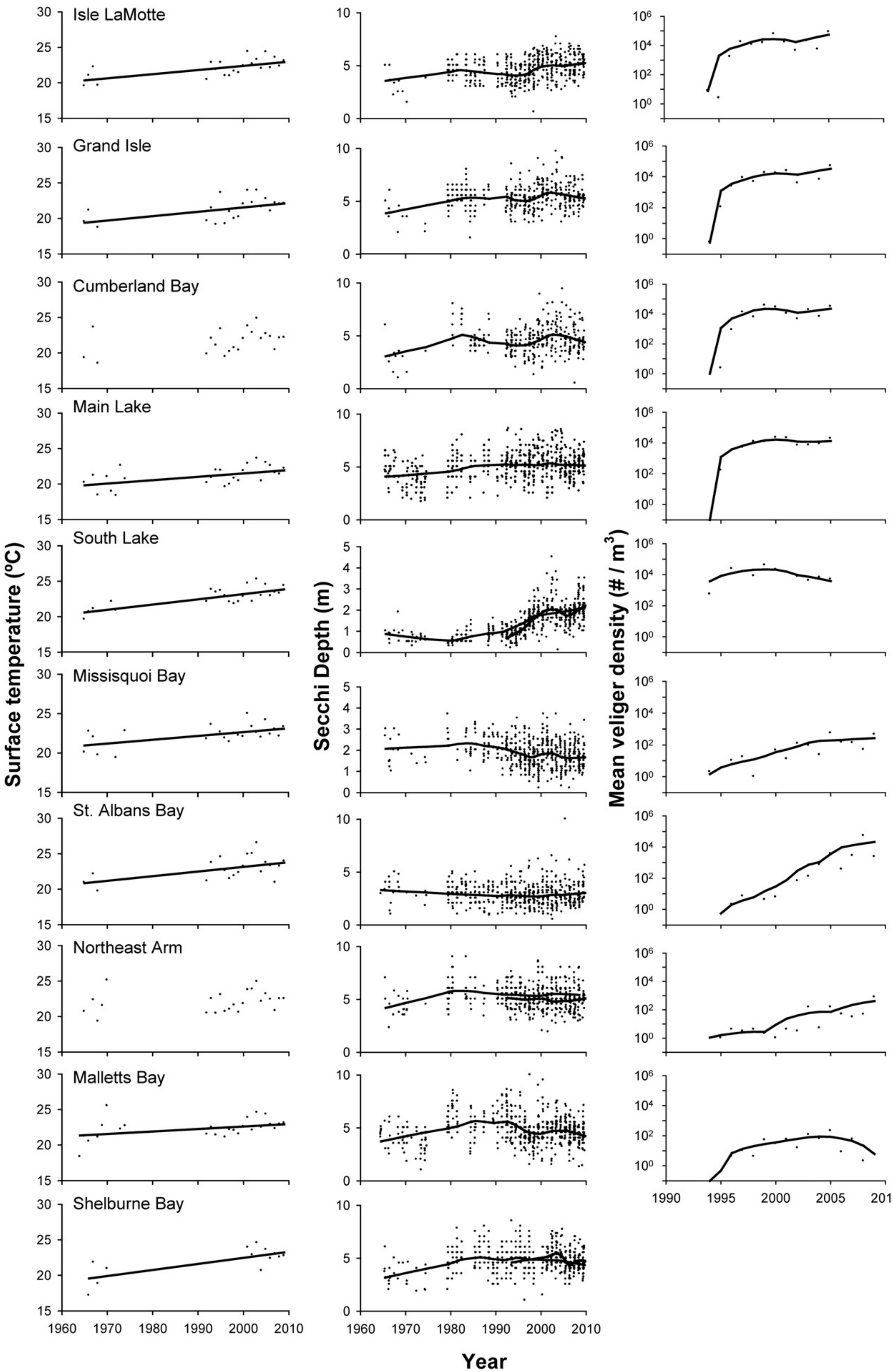
Isle LaMotte
Grand Isle
Cumberland Bay
Main Lake
South Lake
Missisquoi Bay
St. Albans Bay
Northeast Arm
Malletts Bay
Shelburne Bay
Surface temperature (ºC)
Secchi Depth (m)
Mean veliger density (# / m3)
Year

Table 2
Linear regression results for water quality dependent variables vs. time in decimal years. Bold p values indicate slopes that were significantly different from zero ($p<0.05$). Data were from programs including the Henson and Potash surveys (H–P), the Vermont Lay Monitoring Program (LMP), and the Long-Term Water Quality and Biological Monitoring Program on Lake Champlain (LTMP). Footnotes indicate the specific time periods and program data used in the regressions.

Variable/station	Intercept	Slope	p	Variable/station	Intercept	Slope	p
Temp °C				SDT (m)			
Isle LaMotte[a,b]	−80.6	0.0515	**0.010**	Isle LaMotte[a,c]	−66.1	0.0354	<**0.001**
Grand Isle[a,b]	−101.6	0.0615	**0.017**	Grand Isle[a,c]	−35.6	0.0205	**0.001**
Cumberland Bay[a,b]	−71.4	0.0466	0.101	Cumberland Bay[a,c]	−41.2	0.0229	**0.005**
Main Lake[a,b]	−74	0.0477	**0.006**	Main Lake[a,c]	−46.2	0.0257	<**0.001**
South Lake[a,b]	−124.7	0.0739	<**0.001**	South Lake[a,d]	−81	0.0413	<**0.001**
Missisquoi Bay[a,b]	−73.6	0.0481	**0.005**	Missisquoi Bay[a,c]	39.9	−0.0191	<**0.001**
St. Albans Bay[a,b]	−107	0.0651	**0.017**	St. Albans Bay[a,c]	8.6	−0.0029	0.504
Northeast Arm[a,b]	−18.7	0.0205	0.358	Northeast Arm[a,d]	−14.6	0.0101	0.134
Malletts Bay[a,b]	−47.2	0.0349	**0.038**	Malletts Bay[a,c]	16.6	−0.0059	0.212
Shelburne Bay[a,b]	−147.7	0.0851	**0.004**	Shelburne Bay[a,d]	−19.9	0.0124	**0.046**
DO Sept 1 (mg/L)				DO rate (mg/L/d)			
Main Lake 90 m[e,f]	−18.2	0.0145	0.386	Main Lake 90 m[e,f]	0.4	−0.0002	0.251
Northeast Arm 45 m[e,f]	5.2	−0.0005	0.989	Northeast Arm 45 m[e,f]	0.3	−0.0001	0.783
Malletts Bay 25 m[e,f]	−40.9	0.0222	0.765	Malletts Bay 25 m[e,f]	1.2	−0.0006	0.560
Na^+ (mg/L)				Cl^- (mg/L)			
Isle LaMotte[b,g]	−255.7	0.1317	<**0.001**	Isle LaMotte[f,h]	−350.1	0.1811	<**0.001**
Grand Isle[b,g]	−270.5	0.1393	<**0.001**	Grand Isle[f,h]	−401.6	0.207	<**0.001**
Cumberland Bay[b,g]	−267.27	0.1376	<**0.001**	Cumberland Bay[f,h]	−464	0.2382	<**0.001**
Main Lake[b,g]	−259.4	0.1338	<**0.001**	Main Lake[f,h]	−411.5	0.2121	<**0.001**
South Lake[b,g]	−535.3	0.2751	<**0.001**	South Lake[f,h]	19	−0.001	0.978
Missisquoi Bay[b,g]	−101.4	0.0531	<**0.001**	Missisquoi Bay[f,h]	187.8	−0.0901	<**0.001**
St. Albans Bay[b,g]	−205.4	0.1061	<**0.001**	St. Albans Bay[f,h]	66.5	−0.0278	**0.005**
Northeast Arm[b,g]	−167.2	0.0866	<**0.001**	Northeast Arm[f,h]	45.7	−0.0179	**0.013**
Malletts Bay[b,g]	−200.4	0.1033	<**0.001**	Malletts Bay[f,h]	178.77	−0.0844	<**0.001**
Shelburne Bay[b,g]	−298.7	0.1537	<**0.001**	Shelburne Bay[f,h]	−286.3	0.1499	<**0.001**
TP (μg/L)				Chl-a (μg/L)			
Isle LaMotte[i,j]	−54.5	0.0374	0.460	Isle LaMotte[i,k]	34.2	−0.0154	0.238
Grand Isle[i,k]	249	−0.1181	<**0.001**	Grand Isle[i,k]	34.5	−0.0154	0.233
Cumberland Bay[i,k]	562.2	−0.2743	<**0.001**	Cumberland Bay[i,k]	91.4	−0.0439	**0.002**
Main Lake[i,j]	−3.7	0.0102	0.863	Main Lake[i,k]	22	−0.0091	0.449
South Lake[i,j]	418.7	−0.1944	**0.002**	South Lake[i,k]	−2.2	0.0047	0.857
Missisquoi Bay[i,k]	−1201.2	0.6237	<**0.001**	Missisquoi Bay[i,j]	−1635.3	0.8275	<**0.001**
St. Albans Bay[i,j]	106.1	−0.035	0.752	St. Albans Bay[i,k]	−11.8	0.0113	0.847
Northeast Arm[i,j]	−501	0.2609	**0.001**	Northeast Arm[i,j]	−18.7	0.0116	0.761
Malletts Bay[i,j]	−290.9	0.1523	<**0.001**	Malletts Bay[i,k]	7.5	−0.0021	0.828
Shelburne Bay[i,j]	−156.8	0.0875	**0.023**	Shelburne Bay[i,j]	16.3	−0.006	0.712
TN (mg/L)				Ca^{++} (mg/L)			
Isle LaMotte[f,h]	4.4	−0.002	0.058	Isle LaMotte[b,g]	−32.7	0.0245	**0.014**
Grand Isle[f,h]	6.3	−0.003	**0.001**	Grand Isle[b,g]	−11.3	0.0138	0.155
Cumberland Bay[f,h]	9	−0.0043	<**0.001**	Cumberland Bay[b,g]	−7	0.0115	0.303
Main Lake[f,h]	7.4	−0.0035	<**0.001**	Main Lake[b,g]	−14.9	0.0158	**0.026**
South Lake[f,h]	9.44	−0.0045	<**0.001**	South Lake[b,g]	−37.1	0.0301	0.115
Missisquoi Bay[f,h]	11.54	−0.0054	**0.045**	Missisquoi Bay[b,g]	−64.1	0.0388	0.058
St. Albans Bay[f,h]	9.14	−0.0043	**0.001**	St. Albans Bay[b,g]	−191.4	0.1043	<**0.001**
Northeast Arm[f,h]	6.1	−0.0029	<**0.001**	Northeast Arm[b,g]	−126.3	0.0713	<**0.001**
Malletts Bay[f,h]	4.4	−0.002	**0.026**	Malletts Bay[b,g]	−11.3	0.0118	0.099
Shelburne Bay[f,h]	6.9	−0.0032	**0.030**	Shelburne Bay[b,g]	23.5	−0.0033	0.829

[a] 1964–2009.
[b] H–P,LTMP.
[c] H–P,LMP,LTMP.
[d] H–P,LMP.
[e] 1990–2009.
[f] LTMP.
[g] 1964–2005.
[h] 1992–2009.
[i] 1979–2009.
[j] LMP.
[k] LMP,LTMP.

the Lake Champlain trends point to water quality changes in the smaller waterways of the basin. Recent studies in small Vermont streams have determined that Cl^- concentrations in some urban locations exceeded the USEPA chronic criterion of 230 mg/L between 60 and 80% of the time (Denner et al., 2010; Vermont DEC, unpublished data). The Adirondack Park region in New York has also been affected by winter road maintenance practices, with Cl^- exceeding 80 mg/L in some lakes (Langen et al., 2006).

Fig. 3. Long-term trends in August mean surface water temperature (1964–2009), SDT (1964–2009), and season mean zebra mussel veliger densities (1994–2009). Linear regression lines are shown for lake regions where the slope of the August mean temperature vs. year relationship was significantly different from zero ($p<0.05$). Separate LOWESS curves for SDT were fit to the H–P/LMP data and the LTMP data in lake regions where statistically significant differences were found between monitoring programs during concurrently sampled years. Note that the SDT scales vary, some data points were outside of the plot range, and initial zero values for zebra mussel densities were not plotted.

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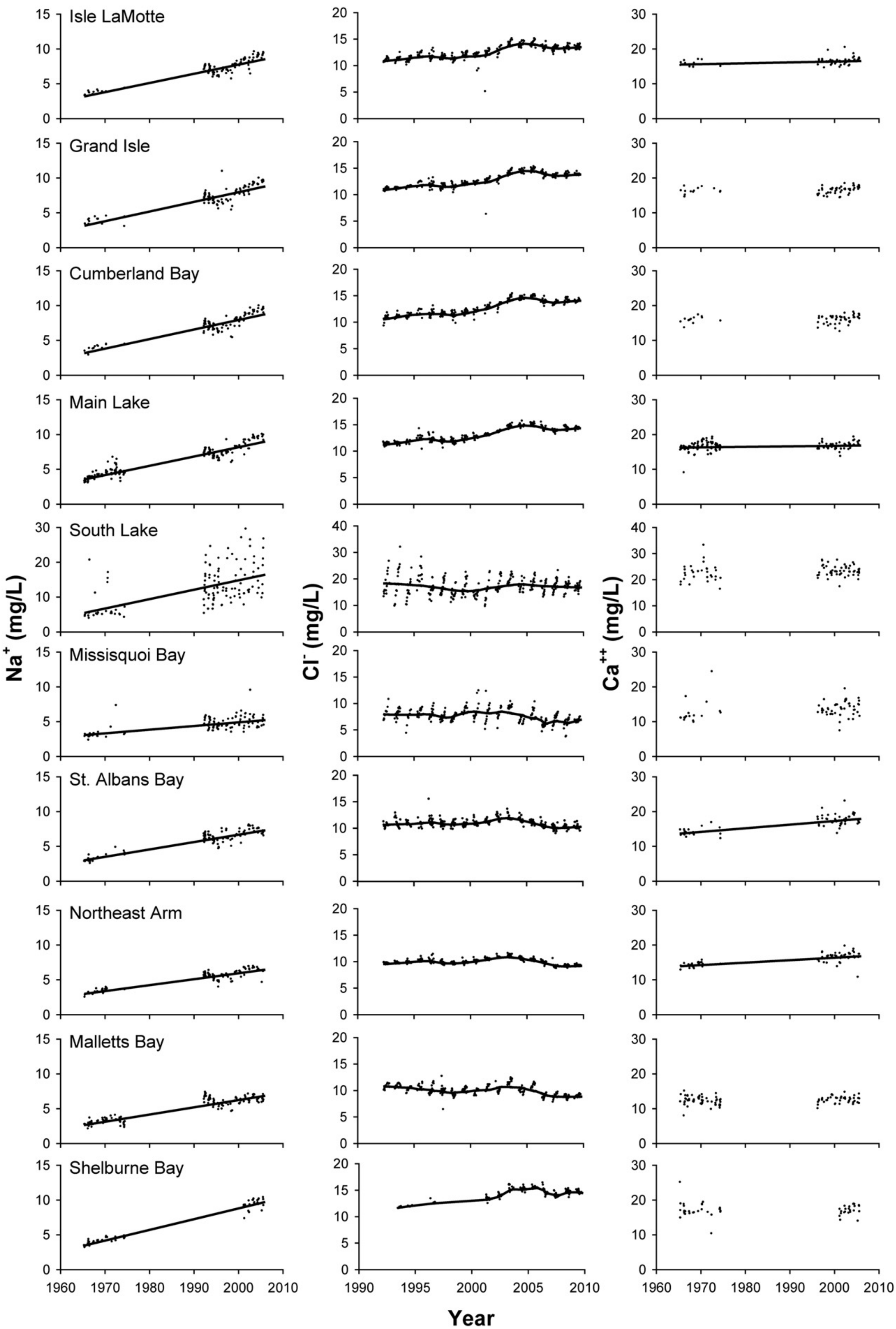

Isle LaMotte
Grand Isle
Cumberland Bay
Main Lake
South Lake
Missisquoi Bay
St. Albans Bay
Northeast Arm
Malletts Bay
Shelburne Bay
Na+ (mg/L)
Cl- (mg/L)
Ca++ (mg/L)
Year
1960 1970 1980 1990 2000 2010
1990 1995 2000 2005 2010

Total phosphorus, chlorophyll-a, and total nitrogen

Trends in TP concentrations in Lake Champlain (Fig. 5) differed among the lake regions, with increases seen in the northeastern regions and stable or declining levels observed along the main axis of the lake (Table 2). A TP increase of 72% (20 μg/L) occurred in Missisquoi Bay over the 1979–2009 monitoring period. The Northeast Arm, Malletts Bay, and Shelburne Bay also had increasing trends during this period, although no trends were seen in St. Albans Bay. No overall trends were observed in the Main Lake or Isle LaMotte regions during the 1979–2009 monitoring period, although decreasing TP concentrations occurred in the Grand Isle, Cumberland Bay, and South Lake regions. A previous analysis limited to the LTMP data found statistically significant increasing linear trends in TP in the Missisquoi Bay, Northeast Arm, and Malletts Bay regions over the period of 1990–2008, but no significant trends in the other regions (Smeltzer et al., 2009).

Chl-a concentrations (Fig. 5, Table 2) showed few trends over the 1979–2009 monitoring period. Missisquoi Bay was an exception where Chl-a levels doubled over this period. A statistically significant decrease was seen in Cumberland Bay. Our findings of increasing TP and Chl-a in Missisquoi Bay since the late 1970s, and elevated but relatively stable levels in St. Albans Bay over this period, are consistent with paleolimnological evidence (Levine et al., 2011).

Given the substantial, long-term efforts to reduce phosphorus loading in the Lake Champlain Basin, the fact that TP and Chl-a concentrations have declined significantly in only a few lake regions and increased in others is disappointing to lake managers. Tributary TP monitoring during 1990–2009, including contributions from non-point sources, showed no overall lake-wide trend in total loadings (Smeltzer et al., 2009). Conversion of land during this period to higher phosphorus-yielding uses (Troy et al., 2007), and greater river flow rates in recent years, may have offset the gains from wastewater treatment. When tributary phosphorus concentrations and loads were normalized for temporal variations in flow (Medalie et al., 2011), decreasing trends were found in many rivers since 1999, suggesting that a watershed response to management efforts may have begun to occur.

There was a general lakewide trend of decreasing levels of TN over the LTMP monitoring period of 1992–2009 (Fig. 5, Table 2). Overall TN declines were about 18% in the Main Lake and adjoining lake regions during this period. TN declines in Missisquoi Bay, St. Albans Bay, and the South Lake were closer to 25% with most of the drop occurring in recent years.

The lakewide decreases in TN are not explained by changes in agricultural practices since nitrogen fertilizer sales and the amount of corn land harvested within the heavily agricultural Missisquoi Bay and St. Albans Bay watersheds both increased since 1990 (Fig. 1d). Reductions in atmospheric nitrogen deposition to the lake's surface or its watershed (Fig. 1e) may have accounted for some of the TN reduction seen in Lake Champlain, but the trends in tributary nitrogen loads during the 1990–2009 monitoring period were not consistent over time. Flow-normalized TN concentrations in nearly all tributaries to Lake Champlain declined since 1999, but these decreases followed a period of generally increasing loads during the prior decade (Medalie et al., 2011). Nitrogen mass balance modeling analyses should be conducted in order to more definitively evaluate the causes for the TN decline in Lake Champlain.

The decreasing TN trend in Lake Champlain contrasts with the increasing nitrate and TN concentrations in Lake Superior (McDonald et al., 2010). However, the Lake Superior trend was documented over a much longer time period (since 1900). Recent data and model scenarios suggest TN and nitrate in Lake Superior may have peaked or begun to decline as a result of reduced loadings or changes in internal processes (McDonald et al., 2010).

Zebra mussel veligers and calcium

After the first zebra mussel adult was discovered in the South Lake region of Lake Champlain in 1993, there was a very rapid increase in zebra mussel veliger densities northward through the Main Lake, Cumberland Bay, Grand Isle, and Isle LaMotte regions (Fig. 3). Season mean veliger densities exceeded 10,000/m^3 within the first few years and have generally stabilized in these regions since then. Veliger densities peaked in the South Lake at 40,000/m^3 in 1999 and have since declined. However, veliger monitoring ended in these regions in 2005. In northeastern lake regions (Missisquoi Bay, St. Albans Bay, Northeast Arm, and Malletts Bay), the veliger population increases were much slower and, with the exception of St. Albans Bay in 2008, season mean densities have not exceeded 1000/m^3.

Lake Champlain has not responded as dramatically to zebra mussel invasion as have other lakes. Increases in SDT have been seen in many areas of Lake Champlain, but declines in TP and Chl-a have been limited in extent. Benthic macroinvertebrate communities in Lake Champlain responded positively to the presence of zebra mussels (Beekey et al., 2004), but incidences of nuisance filamentous green algae are rarely reported in the lake.

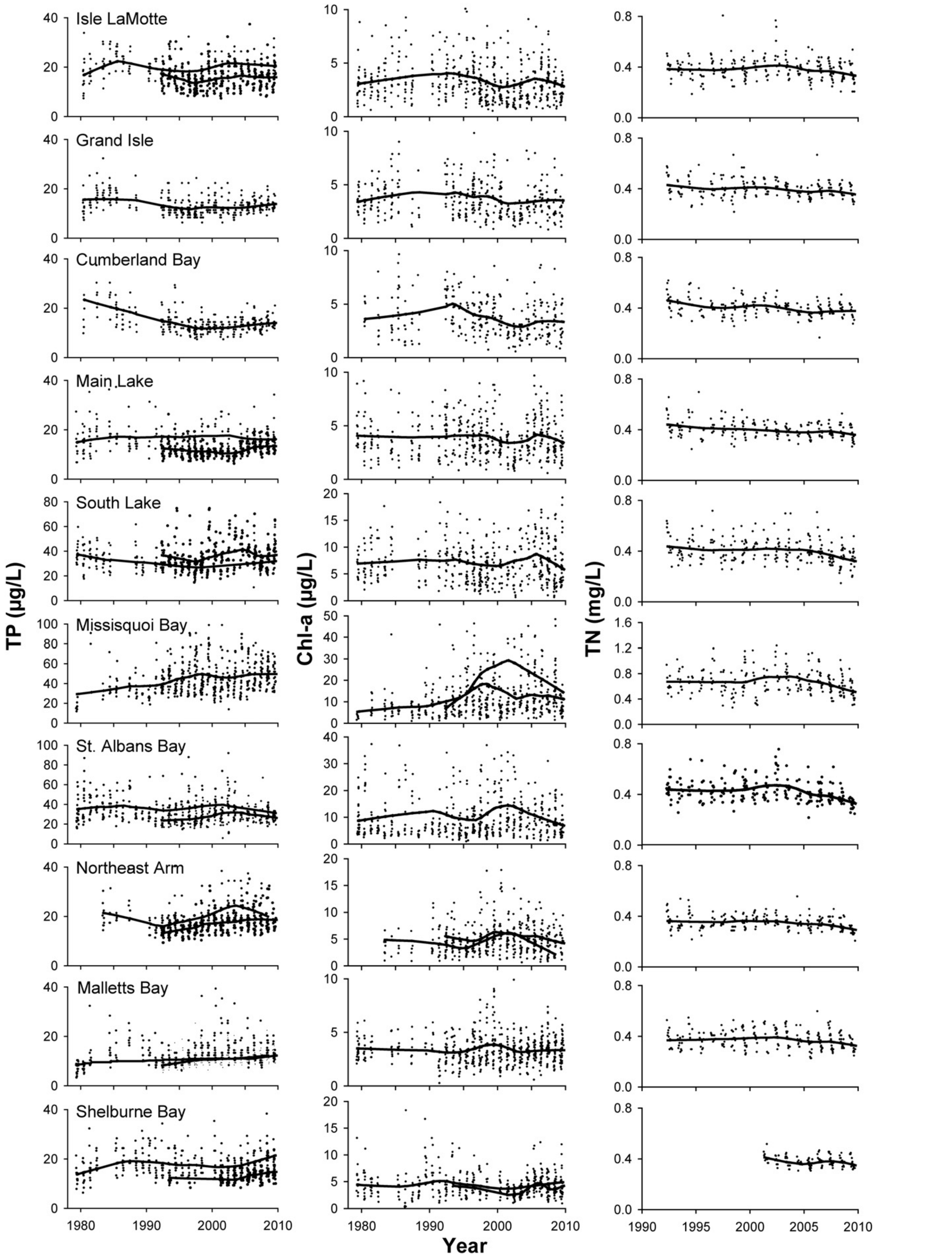
Filtration by adult zebra mussels is probably the major factor responsible for the increasing SDT trends. The mid-1990s timing of the largest transparency increases in the South Lake, Isle LaMotte, and Grand Isle regions (Fig. 3) corresponded to the explosive growth of zebra mussel populations in the lake as indicated by veliger densities. Transparency has not increased, or has decreased, in regions of Lake Champlain where zebra mussel veliger densities have remained relatively low (Missisquoi Bay, St. Albans Bay, the Northeast Arm, and Malletts Bay).

Barbiero et al. (2006) linked transparency increases in Lake Ontario to Ca^{++} uptake by zebra mussels and fewer calcite precipitation events, but no such declines in Ca^{++} have been observed in Lake Champlain. Ca^{++} concentrations in most regions of the lake showed little change between the H–P survey period of 1964–1974 and the LTMP monitoring period of 1992–2005 (Fig. 4, Table 2). However, small but significant positive trends were observed in the Isle LaMotte, Main Lake, St. Albans Bay, and Northeast Arm regions (linear regression lines were shown in Fig. 4 instead of LOWESS plots because of the discontinuity in the time series).

The reason for the much slower expansion of zebra mussel populations in the northeastern regions of Lake Champlain is not clear. Missisquoi Bay, the Northeast Arm, and Malletts Bay are each separated from adjoining lake regions by causeways, but openings in the causeways allow ample opportunity for the introduction of seed populations of veligers through water circulation. Missisquoi Bay and Malletts Bay have the lowest Ca^{++} concentrations among Lake Champlain regions, averaging less than 15 mg/L (Fig. 4). Low calcium is considered limiting to zebra mussels (Mellina and Rasmussen, 1994; Hincks and Mackie, 1997; Frischer et al., 2005). Whittier et al. (2008) considered invasion probability low in areas where Ca^{++} was less than 20 mg/L, noting also that some authors consider 20 mg/L Ca^{++} as being necessary to sustain a reproducing population.

Low calcium is most likely limiting zebra mussel spread in Malletts Bay, where there is sufficient hard substrate available for colonization. Substrate in Missisquoi Bay, St. Albans Bay, and the Northeast Arm is primarily soft, though native mussels, aquatic macrophytes, docks, and other infrastructure offer suitable attachment sites. Zebra mussels were slow to colonize soft sediment in other parts of Lake Champlain, but extensive mats were apparent in some locations by 2000 (Beekey et al.,

Fig. 4. Long-term trends in Na^{+} (1964–2005), Cl^{-} (1992–2009), and Ca^{++} (1964–2005). Linear regression lines are shown for lake regions where the slope of the Na^{+} or Ca^{++} vs. year relationship was significantly different from zero ($p<0.05$). LOWESS trend lines are shown for Cl^{-}. Note that the scales vary, and some data points were outside of the plot range.



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2004). If the trend of increasing Ca^{++} concentrations in northeastern regions of Lake Champlain continues, then zebra mussel populations could expand because substrate is unlikely to be limiting in those areas. Quagga mussels (*Dreissena bugensis*) have not yet been found in Lake Champlain.

Phytoplankton

The LTMP phytoplankton cell count data were used to identify the dominant genera present in each region of Lake Champlain during 2006–2009, and compared in Table 3 with observations from previous studies. The earlier studies by Myer and Gruendling (1979) and Shambaugh et al. (1999) involved counts on whole-water samples, rather than 63 μm mesh net tows as used by the LTMP. Therefore, the dominant genera listed in Table 3 for the earlier time periods were restricted to those that would have been captured as net phytoplankton. Spring, summer, and fall data were combined to assess the dominant genera for these comparisons.

Diatoms (Chrysophyta) were the dominant phytoplankton taxa present throughout Lake Champlain during 1970–1974 and 1991–1992, with the exception of St. Albans Bay where cyanobacteria dominated. Diatoms remain prevalent in most regions of the lake, but there has been a shift to increasing cyanobacteria dominance in northeastern lake regions during the recent time period of 2006–2009. Large colonial and filamentous cyanobacteria are now the dominant taxa in the Northeast Arm and Missisquoi Bay, as well as in St. Albans Bay. While Myer and Gruendling (1979) noted few cyanobacteria in Missisquoi Bay during the 1970s, the bay is now subject to blooms of *Aphanizomenon*, *Microcystis*, and *Anabaena* and the production of cyanotoxins such as microcystin (Watzin et al., 2011). These findings of relatively recent proliferation of cyanobacteria in Missisquoi Bay are consistent with fossil pigment evidence in sediment cores (Levine et al., 2011).

The observed shifts in the Lake Champlain phytoplankton community were likely influenced by a complex interaction of nutrient, food web, and other environmental changes in the lake and its watershed. Cyanobacteria tend to dominate by various competitive mechanisms in lakes where TN:TP ratios or dissolved inorganic nitrogen concentrations are low (Smith, 1983; Nurnberg, 2007). The increased presence of cyanobacteria in northeastern regions of the lake may be related to the decline in TN:TP ratios as a result of decreasing TN concentrations (Fig. 5). However, increases in TP in northeastern regions of Lake Champlain provide an alternate explanation for the greater cyanobacteria presence (Watson et al., 1997; Downing et al., 2001).

In shallow regions such as Missisquoi Bay, higher temperatures at the sediment–water interface could be accelerating internal phosphorus loading during the summer (Jensen and Andersen, 1992). Increased thermal stability resulting from the warmer summer surface water temperatures (Fig. 3) also facilitates cyanobacteria dominance (Wagner and Adrian, 2009). Zebra mussel filtration, and reduction in competition from green algae, have been linked to increases in cyanobacteria in some lakes, and to the proliferation of *M. aeruginosa* in particular (Makarewicz et al., 1999; Vanderploeg et al., 2001; Nichols et al., 2002; Raikow et al., 2004). However, locations in Lake Champlain such as Missisquoi Bay, the Northeast Arm, and Malletts Bay where *Microcystis* or other cyanobacteria have increased host relatively small populations of these mussels (Fig. 3). The introduction of alewife (*Alosa pseudoharengus*) to Lake Champlain in 2003 was linked with an observed loss of large zooplankton (Mihuc et al., 2011), a top–down food web effect that may release cyanobacteria from grazing by large daphnids and other zooplankton (Elser, 1999).

Table 3
Historical changes in the dominant genera of large-celled phytoplankton in Lake Champlain based on cell and colony density observations during spring, summer, and fall.

Lake region	Time period 1970–1974[a]	1991–1992[b]	2006–2009[c]
South Lake	*Microcystis*[d], *Aulocoseira*[e], *Stephanodiscus*[e], *Eudorina*[f], *Aphanizomenon*[d]	Large centric diatoms[e], Large pennate diatoms[e], *Aphanizomenon*[d]	*Aulocoseira*[e], *Aphanizomenon*[d], *Ulothrix*[f]
Shelburne Bay		*Asterionella*[e], *Aulocoseira*[e], *Dinobryon*[e]	*Fragilaria*[e], *Aulocoseira*[e], *Woronichinia*[d]
Main Lake	*Aulocoseira*[e], *Fragilaria*[e], *Anabaena*[d], *Asterionella*[e], *Synedra*[e]		*Fragilaria*[e], *Woronichinia*[d], *Asterionella*[e]
Malletts Bay	*Fragilaria*[e], *Synedra*[e], *Tabellaria*[e], *Peridinium*[g]	*Asterionella*[e], *Aphanizomenon*[d], *Fragilaria*[e]	*Woronichinia*[d], *Fragilaria*[e], *Aphanothece*[d]
Cumberland Bay		*Asterionella*[e], *Fragilaria*[e], *Aulocoseira*[e]	*Fragilaria*[e], *Aphanothece*[d], *Asterionella*[e]
Northeast Arm	*Fragilaria*[e], *Synedra*[e]	*Aphanizomenon*[d], *Fragilaria*[e], *Mougeotia*[f]	*Woronichinia*[d], *Fragilaria*[e], *Aphanizomenon*[d]
Grand Isle		*Asterionella*[e] *Fragilaria*[e] *Aulocoseira*[e]	*Fragilaria*[e] *Aulocoseira*[e] *Woronichinia*[d]
St. Albans Bay	*Anabaena*[d]	*Anabaena*[d], Unidentified trichome[d], *Microcystis*[d]	*Aphanizomenon*[d], *Anabaena*[d], *Aulocoseira*[e], *Fragilaria*[e]
Isle La Motte		Large pennate diatoms[e], *Asterionella*[e], *Aulocoseira*[e]	*Fragilaria*[e], *Microcystis*[d], *Aulocoseira*[e]
Missisquoi Bay	*Aulocoseira*[e], *Asterionella*[e], *Diatoma*[e], *Stephanodiscus*[e]	Large centric diatoms[e], *Microcystis*[d], *Pediastrum*[f]	*Aphanizomenon*[d], *Microcystis*[d], *Anabaena*[d]

[a] Myer and Gruendling (1979).
[b] Shambaugh et al. (1999).
[c] LTMP, this study.
[d] Cyanobacteria.
[e] Chrysophyta.
[f] Chlorophyta.
[g] Pyrrophyta.

Differences between sampling programs

There were several lake regions where the mean of the long-term annual means for SDT, TP, or Chl-a differed between the LTMP and the LMP sampling programs during the same monitoring period, and where LOWESS curves were therefore plotted separately for the two programs in Figs. 3 and 5. However, the directions of the trends indicated for these variables in the LOWESS plots were similar between the two monitoring programs, even where differences in the long-term mean values existed.

Restricting the LTMP data to the June–September season coincident with the LMP program data did not eliminate the bias for any lake region. The direction of the bias, when present, was not consistently positive or negative among the lake regions, which suggests that a difference in sampling technique between programs (Table 1) was probably not the major factor responsible for the bias.

Differences in navigation methods used by the LMP (visual landmarks) and the LTMP (electronic aids) could have led to samples being obtained from slightly different locations within these lake regions.

Fig. 5. Long-term trends in TP (1979–2009), Chl-a (1979–2009), and TN (1992–2009). Separate LOWESS curves were fit to the LMP and LTMP data in lake regions where statistically significant differences were found between monitoring programs during concurrently sampled years. Note that the scales vary, and some data points were outside of the plot range.

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Strong spatial water quality gradients are known to exist in some areas of Lake Champlain, particularly in the South Lake, Shelburne Bay, and St. Albans Bay, and errors in locating sampling locations might explain the discrepancies in the results for these areas. Providing citizen monitors with navigation aids such as global positioning system devices would be an appropriate way to eliminate this potential problem in future monitoring programs on Lake Champlain and other large lakes.

Conclusions

The "long-term" monitoring window of 18–46 years for the data presented here represents an extremely brief period of time relative to the 9000 years that Lake Champlain has existed in its present geologic form. The fact that measureable environmental trends were observed during the monitoring period suggests that anthropogenic influences were primarily responsible. However, the changes in the lake did not always occur as predicted from trends in environmental stressors and management activities. The spread of zebra mussels has been slower than expected in northeastern lake regions and may be limited by low Ca^{++} concentrations. There was no proliferation of cyanobacteria species such as *M. aeruginosa* that could be linked to zebra mussels, as has occurred in some of the Great Lakes. TP and Chl-a declined in some areas of the lake but not in the more eutrophic northeastern lake regions, despite significant management efforts at controlling point and nonpoint sources in the watershed. The lakewide decline in TN was a surprising finding, given the increases in corn production and fertilizer use in the watershed, and might have been due in part to regional reductions in atmospheric nitrogen deposition rates.

The scope and sometimes unexpected nature of environmental changes that have occurred in Lake Champlain illustrate the importance of continuing the long-term monitoring programs. The awareness and understanding of alterations in the lake's ecosystem gained from monitoring can be used to direct management responses in a more timely and effective manner. Lake Champlain experienced historically unprecedented flooding during the spring of 2011, followed by destructive river flows from Tropical Storm Irene in August, 2011. The data provided by the ongoing monitoring programs will be invaluable in assessing the environmental effects of these extreme weather events.

This paper presents only a limited subset of the variables and monitoring sites encompassed in the current monitoring databases, and numerous research questions remain. The authors hope that the availability of the Long-Term Monitoring Program dataset on the internet may stimulate further analyses and investigation of ecological changes in Lake Champlain.

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References

Austin, J.A., Colman, S.M., 2007. Lake Superior summer water temperatures are increasing more rapidly than regional air temperatures: a positive ice-albedo feedback. Geophys. Res. Lett. 34, L06604. doi:10.1029/2006GL029021.

Barbiero, R.P., Tuchman, M.L., 2004. Long-term dreissenid impacts on water clarity in Lake Erie. J. Great Lakes Res. 30 (4), 557–565.

Barbiero, R.P., Tuchman, M.L., Millard, E.S., 2006. Post-dreissenid increases in transparency during summer stratification in the offshore waters of Lake Ontario: is a reduction in whiting events the cause? J Great Lakes Res. 32 (1), 131–141.

Beekey, M.A., McCabe, D.J., Marsden, J.E., 2004. Zebra mussel colonisation of soft sediments facilitates invertebrate communities. Freshw. Biol. 49, 535–545.

Bogdan, K.G., 1978. Estimates of the annual loading of total phosphorus to Lake Champlain. Prep. for the Eutrophication Task Force, Lake Champlain Basin Study. New England River Basins Commission, Burlington, VT.

Burns, N.M., Rockwell, D.C., Bertram, P.E., Dolan, D.M., Ciborowski, J.J.H., 2005. Trends in temperature, Secchi depth, and dissolved oxygen depletion rates in the Central Basin of Lake Erie, 1983–2002. J. Great Lakes Res. 31 (Supplement 2), 35–49.

Canfield Jr., D.E., Brown, C.D., Bachmann, R.W., Hoyer, M.V., 2002. Volunteer lake monitoring: testing the reliability of data collected by the Florida LAKEWATCH Program. Lake Reserv. Manage. 18 (1), 1–9.

Chapra, S.C., Dove, A., Rockwell, D.C., 2009. Great Lakes chloride trends: long-term mass balance and loading analysis. J. Great Lakes Res. 35 (2), 272–284.

Cohn, A.B., Manley, T.O., Manley, P.L., Smeltzer, E., Watzin, M.C., 2007. Lake Champlain: research and management in the presence of history. LakeLine 27 (4), 46–56 North American Lake Management Society. Madison, WI..

Daley, M.L., Potter, J.D., McDowell, W.H., 2009. Salinization of urbanizing New Hampshire streams and groundwater: effects of road salt and hydrologic variability. J. N. Am. Benthol.l Soc. 28 (4), 929–940.

Denner, J.C., Clark Jr., S.F., Smith, T.E., Medalie, L., 2010. Effects of highway road salting on the water quality of selected streams in Chittenden County, Vermont, November 2005–2007. U.S. Geological Survey Scientific Investigations Report 2009–5236. . 43 p.

Dobiesz, N.E., Lester, N.P., 2009. Changes in mid-summer water temperature and clarity across the Great Lakes between 1968 and 2002. J. Great Lakes Res. 35 (3), 371–384.

Downing, J.A., Watson, S.B., McCauley, E., 2001. Predicting cyanobacteria dominance in lakes. Can. J. Fish. Aquat. Sci. 58 (10), 1905–1908.

Effler, S.W., Perkins, M., Johnson, D.L., 1991. Optical heterogeneity in Lake Champlain. J. Great Lakes Res. 17 (3), 322–332.

Eimers, M.C., Winter, J.G., Scheider, W.A., Watmough, S.A., Nicholls, K.H., 2005. Recent changes and patterns in the water chemistry of Lake Simcoe. J. Great Lakes Res. 31 (3), 322–332.

Elser, J.J., 1999. The pathway to noxious cyanobacteria blooms in lakes: the food web as the final turn. Freshw. Biol. 42, 537–543.

Elser, J.J., Andersen, T., Baron, J.S., Bergstrom, A.K., Jansson, M., Kyle, M., Nydick, K.R., Steger, L., Hessen, D.O., 2009. Shifts in lake N:P stoichiometry and nutrient limitation driven by atmospheric nitrogen deposition. Science 326, 835–837.

Eyles, N., Meriano, M., 2010. Road-impacted sediment and water in a Lake Ontario watershed and lagoon, City of Pickering, Ontario, Canada: an example of urban basin analysis. Sediment. Geol. 224 (1), 15–28.

Fahnenstiel, G., Nalepa, T., Pothoven, S., Carrick, H., Scavia, D., 2010. Lake Michigan lower food web: long-term observations and *Dreissena* impact. J. Great Lakes Res.. doi:10.1016/j.jglr.2010.05.009

Frischer, M.E., McGrath, B.R., Hansen, A.S., Vescio, P.A., Wyllie, J.A., Wimbush, J., Nierzwicki-Bauer, S.A., 2005. Introduction pathways, differential survival of adult and larval zebra mussels (*Dreissena polymorpha*), and possible management strategies, in an Adirondack Lake, Lake George, NY. Lake Reserv. Manage. 21 (4), 391–402.

Hampton, S.E., Izmesteva, L.R., Moore, M.V., Katz, S.L., Dennis, B., Silow, E., 2008. Sixty years of environmental change in the world's largest freshwater lake—Lake Baikal, Siberia. Global Change Biol. 14 (8), 1947–1958.

Helsel, D.R., Hirsch, R.M., 2005. Statistical methods in water resources. U.S. Geological Survey, Techniques of Water Resources Investigations, Book 4, Chapter A3. .

Henson, E.B., Potash, M., 1966. A synoptic survey of Lake Champlain, summer 1965. Univ. Michigan Great Lakes Res. Div. Publ. No. 15, pp. 38–43.

Henson, E.B., Potash, M., 1987. Sampling strategies for detecting water quality trends in Lake Champlain. Completion Report to the U.S. Dept. Interior Office of Water Resources and Technology. Project No. 03. University of Vermont, Burlington.

Higgins, S.N., Vander Zanden, M.J., 2010. What a difference a species makes: a meta-analysis of dreissinid mussel impacts on freshwater ecosystems. Ecol. Monogr. 80 (2), 179–196.

Hincks, S.S., Mackie, G.L., 1997. Effects of pH, calcium, alkalinity, hardness, and chlorophyll on the survival, growth, and reproductive success of zebra mussel (*Dreissena polymorpha*) in Ontario lakes. Can. J. Fish. Aquat. Sci. 54 (9), 2049–2057.

Jankowski, T., Livingstone, D.M., Buhrer, H., Forster, R., Niederhauser, P., 2006. Consequences of the 2003 European heat wave for lake temperature profiles, thermal stability, and hypolimnetic oxygen depletion: implications for a warmer world. Limnol. Oceanogr. 51 (2), 815–819.

Jensen, H.S., Andersen, F.O., 1992. Importance of temperature, nitrate, and pH for phosphate release from aerobic sediments of four shallow, eutrophic lakes. Limnol. Oceanogr. 37 (3), 577–589.

Kauschal, S.S., Groffman, P.M., Likens, G.E., Belt, K.T., Stack, W.P., Kelly, V.R., Band, L.E., Fisher, G.T., 2005. Increased salinization of fresh water in the northeastern United States. Proc. Nat. Acad. Sci. 102 (38), 13517–13520.

Langen, T.A., Twiss, M., Young, T., Janoyan, K., Stager, J.C., Osso Jr., J., Prutzman, H., Green, B., 2006. Environmental impacts of winter road management at the Cascade Lakes and Chapel Pond. Clarkson Center for the Environment. Report 1. Clarkson University, Potsdam, NY.

Levine, S.N., Lini, A., Ostrofsky, M.L., Bunting, L., Burgess, H., Leavitt, P.R., Reuter, D., Lami, A., Guilizzoni, P., 2011. The eutrophication of Lake Champlain's Northeast Arm: insights from paleolimnological analyses. J. Great Lakes Res.. doi:10.1016/j.jglr.2011.07.007

Lovett, G.M., Burns, D.A., Driscoll, C.T., Jenkins, J.C., Mitchell, M.J., Rustad, L., Shanley, J.B., Likens, G.E., Haeuber, R., 2007. Who needs environmental monitoring? Front. Ecol. Environ. 5 (5), 253–260.

MACTEC Engineering and Consulting, Inc., 2010. Clean Air Status and Trends Network (CASTNET) 2008 annual report. Prep. for U.S. Environmental Protection Agency. Washington, D.C.

Makarewicz, J.C., Lewis, T.W., Bertram, P., 1999. Phytoplankton composition and biomass in the offshore waters of Lake Erie: pre- and post-*Dreissena* introduction (1983–1993). J. Great Lakes Res. 25 (1), 135–148.

Manley, T.O., Hunkins, K.L., Saylor, J.H., Miller, G.S., Manley, P.L., 1999. Aspects of summertime and wintertime hydrodynamics of Lake Champlain. In: Manley, T.O., Manley, P.L. (Eds.), Lake Champlain in Transition: From Research Toward Restoration: American Geophysical Union. Water Sci. Appl., 1, pp. 67–115. Washington, D.C.

Marsden, J.E., 1992. Standard protocols for monitoring and sampling zebra mussels. Illinois Natural History Survey Biological Notes 138. . Champaign, IL.

Marsden, J.E., Hauser, M., 2009. Exotic species in Lake Champlain. J. Great Lakes Res. 35 (2), 250–265.

Matthews, D.A., Effler, S.W., 2006. Assessment of long-term trends in the oxygen resources of a recovering urban lake, Onondaga Lake, NY. Lake Reserv. Manage. 22 (1), 19–32.

McDonald, C.P., Urban, N.R., Casey, C.M., 2010. Modeling historical trends in Lake Superior total nitrogen concentrations. J. Great Lakes Res. 36, 715–721.

Medalie, L., Smeltzer, E., 2004. Status and trends of phosphorus in Lake Champlain and its tributaries, 1990–2000. In: Manley, T., et al. (Ed.), Lake Champlain: Partnership and Research in the New Millennium. Kluwer Academic/Plenum Publishers, New York, pp. 191–219.

Medalie, L., Hirsch, R.M., Archfield, S.A., 2011. Use of flow-normalization to evaluate nutrient concentration and flux changes in Lake Champlain tributaries, 1990–2009. J. Great Lakes Res.. doi:10.1016/j.jglr.2011.10.062

Mellina, E., Rasmussen, J.B., 1994. Patterns in the distribution and abundance of zebra mussel (*Dreissena polymorpha*) in rivers and lakes in relation to substrate and other physicochemical factors. Can. J. Fish. Aquat. Sci. 51 (5), 1024–1036.

Mida, J.L., Scavia, D., Fahnenstiel, G.L., Pothoven, S.A., Vanderploeg, H.A., Dolan, D.M., 2010. Long-term and recent changes in southern Lake Michigan water quality with implications for present trophic status. J. Great Lakes Res.. doi:10.1016/j.jglr.2010.03.010

Mihuc, T.B., Dunlap, F., Binggeli, C., Pershyn, C., Myers, L., Groves, A., Waring, A., 2011. Long-term patterns in Lake Champlain's zooplankton: 1992–2010. J. Great Lakes Res.. doi:10.1016/j.jglr.2011.08.006

Mullaney, J.R., Lorenz, D.L., Arntson, A.D., 2009. Chloride in groundwater and surface water in areas underlain by the glacial aquifer system, northern United States. U.S. Geological Survey Scientific Investigations Report 2009–5086. . 41 p.

Myer, G.E., Gruendling, G.K., 1979. Limnology of Lake Champlain. Prep. for New England River Basins Commission. . Burlington, VT.

Nichols, K.K., Heintsch, L., Carney, E., 2002. Univariate step-trend and multi-variate assessments of the apparent effects of P loading reductions and zebra mussels on the phytoplankton of the bay in Quinte, Lake Ontario. J. Great Lakes Res. 28 (1), 15–31.

Novotny, E.V., Stefan, H.G., 2009. Projections of chloride concentrations in urban lakes receiving road de-icing salt. Water Air Soil Pollut.. doi:10.1007/s11270-009-0297-0

Nurnberg, G.K., 2007. Low-nitrate-days (LND), a potential indicator of cyanobacteria blooms in a eutrophic hardwater reservoir. Water Qual. Res. J. Can. 42 (4), 269–283.

Picotte, A., Pomeroy, S., 2000. Vermont Lay Monitoring Manual. Vermont Agency of Natural Resources, Waterbury, VT.

Potash, M., Henson, E.B., 1975. Chemical changes in Lake Champlain: a decade of observations. Verh. Internat. Verein. Limnol. 19, 421–428.

Potash, M., Sundberg, S.E., Henson, E.B., 1969. Characterization of water masses of Lake Champlain. Verh. Internat. Verein. Limnol. 17, 140–147.

Raikow, D.F., Sarnelle, O., Wilson, A.E., Hamilton, S.K., 2004. Dominance of the noxious cyanobacterium *Microcystis aeruginosa* in low-nutrient lakes is associated with exotic zebra mussels. Limnol. Oceanogr. 49 (2), 482–487.

Robinson, K.W., Campbell, J.P., Jaworski, N.A., 2003. Water-quality trends in New England rivers during the 20th century. U.S. Geological Survey Water Resources Investigations Report 03–4012.

Rockwell, D.C., Warren, G.J., Bertram, P.E., Salisbury, D.K., Burns, N.M., 2005. The U.S. EPA Lake Erie indicators monitoring program 1983–2002: trends in phosphorus, silica, and chlorophyll *a* in the Central Basin. J. Great Lakes Res. 31 (Suppl. 2), 23–34.

Schindler, D.W., 2009. Lakes as sentinels and integrators for the effects of climate change on watersheds, airsheds, and landscapes. Limnol. Oceanogr. 54, 2349–2358 (6, part 2).

Shambaugh, A.D., Duchovnay, A., McIntosh, A., 1999. A survey of Lake Champlain's plankton. In: Manley, T.O., Manley, P.L. (Eds.), Lake Champlain in Transition: from research toward restoration: American Geophysical Union. Water Sci. Appl., 1, pp. 323–340. Washington, D.C.

Smeltzer, E., Quinn, S., 1996. A phosphorus budget, model, and load reduction strategy for Lake Champlain. Lake Reserv. Manage. 12 (3), 381–393.

Smeltzer, E., Dunlap, F., Simoneau, M., 2009. Lake Champlain phosphorus concentrations and loading rates, 1990–2008. Lake Champlain Basin Program Tech. Rep. 57. . Grand Isle, VT.

Smith, V.H., 1983. Low nitrogen to phosphorus ratios favor dominance by blue-green algae in lake phytoplankton. Science 221, 669–671.

Stager, J.C., Thill, M., 2010. Climate change in the Lake Champlain Basin: what natural resource managers can expect and do. Prep. for The Nature Conservancy. Keene Valley, NY and Montpelier, VT.

Stangel, P., Shambaugh, A., 2005. Lake Champlain 2004 zebra mussel monitoring program. Final Report. Vermont Dept. Environ. Conserv, Waterbury, VT.

Trowbridge, P.R., Sassan, D., Heath, D.L., Walsh, E.M., 2010. Relating road salt to exceedances of the water quality standard for chloride in New Hampshire Streams. Environ. Sci. Technol. 44 (13), 4903–4909.

Troy, A., Wang, D., Capen, D., 2007. Updating the Lake Champlain Basin land use data to improve prediction of phosphorus loading. Lake Champlain Basin Program Tech. Rep, 54. Grand Isle, VT.

U.S. Environmental Protection Agency, 1988. Ambient Water Quality Criteria for Chloride —1988. Office of Water, Washington D.C. EPA 440/5-88-001.

Vanderploeg, H.A., Liebig, J.R., Carmichael, W.W., Agy, M.A., Johengen, T.H., Fahnenstiel, G.L., Nalepa, T.F., 2001. Zebra mussel (*Dreissena polymorpha*) selective filtration promoted toxic *Microcystis* blooms in Saginaw Bay (Lake Huron) and Lake Erie. Can. J. Fish. Aquat. Sci. 58 (6), 1208–1221.

Vermont Agency of Natural Resources and Vermont Agency of Agriculture, Food and Markets, 2010. Vermont clean and clear action plan 2009 annual report. Submitted to the Vermont General Assembly. . Montpelier, VT.

Vermont Department of Environmental Conservation and New York State Department of Environmental Conservation, 1997. A phosphorus budget, model, and load reduction strategy for Lake Champlain. Lake Champlain Diagnostic-Feasibility Study Final Report. . Waterbury, VT and Albany, NY.

Vermont Department of Environmental Conservation and New York State Department of Environmental Conservation, 2010. Long-term water quality and biological monitoring project for Lake Champlain. Quality Assurance Project Plan. : Prep. for Lake Champlain Basin Program. Grand Isle, VT.

Wagner, C., Adrian, R., 2009. Cyanobacteria dominance: quantifying the effects of climate change. Limnol. Oceanogr. 54, 2460–2468 (6, part 2).

Watson, S.B., McCauley, E., Downing, J.A., 1997. Patterns in phytoplankton taxonomic composition across temperate lakes of differing nutrient status. Limnol. Oceanogr. 42 (3), 487–495.

Watzin, M.C., 2007. The promise of adaptive management. In: Schepf, M., Cox, C. (Eds.), Managing agricultural landscapes for environmental quality: strengthening the science base. Soil and Water Conservation Society Press, Ankeny, IA, pp. 147–158.

Watzin, M., Fuller, S., Bronson, L., Gorney, R., Shuster, L., 2011. Monitoring and evaluation of cyanobacteria in Lake Champlain, summer 2009. Lake Champlain Basin Program Tech. Rep. No. 61. . Grand Isle, VT.

Wetzel, R.G., Likens, G.E., 2000. Limnological Analyses, 3rd ed. Springer-Verlag, New York. 429 pp.

Whittier, T.R., Ringold, P.L., Herlihy, A.T., Pierson, S.M., 2008. A calcium-based invasion risk assessment for zebra and quagga mussels (*Dreissena* spp.). Front. Ecol. Environ. (6). doi:10.1890/070073.