The eutrophication of an urban lake: the physical, chemical, and biological evidence

by

Leslie E. Nasmith

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Abstract

Cultural eutrophication of urban freshwater ecosystems is an increasingly common problem. Contemporary and continuous monitoring of lake trophy, particularly where historical data exist, allows for both short-term (seasonal) and long-term (decadal) trends to be assessed and the insights gained can provide a context for the contemporary and future prospects for the lake and a basis for improved watershed management decisions and practices. During summer 2004, water samples were collected and a suite of variables (water temperature; dissolved oxygen, calcium, nitrate, chloride, and ammonium concentration; conductivity, pH, turbidity, transmittance, and Secchi depth) were measured weekly over a 14 week period in Springfield Lake, Halifax Regional Municipality, Halifax, N.S. Biological variables measured included chlorophyll-*a* concentration in the summer, and the benthic macroinvertebrate community in autumn. These data are temporally analyzed (seasonal trends etc.) and are used in a comparison with historical data from the 1970s and 1990s (Mandaville 2000). Temporal comparisons show that Springfield Lake has become increasingly eutrophic over the last 3 decades as evidenced by the decrease in dissolved oxygen saturation in the lower 1 to 3 m of the water column ($p<0.03$), and increases in chlorophyll *a* ($p=<0.001$), and calcium ion ($p=0.02$). In parallel, nitrate and ammonium ion concentrations and phosphorous (limited data) were all high in 2004. In contrast, neither Secchi disk depth ($p=0.46$) nor turbidity has changed ($p=0.05$) over the past 3 decade and the Lake is now less acidic ($p<0.001$) than in 1977. The benthic macroinvertebrate community in 1996 indicated the lake had poor to very poor water quality with probable moderate pollution. The same indication of water quality was observed in 2004 with moderate to severe pollution. The long-term trend in Springfield Lake is one of increasing eutrophication and is likely due to anthropogenic causes. There is some evidence that the lake is being influenced by runoff after rain events, causing increases in conductivity and calcium, the latter possibly due to watershed runoff of lime used for lawn maintenance. Springfield Lake is a headwater lake and therefore provides a unique opportunity for management and without proper and immediate management Springfield Lake could very likely become eutrophic.

Dedication

This work is dedicated to Mungo Nasmith (1881-1956) in appreciation of his purchase of 2.5 acres of land on Oxbow Lake, Muskoka, Ontario, and the cabins he built there for his family. Though I never met him, it is because of him that I have my existing respect for and appreciation of nature.

Table of Contents

List of figures and tables

List of Abbreviations

$\mu g\ L^{-1}$	microgram per liter, equivalent to parts per billion
$\mu S\ cm^{-1}$	micro siemens per centimeter, equivalent to micro mho
ADST	Atlantic Daylight Savings Time
ASTP	average score per taxon
BBI	Belgian Biotic Index
BMWP	Biological Monitoring Working Party (De Pauw and Vanhooren 1983)
Ca^{2+}	Calcium ion
Chl-*a*	Chlorophyll-*a*
Cl^{-}	Calcium ion
D	Simpson's Diversity Index (Simpson 1949)
DO	Dissolved Oxygen
FBI	Family Biotic Index (Hilsenhoff 1988 as cited in Kirsh 1999)
GPS	Global Positioning System
H	Shannon-Wiener Diversity Index (Shannon and Weiner 1963)
HRM	Halifax Regional Municipality
J	Evenness Index (Hill 1973)
LED	Light Emitting Diode
NH^{4+}	Ammonium ion
NO_3^{-}	Nitrate ion
N:P	nitrogen to phosphorous ratio
NTU	Nephelometric Turbidity Units
QEII	Queen Elizabeth II
r	Pearson's correlation coefficient
S	Number of taxa per sample
SCD	Secchi depth
SU	systematic units
TSI	Trophic State Index (Carlson 1977)

Acknowledgements

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Introduction

Freshwater makes up < 5% of the world's water, though it is arguably the most valuable resource for all life on the planet (Beeton 2002). A freshwater lake ecosystem includes the lake, the shoreline, and the watershed (Dillon and Rigler 1975). Freshwater ecosystems serve many uses to humans, primarily as potable water, and secondarily for food resources, irrigation, waste disposal, and recreation (Beeton 2002). Independent of human need, lakes are integral to the functioning of the ecosystem, as habitat for numerous aquatic organisms, and as a food and water resource for terrestrial organisms. Lakes serve in local (micro) climate regulation and act as carbon sinks (Jansson and Nohrstedt 2001). The very services that make preservation of lakes essential are the same services that attract human populations to live on or near them (Beeton 2002) and consequently and paradoxically degrade the ecosystem (Winter and Dillon 2005)

Human presence impacts ecosystems through the creation of unnatural nutrient sources, alteration of shoreline habitat, increased runoff and erosion, sewage pollution, and water removal or damming. Global anthropogenic stressors leading to increased acid precipitation, ultra violet radiation, and climate change (Williamson *et al.* 1999) exacerbate the degradation of freshwater systems. All of these stressors have negative impacts on the lakes; decreasing water and habitat quality, and upsetting the nutrient balance, inevitably leading to eutrophication (Williamson *et al.* 1999).

There are four states on the lake trophy continuum; though there is no clear delineation between these states (Carlson 1977). According to Wetzel (1975), oligotrophy is characterised by low rates of productivity due to low input of inorganic nutrients, mesotrophy is a state of intermediate productivity, and eutrophy is a condition of high nutrient concentration and primary production. Hypereutrophy is a state of extreme nutrient loading (Carlson and Simpson 1996). Trophic status is the most frequently used descriptor for lake classification (Williamson *et al.* 1999) and different classifications use different criteria that may result in a lake being classified simultaneously in different states (Carlson 1977). Given the differences among lakes in terms of morphology and environment, changes between states do not occur under any standard set of circumstances nor at a standard rate (Carlson 1977). Thus, there is no clear definition of when nutrient enrichment in a lake causes adverse effects (Pretty *et al.* 2003). The Trophic State Index (*TSI*) developed by Carlson (1977) places lakes on a scale of 1-100 based on Secchi disk depth, chlorophyll-*a*, or total phosphorous concentration where minimum values represent the oligotrophic condition and maximum values represent a eutrophic or hypereutrophic condition.

Cultural eutrophication is a frequently observed environmental problem (Carpenter *et al.* 1998) driven by point and non-point nutrient sources (Pretty *et al.* 2003) and it has been linked directly to urbanization and expanded agricultural practices (Diaz 2001). Eutrophication can cause oxygen depletion in the water column and the hypoxic zones in the world are closely associated with developed watersheds or human-inhabited areas that deliver large quantities of nutrients to the water (Diaz 2001). In addition to oxygen depletion, changes in tropic status have been linked to changes in macrofauna and macrophytes (Staicer 1994a and 1994b). Cultural eutrophication also has inherent socio-economic costs. Eutrophication of lakes reduces the value of the water body, including reduced value of waterside dwellings, cost of dredging/cutting excessive weed-growth, and reduced recreational value of water bodies for leisure and fishing (Pretty *et al.* 2003).

Changes in lake trophy can occur naturally over time without anthropogenic influence (Venugopalan *et al.* 1998). Therefore, eutrophication of a lake is not necessarily an issue unless the eutrophication in unnatural. Nevertheless, the past condition of a lake is necessary to place contemporary trophic status in context. Direct comparison with historical data is the best method for assessing acute and chronic change and ascribing cause (Hall and Smol 1996). Long-term lake monitoring projects are relatively few and often lakes will be sampled once a year or with an irregularity that provides poor resolution of temporal change. Historical and contemporary data are necessary and essential to adequately assess change over time.

A primary objective of long-term limnological monitoring is to assess the state of a lake with respect to its historical condition so that changes can be studied and documented, and that negative impact processes may be mitigated thorough proper management. Management of freshwater resources is a large and often difficult task that includes numerous interests: federal, provincial, and local governments; homeowners; non-governmental organizations; and private businesses. The ultimate goal of any urban lake management initiative should be the maintenance or return of a water body to its natural condition, based on what is known historically. For eutrophic lakes this means long term (years to decades) management (Carpenter *et al*. 1998) as the restoration of eutrophied lakes has proven to be complex (Sheffer *et al.*1993) and not simply case of nutrient reduction (Phillips *et al.*1999; Lau and Lane 2002).

The objective of my study was to determine the contemporary physical, chemical, and biological state of the water in Springfield Lake, an urban lake in the Halifax Regional Municipality, and to compare the contemporary state to historical data collected in the 1970s to quantify changes over the last three decades. Measures of the contemporary biological state include estimates of the benthic macroinvertebrate diversity that I compare to similar estimates determined in the late 1990s.

My working (and perhaps mundane) hypothesis is that Springfield Lake is more eutrophic now than in the 1970s and that there has been a trend toward a more eutrophic condition as shown in the physical, chemical, and biological data. I further propose that any significant change should be reflected in the benthic macroinvertebrate data. My overall aim is to provide the analyses to document changes in Springfield Lake over time such that appropriate management actions can be taken where necessary and relevant.

Materials and Methods

Study area

Springfield Lake is an urban lake located in Middle Sackville, Halifax Regional Municipality, Nova Scotia, Canada (**Figure 1**). The shoreline perimeter is 5.6 km, the mean depth is 3 m and the maximum depth is 5 m (Mandaville 2000). Springfield is a headwater lake that is fed solely by underground springs and runoff. Located in the Schubenacadie watershed, it ultimately feeds into the Bay of Fundy. It lies in pyretic slate bedrock that is high in hydrogen and sulphur (Kerekes *et al*. 1986). The shoreline is fully developed (lined by private homes and cottages) with over 530 houses in the 500 ha watershed (Grace, E., Springfield Lake Watch, Middle Sackville, Nova Scotia., pers. comm.). Homes on streets adjacent to the lake receive municipal sewage treatment. There is a HRM Water Pollution Control Plant (secondary sludge activation), constructed in 1987, that discharges effluent into a stream outflowing at the north end of the lake (Halifax Regional Municipality 2002). There are 12 lift stations around the lake (**Figure 1**), four of which discharge directly into the lake in times of high rainfall in excess of 45, 70, or 80 mm depending on the lift station (unpublished data, Halifax Regional Municipality, Environmental Management Services, 21 Mount Hope Ave., Dartmouth, NS). There is a public beach at the north western end of the Lake that was closed periodically in the summer of 2004 for *Escherichia coli* contamination (unpublished data, Halifax Regional Municipality, Environmental Management Services, 21 Mount Hope Ave., Dartmouth, NS).

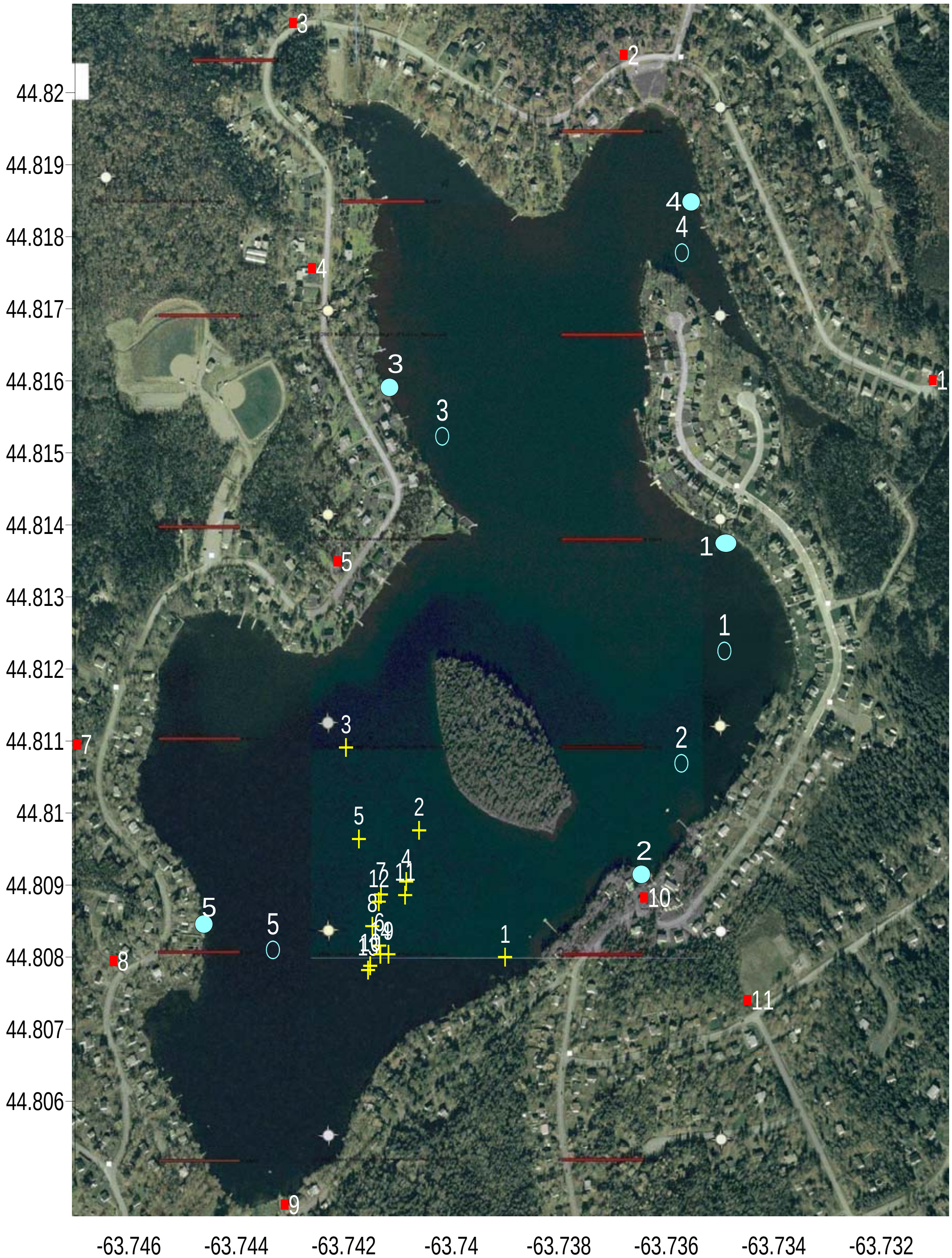


Figure 1. Aerial photographic chart (GPS geo-referenced) of Springfield Lake, Halifax Regional Municipality, showing sampling locations for water quality monitoring (yellow crosses; 04 June to 02 September 2004), benthic macroinvertebrate sampling (empty blue circles; 27 and 29 October 2004), and benthic macroinvertebrate sampling sites from a previous study (Hynes 1998; solid blue circles). Red squares indicate locations of municipal sewer lift stations (10 of 12 shown) where stations 3, 8, and 9 are known to outflow into the lake at times of high rainfall.

Historical data source
Springfield Lake has an irregular bio-chemical monitoring history. To my knowledge, sampling has been conducted and documented in about 6 of the past 30 years, and of those only two years included more than 2 sampling days. Thus, the historical data used for water quality comparisons relied primarily on separate studies conducted in 1974 and 1977 (**Table 1**; Mandaville 2000). These two studies were of greater seasonal duration than my study and only those values collected during the period corresponding to the 2004 sampling period (01 June to 02 September) were used. In cases where depth-stratified sampling was conducted I used only the mean value across depth for the day of record. Historical data used in the temporal comparison of the benthic macroinvertebrates relied on the only known study that was conducted in 1997 (Hynes, 1998; **Table 1**).

Table 1. Summary of historical and contemporary water quality data from Springfield Lake, Halifax Regional Municipality, used for temporal comparisons. Sampling periods are: 20 June to 28 August, 1974 and 02 June to 23 August, 1977 (Mandaville 2000); 16 to 20 October 1997 (Hynes 1998); and 04 June to 02 September 2004 (this study). Shading indicates when measurements were collected for a given variable and the number (n) of measures recorded.

Variable	**1974**	**1977**	**1996**	**2004**
Air temperature (^{o}C)				13
Water temperature (^{o}C)		6		14
Dissolved oxygen (mg l^{-1})		6		14
Nitrate (mg l^{-1})	6	6		14
Ammonium (mg l^{-1})				13
Calcium (mg l^{-1})		6		13
Chloride (mg l^{-1})		6		
pH	10	6		14
Conductivity (µS cm^{-1})	6	6		13
Turbidity (NTU)	6	6		11
Per cent Transmittance				13
Absorbance				13
Secchi Depth (m)	6	6		14
Chlorophyll a (µg l^{-1})	6	6		10
Benthic macroinvertebrates			5	5

Additional Data Sources
Total precipitation and hourly air temperature data for the period 01 June to 31 August for 1974, 1977, and 2004, and hourly wind measurements for the 2004 study period were obtained from the Environment Canada National Climate Data and Information Archive (online data). The data were collected at the Halifax International Airport located in-land about 15 km E of the Lake and are assumed to provide a reasonable approximation of the meteorological conditions at the Lake. Data were modified such that 'trace' precipitation was recorded as missing values.

Water Sampling and Analyses
Water from Springfield Lake was tested once a week for 14 weeks over the period 04 June to 02 September 2004 (6 -8 day sampling intervals). Water was taken within a 3.5- hour window (9:30-13:00) at the same time each day. All sampling was concentrated in the same general location, with the exception of 04 June and 18 June (locations 1 and 3 in **Figure 1**). The sampling location was based on bottom depth (location bottom depth ranged from 4 to 4.7 m; **Table 2**) and was located using a

bathymetric map and a hand-held Garmin® Global Positioning System (GPS; **Figure 1**). While anchored at the station I recorded time, air temperature, Secchi depth, water depth, GPS position, and depth specific temperature and dissolved oxygen (DO) profiles. Secchi depth was recorded using shade-side-of-the-boat observations. The air temperature and water temperature, and dissolved oxygen measures were recorded using a YSI deck unit and a temperature and DO probe (Model 58). Depth profile data were collected at 1 m intervals (surface to 3 m) on the downcast and the upcast and I report the average of these two measurements. Water samples of ~0.4 l were collected, one each from a depth of 0.5 m and 3 m, with a Vernier® water sampler for subsequent assessment of chlorophyll-*a* content (μg l^{-1}). Chlorophyll-*a* water samples were stored immediately upon collection in separate opaque containers and transported to the lab on ice.

Table 2: Lake water quality sampling locations in Springfield Lake, Halifax Regional Municipality, for the sampling period 04 June to 02 September 2004.

	Date	Time (ADST)	Latitude	Longitude	Site Depth (m)
1	04 June	13:00	44.808	63.739	4.0
2	11 June	12:30	44.80976	63.74060	4.3
3	18 June	10:31	44.81091	63.74196	4.3
4	24 June	11:49	44.80905	63.74084	4.4
5	02 July	10:56	44.80964	63.74172	4.3
6	08 July	11:21	44.80816	63.74134	4.4
7	16 July	11:49	44.80887	63.74131	4.3
8	23 July	09:44	44.80843	63.74147	4.4
9	30 July	11:17	44.80804	63.74117	4.5
10	05 August	11:18	44.80788	63.74151	4.4
11	12 August	11:28	44.80886	63.80885	4.7
12	19 August	11:19	44.80877	63.74135	4.5
13	26 August	11:27	44.80782	63.74155	4.3
14	02 September	12:06	44.80804	63.74132	4.4

Prior to returning to shore approximately 9 l of water was collected among depths ranging from 0.5 to 4 m using the Vernier® water sampler. The 9 l depth-integrated water sample was analyzed using the Vernier® Logger Pro 3 data logging system and a suite of calibrated probes were used to measure pH, conductivity, turbidity, transmittance, absorbance, calcium, chloride, ammonium, and nitrate ions (**Table 1**). The pH sensor was calibrated using solutions of pH 4 and 7. The turbidity probe was calibrated against a distilled water blank (0 NTU) and a formalin solution with a known turbidity (100 NTU). The conductivity probe was calibrated with a solution of 1000 μS cm^{-1}. Transmittance and absorbance were measured using a LED light sensor at a wavelength of 470 nm. The results for transmittance and absorbance were compared and found to be significantly negatively correlated ($r > -0.99$, $p=0.0004$) therefore only the results for per cent transmittance are presented. The four ions were measured with separate ion-selective probes calibrated with high (1000 mg l^{-1} for calcium and chloride, 100 mg l^{-1} for ammonium and nitrate) and low (100 mg l^{-1} for calcium and chloride and 1 mg l^{-1} for ammonium and nitrate) solutions.

Of the water collected for chlorophyll-*a* analysis, 30 ml was filtered through glass microfibre filters (25 mm diameter) using a vacuum pump. The filters were stored separately in 10 ml of 90% acetone in 20 ml-scintillation vials. After storage in a freezer for at least 24 hours the samples were processed in the

Dalhousie Oceanography Department Analytical Lab. The samples were analyzed using both an acidification technique and the Welshmeyer technique on a Turner® fluorometer (Ryan, C., Oceanography Dept., Dalhousie University, Halifax N.S., pers. comm.). The acidification technique uses broadband excitation-emission filters that can underestimate chlorophyll-*a* in the presence of chlorophyll-*b;* the Welshmeyer technique uses narrowband filters that reduce the potential of chlorophyll-*b* masking chlorophyll-*a* (Welshmeyer 1994). The results from the Welshmeyer technique were used for this study.

On 08 July a water sample was taken from the 9 l sample for an independent analysis at the Queen Elizabeth II (QEII) Environmental Services Lab, Halifax, Nova Scotia. The results of these QEII analyses were compared to the field results from that day as a means of nominal quality control (**Table 3**). These results generally agreed with the field results with the exception of nitrate, chloride, and conductivity. The field nitrate measurement on that day was 0.42 mg l^{-1}, and the QEII measurement for nitrate/nitrite was 0.05 mg l^{-1}; this discrepancy is discussed later. The field chloride results on 08 July were 0.10 mg l^{-1} whereas the laboratory recorded levels of 22 mg l^{-1}. Throughout the summer the chloride levels I measured were consistently less than 2 mg l^{-1}. It was determined based on the results from the QEII Laboratory, that contemporary chloride data were unsuitable for analysis. It is likely that all values obtained are underestimates of the actual chloride levels. The differences in conductivity measurements (115 µS cm^{-1} for QEII and 390 µS cm^{-1} for my observation) could be attributed to the range setting on the probe. The majority of the samples in my study were collected by inadvertently using the 0-200 µS cm^{-1} range. The historical data, as well as the QEII Lab sample, were analyzed using a range of 0-2000 µS cm^{-1}. Thus, the data I collected using the 0-200 µS cm^{-1} range were corrected after the fact using a range inter-calibration based on nine solutions of varying conductivity (3.31 to 246.6 µS cm^{-1} in the 0-2000 µS cm^{-1} range and 32.3 to 898.1 µS cm^{-1} in the 0-200 µS cm^{-1} range) measured once in each of the ranges. These data, along with estimates collected in the field using both ranges were used to provide a linear regression post-calibration correction (r^2 =0.99, $p<0.001$; **Figure 2**):

$$C_{0\text{-}2000} = -3.05 + 0.28\,(C_{0\text{-}200})$$

Where $C_{0\text{-}2000}$ is the conductivity in the 0-2000 µS cm^{-1} range and $C_{0\text{-}200}$ is the conductivity in the 0-200 µS cm^{-1} range.

Table 3. Summary of comparisons of concentration estimates for six water quality variables determined by the Queen Elizabeth II Environmental Services Laboratory (QEII), Halifax, Nova Scotia, and field-determined (Field) estimates where estimates are based on the same water sample collected from Springfield Lake, Halifax Regional Municipality, on 08 July 2004.

Variable	**QEII**	**Field**
Calcium (mg l^{-1})	5.8	5.5
Chloride (mg l^{-1})	22	0.10
pH	6.6	6.6
Conductivity (µS cm^{-1})	115	390
Nitrate + nitrite (mg l^{-1})	0.05	0.42 (Nitrate)
Total Nitrogen	0.21	

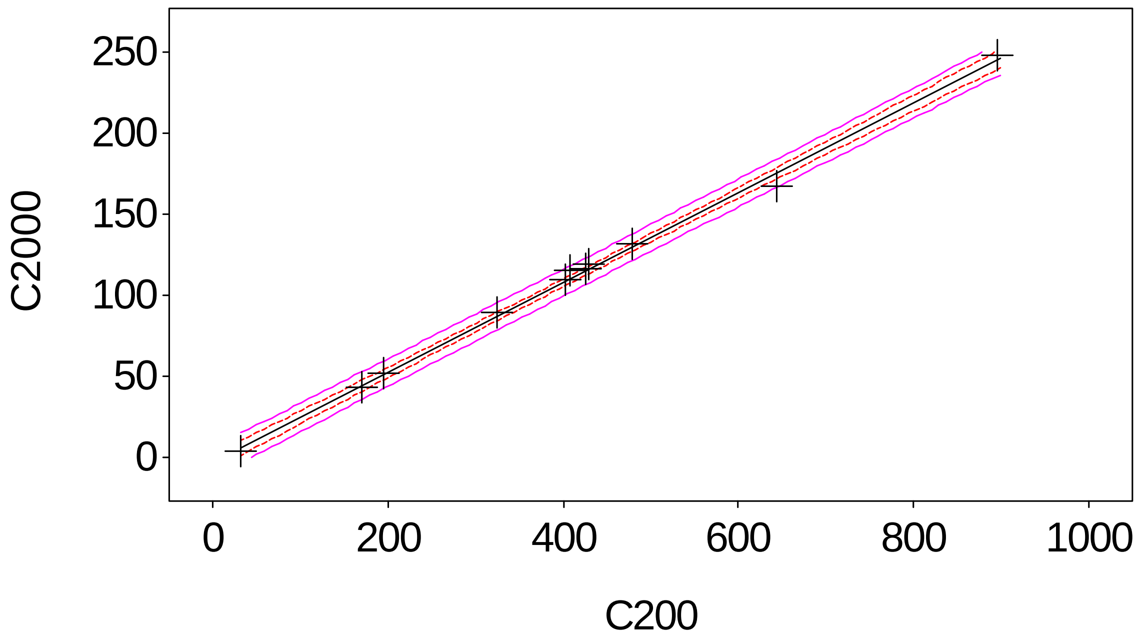


Figure 2. Linear regression post-calibration correction (r^2 =0.99 , $p<0.001$) used for correction of conductivity measures taken in Springfield Lake, Halifax Regional Municipality. Nine solutions were measured with two calibration ranges (C2000: 0-2000 µS cm^{-1} and C200: 0-200 µS cm^{-1}). The inner red lines indicate the 95% confidence interval for the fitted (black) calibration correction equation and the outer purple lines indicate the 95% predicted interval.

Benthic Macroinvertebrate Sampling and Analysis

Benthic macroinvertebrates were collected among five sites over the period 27 and 29 October 2004 (**Table 4**). Two samples were collected at each of the five sites (**Figure 1**) using a 15 l Ekman dredge deployed from an anchored aluminium motor boat. Site selection was based on an earlier study by Hynes (1998; **Figure 1**) though I had to select for deeper water (greater than 2 m) and soft, mud bottoms that could be sampled using the Ekman dredge.

At each benthic collection site air temperature, bottom depth, and GPS position were recorded. Dissolved oxygen and water temperature profile data were collected using methods described above. As the dredge was lifted from the water surface to the boat, any water draining from the dredge was collected. The contents of the dredge were deposited in a 12 l bucket or 17 l plastic container labelled with the site number. The inside of the dredge was rinsed into the container holding the sample. Excess water from the sample was screen filtered (420 µm) and material collected was returned to the sample in the container. Two such samples were collected at each site. Sample volume was estimated by measuring the height of the sediment in the container and calculating the volume based on the dimensions of the containers. The samples were filtered on land using a series of sieves to a terminal mesh size of 420 µm. Everything retained by the sieves was kept for examination with the exception of rocks and vegetation that were rinsed into the sample and discarded. Filtered samples were placed in bags according to site and sample and placed in a cooler for transport to the lab where they were held frozen until sorted under a dissecting microscope where all invertebrates were identified to Family and enumerated.

Table 4: Locations of 2004 benthic macroinvertebrate sampling sites in Springfield Lake, Halifax Regional Municipality.

Site	Date	Time (ADST)	Latitude	Longitude	Site Depth (m)	Distance from 1997 site (m)	Sample Volume (l) A	B
1	27 October	16:26	44.81225	-63.73492	4.0	60	-	-
2	29 October	10:10	44.81069	-63.73572	2.0	170	2	2
3	29 October	13:33	44.81523	-63.74017	4.0	30	2	1
4	29 October	11:33	44.81778	-63.73571	2.0	54	3	3
5	29 October	14:18	44.80810	-63.74332	3.4	120	2	4

Trophic State, Diversity and Biological Indices

The Trophic State Index (*TSI*; Carlson 1977) estimates were calculated using Secchi disk transparency as follows:

$$TSI\ (SCD) = 60 - 14.41\ [\log_e (SCD)]$$

where *SCD* is the Secchi depth in meters and using Chlorophyll-*a* values:

$$TSI\ (Chl\text{-}a) = 9.81[\log_e (Chl\text{-}a)] + 30.6$$

where *Chl-a* is the chlorophyll-*a* concentration in $\mu g\ l^{-1}$ (**Appendix Table 1**).

The macroinvertebrate community was assessed using four diversity indices and four biological indices. The first measure of diversity was the number of taxa per sample (*S*). The second, Simpson's Index of Diversity (*D*), the probability that two individuals randomly selected from a sample will belong to the same species, was calculated using:

$$D = 1 - \sum_{i=1}^{q} (p_i)^2$$

where p_i is the proportion of individuals in the i^{th} taxon (Family) of the sample, and q is the total number of taxa in the sample. The third, Shannon-Weiner Diversity Index (*H*) was calculated using:

$$\overset{q}{\underset{i=1}{H}} = - \sum p_i \log_2 p_i$$

Where p_i is the proportion of individuals in the i^{th} taxon of the sample, and q is the total number of taxa in the sample. The final diversity index, evenness (*J*), was calculated as:

$$J = \frac{H}{\log_{10} q}$$

where H is the Shannon-Weiner Diversity Index value and $\log_{10} q$ is the maximum H, the value of H when all species are equally abundant (Hill 1973), with q being the number of taxon in the sample.

The first biological index used was the Family Biotic Index:

$$FBI = \sum \frac{x_i t_i}{n}$$

where x_i is the number of individuals in the i^{th} taxon of the sample, t_i is the tolerance value of the i^{th} taxon (**Appendix Table 2**), and n is the total number of organisms in the sample. The second index, the Belgian Biotic Index (*BBI*) was determined using a standard table (**Appendix Table 3**) with water quality being interpreted from the biotic index values (De Pauw and Vanhooren 1983). The third, the Biological Monitoring Working Party (*BMWP*) index was calculated by adding the taxon-specific pollution tolerance scores (**Appendix, Table 2**) of all the taxa present in a sample (Kirsh 1999). The final biotic index, the average score per taxon (*ASTP*) was calculated by dividing the *BMWP* value for a sample by the number of Families in that sample (Kirsh 1999).

Statistical analysis

Where a datum deviated from the mean by more than 2 standard deviations it was considered a statistical outlier and was omitted from analyses. Comparisons of change in water quality variables between or among years were made using a one-way Analysis of Variance test with a rejection level of $\alpha = 0.05$. Pearson's coefficient of correlation was used to describe linear associations between variables, with significance assessed at $P < 0.05$. Diversity indices were compared between years using one-way Analysis of Variance as above, with the sites aggregated for a sample of $n = 5$ for both years. All statistical analyses were performed using Statistix 8 software (Analytical Software 2003).

Results

Dissolved Oxygen and Temperature

Dissolved oxygen (DO) saturation (%) during the study period of 2004 was different among depth strata (**Figure 3a**). Surface water was well-saturated or supersaturated (90 –127%) throughout the summer and was greater than at any other depth ($p < 0.001$). DO saturation in the 1-3 m depth range decreased throughout the summer. The mean DO saturation at 1 m was 64±33%; saturation was high for June (88-105%), decreased throughout the summer with some fluctuation, and reached 86% on 02 September. The mean DO saturation at 2 m was 41±30%. Saturation at 2 m was high (83-104%) in June, declined throughout the rest of the summer, and increased at the end of August, reaching 42% at the beginning of September. The mean saturation at 3 m was 37±30 %. Saturation was high (93-101%) for early June, then saturation declined throughout the summer to as low as 13% and increased again at the end of August to reach 35 % on 02 September.

In contrast to 2004, the water column in 1977 was well-saturated in DO (≥85% at all depths) over the entire summer, though it decreased slightly throughout the summer at all depths (**Figure 4a**). Mean surface saturation was 95±2 %. Maximum saturation occurred at the beginning of summer, on 02 June (97%) and minimum surface saturation (92%) occurred at the end of summer, on 23 August. Saturation at the surface was greater than at other depths, though not significant ($p = 0.06$) in contrast to the 2004 measures. Mean DO saturation at 1 m in 1977 was 94±2 %. Saturation at 1 m was relatively stable throughout the summer (92-95%). The summer mean DO saturation was 94±2 % at 2 m and 93±3 % at 3 m; 92±3 % at 4 m and 90±4% at 5 m.

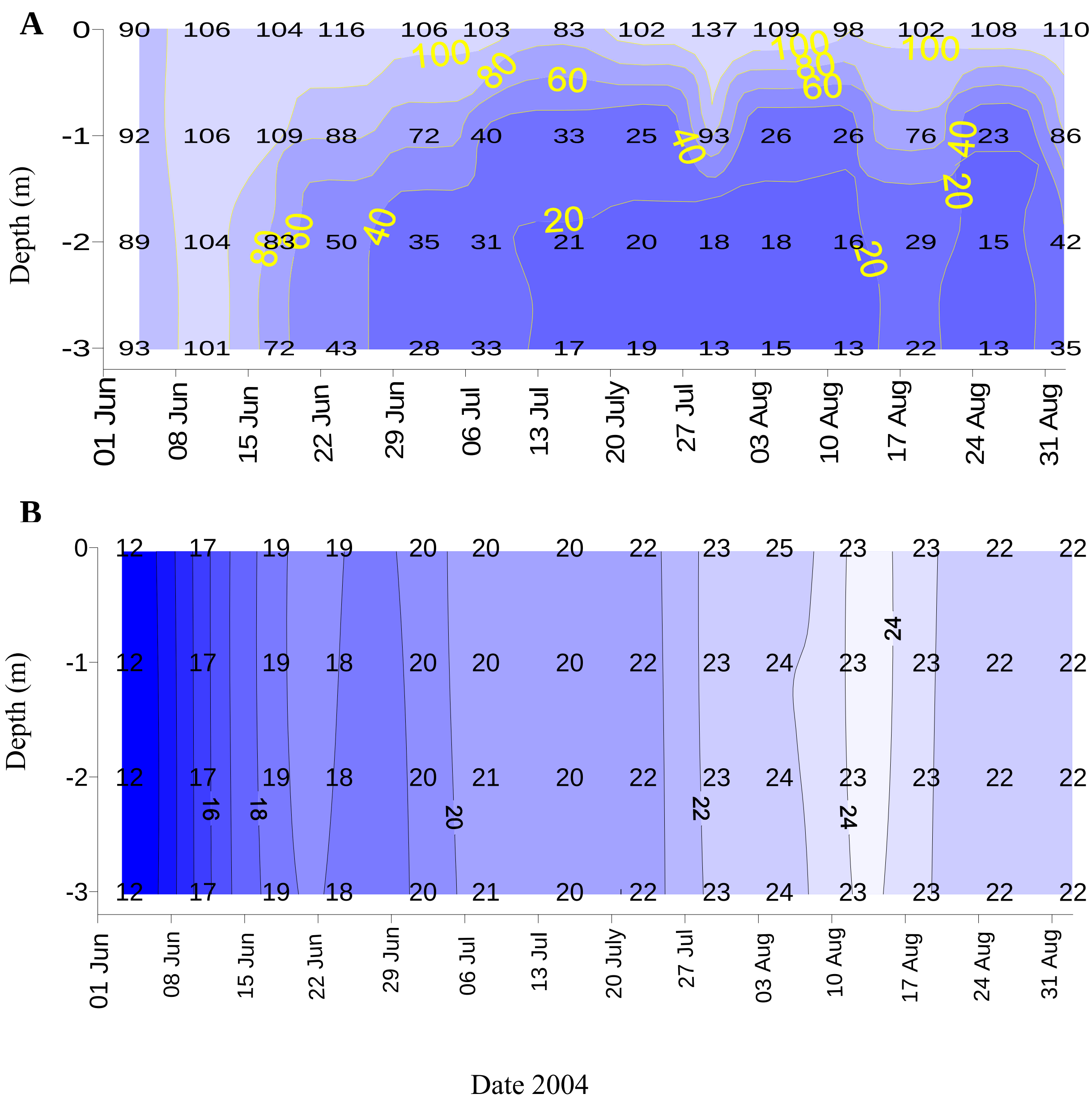


Figure 3. Per cent oxygen saturation isopleths (A; % in yellow) and isotherms (B; °C in black) for Springfield Lake, Halifax Regional Municipality, for the period 04 June to 02 September 2004. Depth-specific observed values (surface to 3 m) are noted for each sampling day.

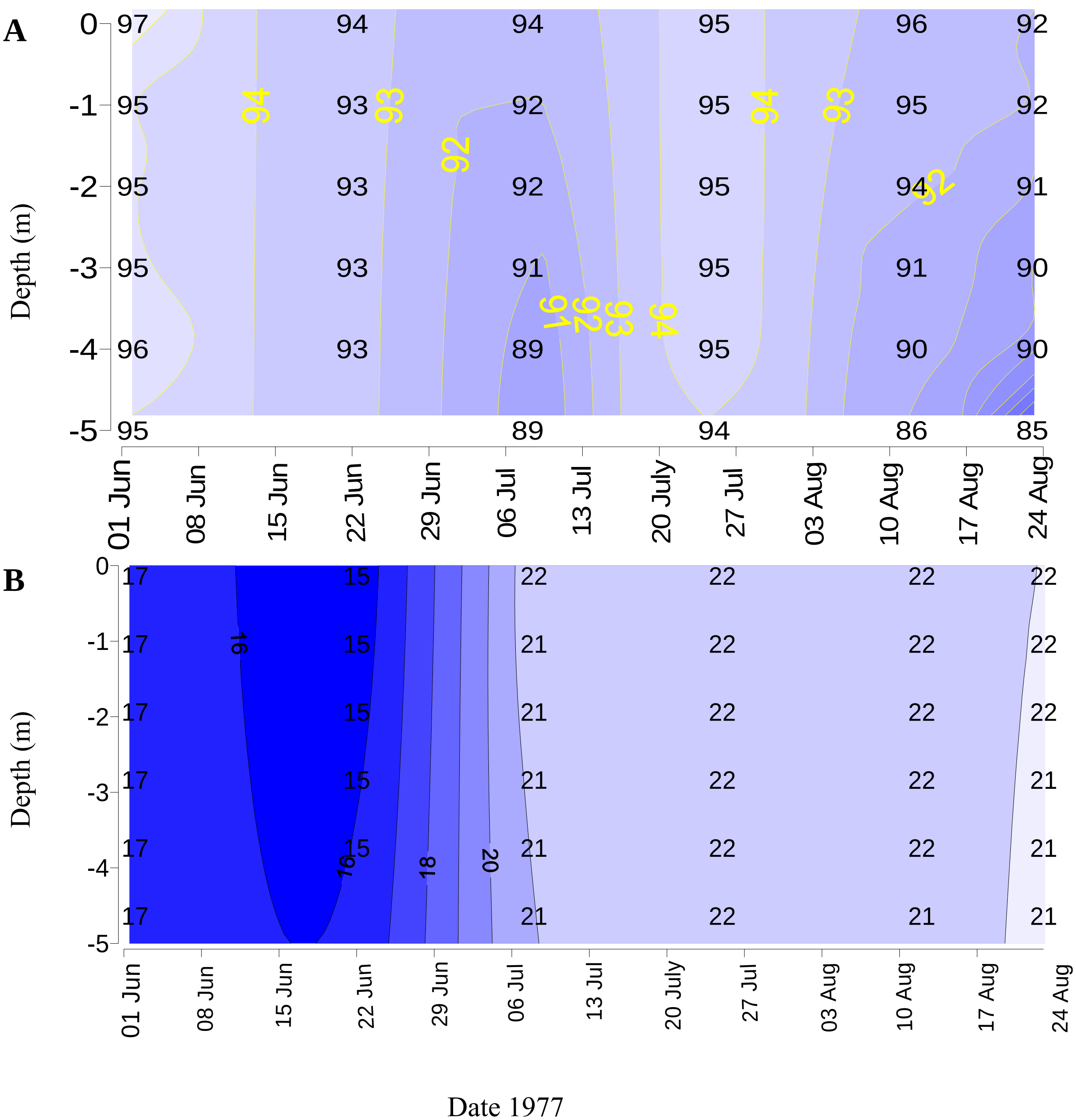


Figure 4. Per cent oxygen saturation isopleths (A; % in yellow) and isotherms (B; °C in black) for Springfield Lake, Halifax Regional Municipality, for the period 02 June to 23 September 1977. Depth-specific observed values (surface to 5 m) are noted for each sampling day.

Dissolved oxygen (mg l^{-1}) at the surface was not different between 1977 and 2004 ($p=0.18$) although per cent DO saturation was significantly higher in the surface layer in 2004 ($p=0.03$). Both DO concentration and per cent saturation (**Figure 5**) were significantly lower in 2004 than in 1977 at each of the 1, 2, and 3 m depths (**Table 5**)

The 2004 mean water temperature (all depths measured) was 20.5±3 °C and heat was well mixed in the water column as there was no difference in temperature with respect to depth ($p=0.99$; **Figure 3b**). The depth-averaged water temperature increased throughout the summer, reached a maximum of 24.4 °C on 05 August and decreased slightly to 22.1 °C on 02 September.

As shown for 2004, heat in 1977 was well mixed throughout the water column with no difference in temperature among depths ($p = 0.97$; **Figure 4b**) with a mean depth-averaged water temperature of 20±3 °C. In 1977 the depth-averaged water temperature increased thorough June and into July before decreasing and becoming stable throughout August at around 21 °C. Thus, neither heat mixing depth nor water temperature were different between 1977 and 2004 ($p = 0.30$).

As seen for Lake water temperature, there was no difference ($p=0.16$) in mean air temperature over the course of the summer among 2004 (17±4 °C), 1974 (17±3 °C), and 1977 (17±4°C; **Figure 6**). For all years, air temperature increased, with fluctuations, throughout most of the summer and was stable through to the end of August.

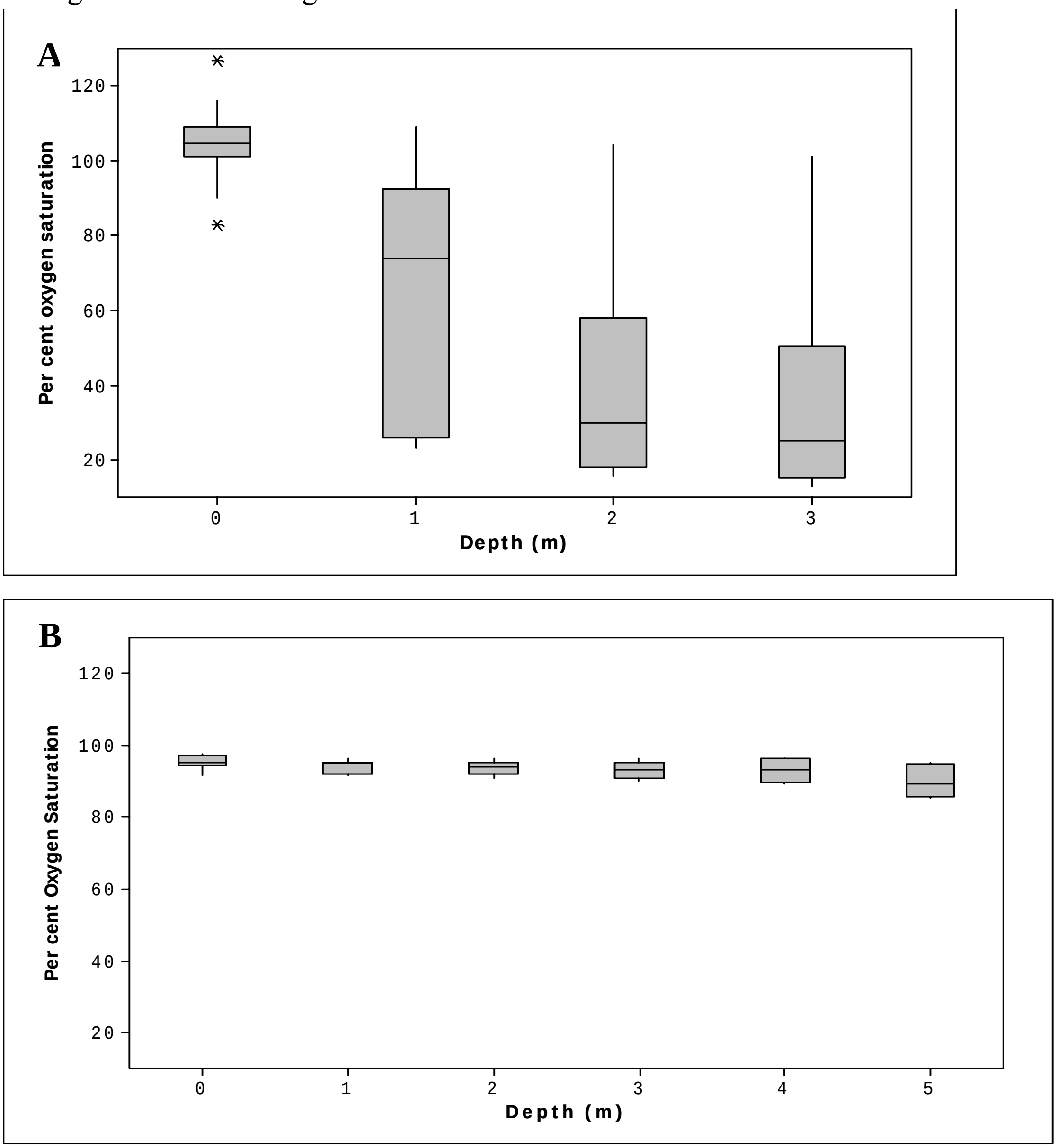


Figure 5. Box and whisker representations (data range, mean, first and third quartiles, and outliers) of per cent oxygen saturation at depth over the sampling period (A) 02 June to 04 September 2004 (n=14), and (B) 02 June to 23 August 1977 (n= 6) in Springfield Lake, Halifax Regional Municipality.

Table 5: ANOVA results (p-values) from the comparison of summer mean dissolved oxygen (DO; mg l^{-1}) and per cent dissolved oxygen saturation at depth in Springfield Lake, Halifax Regional Municipality, between the years 1977 and 2004.

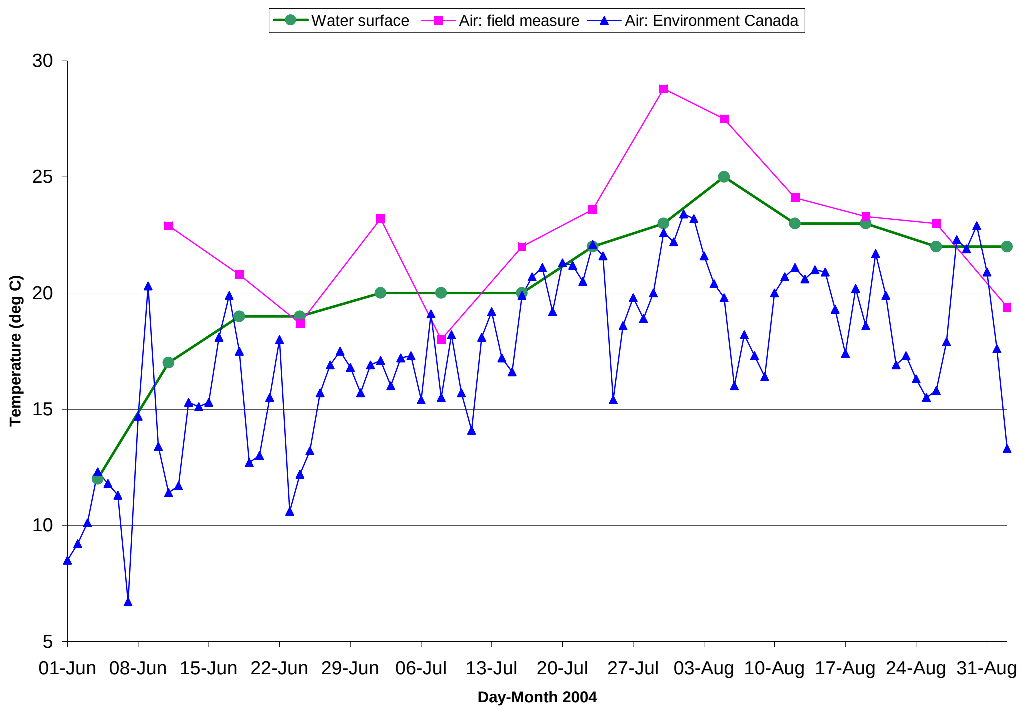
Depth (m)	**P- values for dissolved oxygen (mg l^{-1})**	**P-values for per cent DO saturation (%)**
Surface	0.18	0.03
1	0.03	0.03
2	<0.01	<0.01
3	<0.01	<0.01

Correspondingly, there was no difference (p=0.16) in mean daily rainfall (mm) among the sampling periods 1974 (3.1±8 mm), 1977 (5.8±11 mm) and 2004 (3.6±8 mm) though mid-July in 1974 was much drier than the same period in 1977 and 2004 (**Figure 7**). Correlation analyses between the total daily rainfall (sampling day plus two days prior) and the water quality variables in this study yielded no significant results.

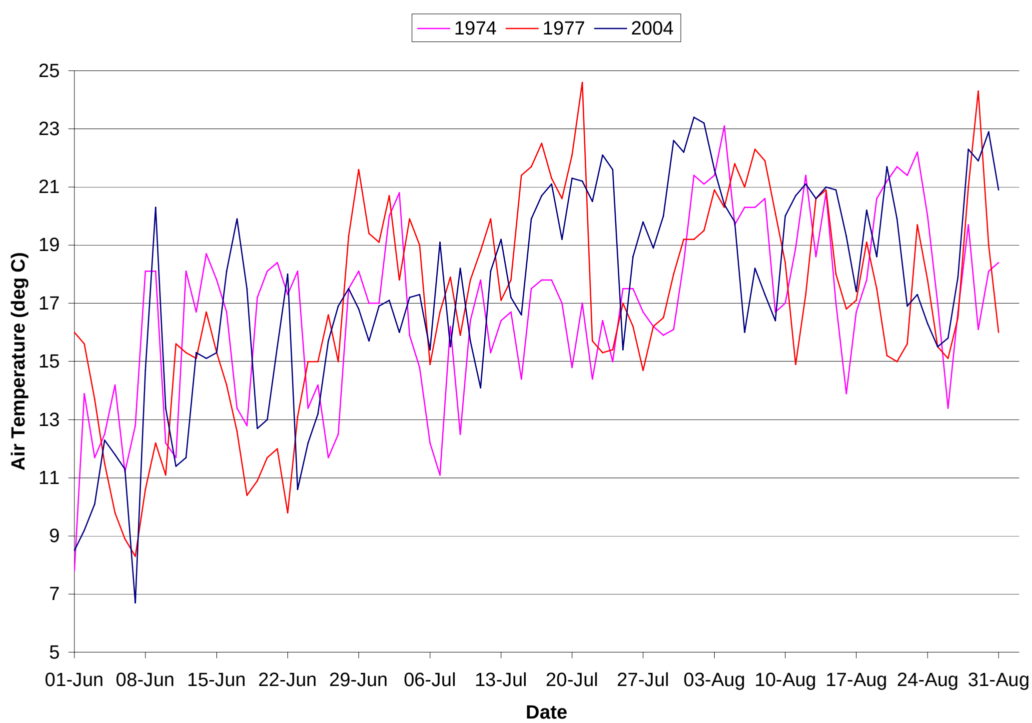
Wind speed data were examined for summer in 2004 to assess potential explanations for the observed deeper DO mixing in the water column. I used the square of wind speed because it approximates wind stress (**Figure 8**). Deep DO mixing was observed on 30 July and 19 August. The mixing observed on 19 August occurred at a time of high wind stress (>500 km^2 h^{-2}), however the mixing on 30 July occurred prior to a wind event, during low wind stress (<200 km^2 h^{-2}). The stratification on 05 August occurred during low wind stress (<200 km^2 h^{-2}) but the stratification on 12 August occurred despite high wind stress (>400 km^2 h^{-2}). Thus, the relationship between wind stress and DO mixing in the Lake is uncertain.

Biological Properties

The mean Secchi depth over the course of the sampling period was not different (p=0.46) among 1974 (3.7±0.8 m.), 1977 (3.3±0.5 m), and 2004 (3.4±0.5 m.). In 1974 and 1977 Secchi depth reached a maximum in late-June and decreased throughout the summer while Secchi disk depth in 2004 deviated from this pattern, reaching a maximum in mid-July, with an overall fluctuating decrease, throughout the rest of the summer (**Figure 9a**). It is possible that Secchi depth in 2004 was influenced by precipitation, as increases in Secchi depth occurred on 16 July and 12 August shortly after periods of rain on 15 July and 9 August (>15 mm; **Figure 7c**), though this relationship is not significant (p=0.5).



A



B

Figure 6: (A) Springfield Lake surface water temperature (circle), local air temperature (square), and Environment Canada air temperature (triangles) estimates for the period 01 June to 31 August 2004; and (B) Environment Canada air temperature mean daily air temperature estimates for the sampling periods 01 June to 30 August 1974, 1977, and 2004 based on Environment Canada (2005) data for Halifax International Airport.

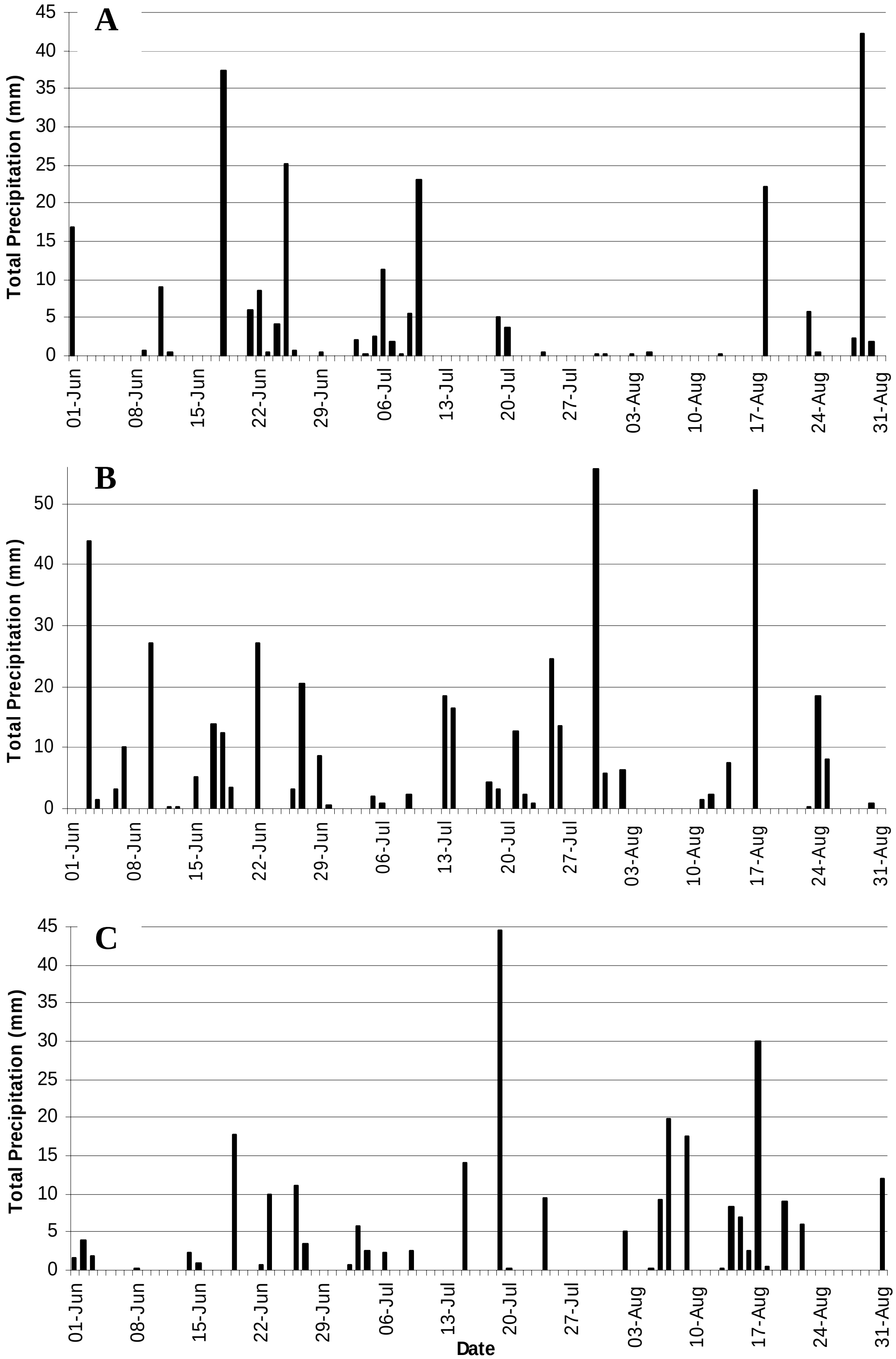


Figure 7: Daily total precipitation (mm) during study periods 01 June to 31 August for: (A) 1974; (B) 1977; and (C) 2004 based on Environment Canada (2005) data for Halifax International Airport.

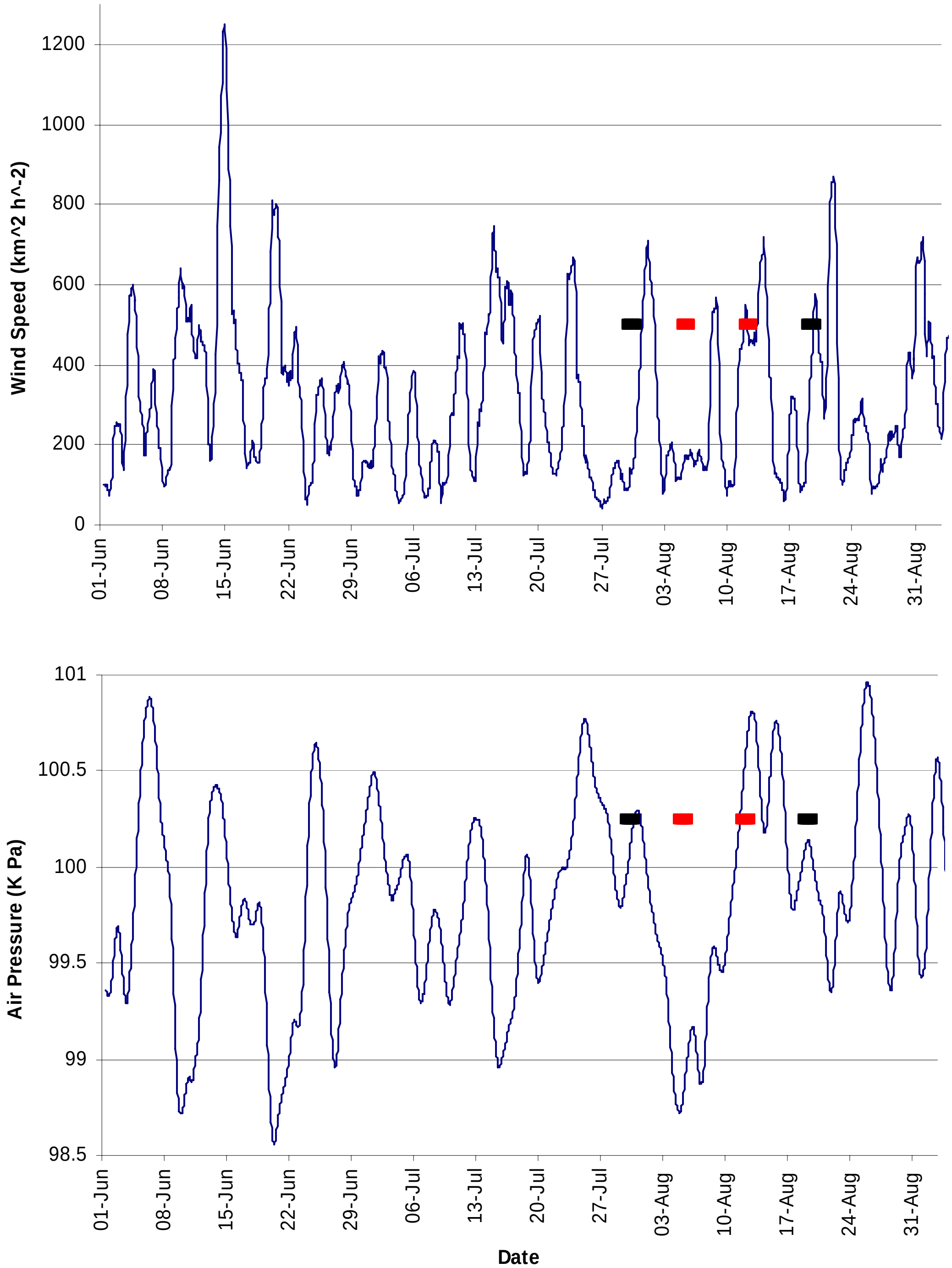


Figure 8. Hourly wind speed squared (km^2 h^{-2}) and air pressure (kPa) for the study period 01 June to 31 August 2004, smoothed using a 25-hour (centred) moving average. Bars span the 24-hour periods corresponding to dissolved oxygen mixing events (see **Fig. 3a**) on 30 July and 19 August (black) and stratification on 05 and 12 August 2004 (red). Wind data are based on Environment Canada (2005) data for Halifax International Airport.

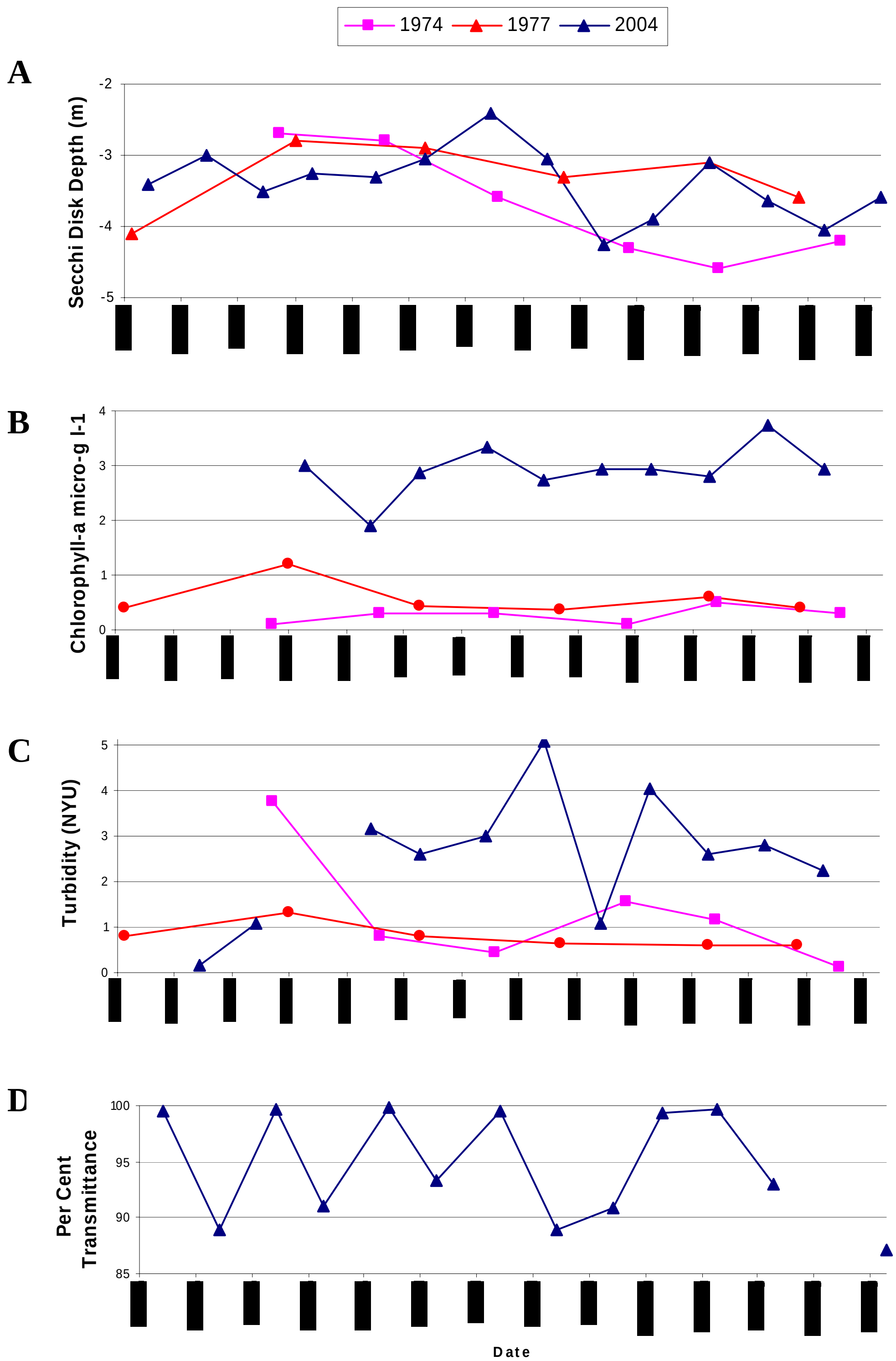


Figure 9. A) Secchi disk depth (m); B), chlorophyll-*a* (µg l^{-1}); C) turbidity (NTU); and D) per cent transmittance for Springfield Lake, Halifax Regional Municipality, for the sampling periods 20 June to 28 August 1974 (square), 02 June to 23 August 1977 (circle), and 04 June to 02 September 2004 (triangle).

The average surface chlorophyll-*a* levels in 2004 were 0.6 µg l^{-1} lower than the levels at 3 m, with largest disparity on 23 July when the surface chlorophyll-*a* was 2.75 µg l^{-1} and at 3 m was 5.6 µg l^{-1}. However, as these measures were not statistically different ($p = 0.09$) I aggregated the two as a single average chlorophyll-*a* measurement.

The summer mean concentrations of chlorophyll-*a* observed in 1974 (0.3 ± 0.2 µg l^{-1}) and 1977 (0.6 ± 0.3µg l^{-1}) were significantly lower ($p<0.001$) than that observed in 2004 (3.2 ± 0.6 µg l^{-1}; **Figure 9b**). Chlorophyll-*a* in 1974 was stable throughout the summer, and in 1977 chlorophyll-*a* reached a maximum of 1.3 µg l^{-1} early in the summer on 22 June. After this the concentration decreased and remained stable (0.4 to 0.6 µg l^{-1}) for the remainder of the summer (**Figure 9b**). Chlorophyll-*a* concentrations in 1974 were not different from those of 1977 ($p=0.07$).

The mean turbidity in 1974 (1.3 ± 1.7 NTU) and 1975 (0.78 ± 0.2 NTU) though lower, were not different ($p=0.05$) from that observed in 2004 (2.6 ± 1.3 NTU). Turbidity in 1974 was high in June (3.8 NTU), then decreased, peaking again at 1.6 NTU on 02 August before decreasing throughout August (**Figure 9c**). The mean turbidity in 1977 was more stable in 1977, and in 2004 there was more variation. There may be a rain influence on the turbidity data, with increases in turbidity following periods of rain but there was no significant relation ($p=0.63$).

The summer 2004 mean transmittance was 95 ± 5% and there does not appear to be any decreasing or increasing trend in the data (**Figure 9d**). There was no significant correlation between transmittance and either Secchi depth ($p=0.5$) or chlorophyll-*a* ($p=0.16$).

Nutrient Properties

The mean nitrate concentration over the course of the sampling period in 2004 was 0.62 ± 0.2 mg l^{-1}, and it was relatively stable during most of June and July (0.4 to 0.61 mg l^{-1}; **Figure 10a**), and reached a maximum of 0.97 mg l^{-1} on 12 August and subsequently fluctuated by ±0.1 mg l^{-1} into September. There are indications that nitrate concentrations increased after high rainfall, e.g., the maximum observed on 12 August (0.98 mg l^{-1}) followed a period when over 45mm of rain fell between 06 and 09 August (see **Figure 7c**).

The mean nitrate/nitrite concentrations in 1974 (0.02 ± 0.01 mg l^{-1}) and 1977 (0.006 ± 0.003 mg $^{-1}$) were two orders of magnitude lower than observed in 2004 where the mean nitrate was 0.62 ± 0.2 mg $^{-1}$ (**Figure 10a**). Such a large change may suggest a difference in the nature of the measurement in the 1970s relative to those in 2004.

The methods used to determine nitrate/nitrite in the 1970s are not known. Nitrate/nitrite is a different variable than nitrate, so direct comparisons of the two are not prudent. The nitrate concentrations in the 1970s are not known, however there is one nitrate/nitrite measure from 08 July 2004, analyzed by the QEII lab (**Table 3**). This value of 0.05 mg l^{-1} is comparable to the nitrate/nitrite range observed in 1974 (0.01 to 0.03 mg l^{-1}) and greater than that observed in 1977 (0.002 to 0.011 mg l^{-1}).

Regardless of the concern with nitrate, ammonium was positively correlated with nitrate in 2004 ($r=0.60$, $p=0.03$). The mean ammonium concentration over the 2004 sampling period was 0.26 ± 0.04 mg l^{-1} and concentrations fluctuated between 0.17 and 0.32 mg l^{-1}, and showed a general increasing trend as seen in nitrate.

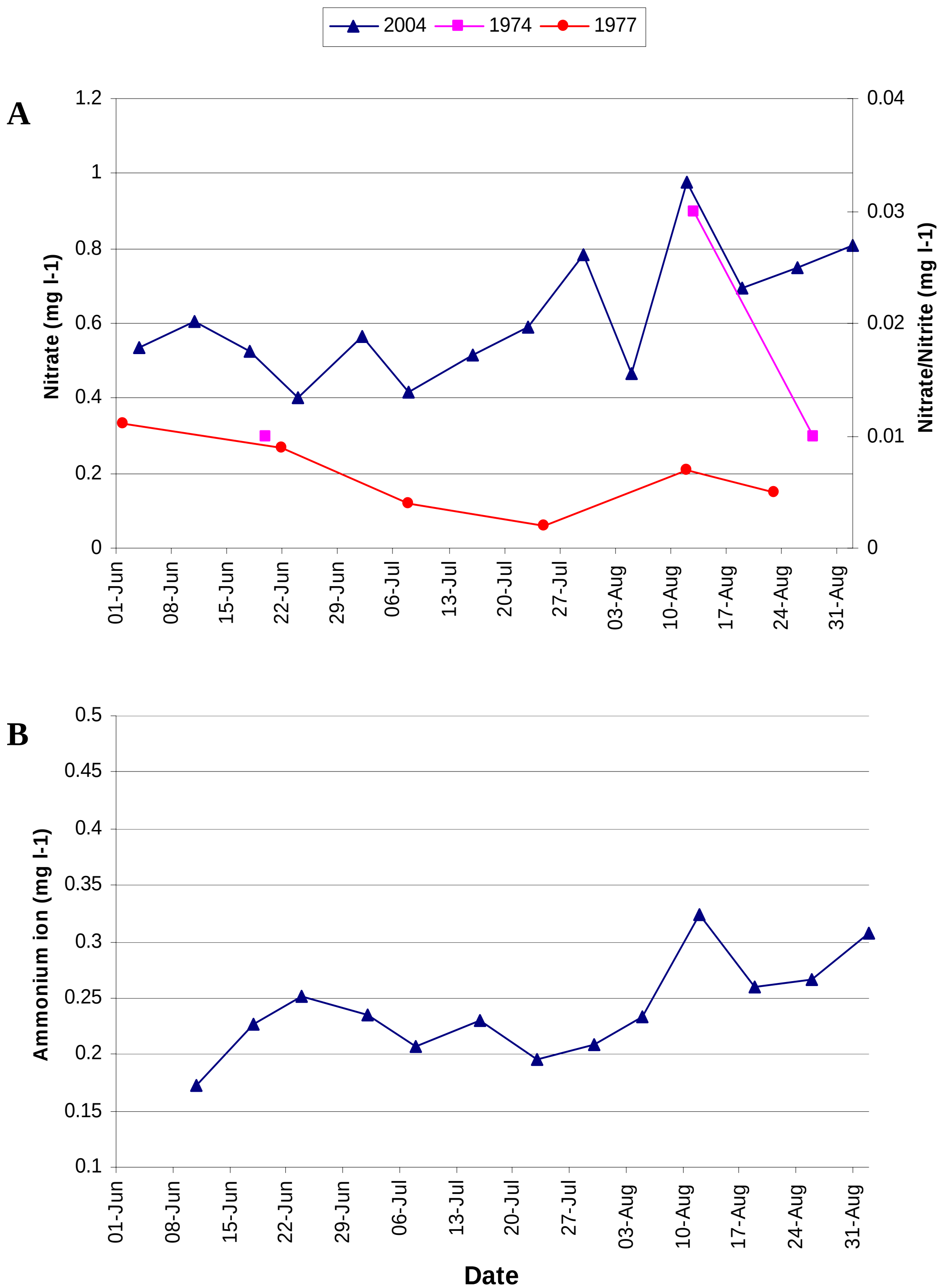


Figure 10. A) Nitrate (2004) and nitrate/nitrite (1974 and 1977; mg l^{-1}) and B) ammonium (mg l^{-1}) concentration for Springfield Lake, Halifax Regional Municipality, for the sampling periods 20 June to 28 August 1974 (square), 02 June to 23 August 1977 (circle), and 04 June to 02 September 2004 (triangle).

On 08 July 2004 a sample processed by the QEII Environmental Services lab measured the total phosphorous in the lake to be 18 µg l^{-1}. The mean total phosphorous concentration was 8.3 ± 3 µg l^{-1} in 1974 and 9.5 ± 2 µg l^{-1} in 1977. Thus there is evidence of an increasing trend in total phosphorous, though the increase from 1974 to 1977 is not significant (p=0.3).

Other Dissolved Ion Properties

Conductivity in 1974 (45±4 µS cm^{-1}) was lower (p<0.001) than conductivity in 1977 (72±2 µS cm^{-1}) and conductivity in 1977 was lower (p<0.001) than in 2004 (119±25 µS cm^{-1}). Thus, an annual increase of about 2.5 µS cm^{-1} is apparent in the Lake. The mean conductivity in 1974 showed a weak increasing trend over the summer (**Figure 11a**). The mean conductivity in 1977 was not quite double that of 1974, though no increasing trend was apparent (**Figure 11a**). Conductivity in 2004 fluctuated between 105 and 141 µS cm^{-1} with an overall increasing trend through the summer (**Figure 11a**). The conductivity series for 2004 has maxima corresponding to precipitation, including the increase on 12 August that was also observed in nitrate and ammonium ions. The maximum conductivity observed was on 02 September (141 µS cm^{-1}), two days after a rain event on 31 August (**Figure 7c**). These results suggest that some of the fluctuations observed in variables such as nitrate, ammonium, and turbidity may be related to inflow of watershed runoff following rain. Conductivity in 2004 was positively correlated with calcium (r=0.78, p=0.42), ammonium (r=0.86, p=0.33), and nitrate (r=0.84, p=0.36) ions, further suggesting that runoff may be a factor, though this relationship is not apparent from a statistical perspective.

Calcium concentrations in 1977 (3.6 ± 0.13 mg l^{-1}) were significantly lower (p=0.02) than in 2004 (9.1 ± 4 mg l^{-1}). In 1977 calcium had a narrow range of 3.5 to 3.9 mg l^{-1} and showed little change over the summer. This is different from 2004 when calcium decreased by 3 mg l^{-1} per week until 08 July when the concentration began an increase through July and decreased toward 4 mg l^{-1} by 26 August (**Figure 11b**). The highest calcium levels were observed at the beginning of the summer and after mid August concentrations were never more than 10 mg l^{-1}. Calcium in 2004 showed similar increases after rain events that were observed for conductivity and other variables (see above).

The pH in 1974 (6.5±0.5) was less acidic (p < 0.001) than in 1977 (5.6±0.2), but was not different (p 0.88) from 2004 (6.5±0.2); the mean pH in 1977 (p<0.001) was more acidic than in 2004 (**Figure 11c**). The pH in 1974 showed more fluctuation than in 2004, while pH in 1977 was relatively stable through to late July before beginning to decay similar to that observed in 2004. The pH in 2004 increased overall throughout June (from 5.7 to 6.8), then began a steady decay in August reaching 6.0 by the end of the sampling period (**Figure 11c**).

Trophic State

The *TSI* (Carlson 1977) was calculated using the mean chlorophyll-*a* and Secchi disk depth values for each sampling day in 1974, 1977 and 2004 (**Figure 12**). The 2004 (41.9±1.8) chlorophyll-*a* *TSI* values were significantly greater (p=0.003) than those from 1974 (16.0±6.5) and 1977 (23.3±6.6). There is an increasing trend in these *TSI* values, both in the actual number and the interpretation of 1974 and 1977 reflecting an oligotrophic state and 2004 a mesotrophic state (**Figure 12a**). The Secchi depth *TSI* values were not different (p=0.47) among 1974 (41.4±3.3), 1977 (42.9±2.1) and 2004 (42.5±2.1) and there is also no difference in the interpretation of the values, with the Lake being classified as mesotrophic for all three years studied (**Figure 12b**).

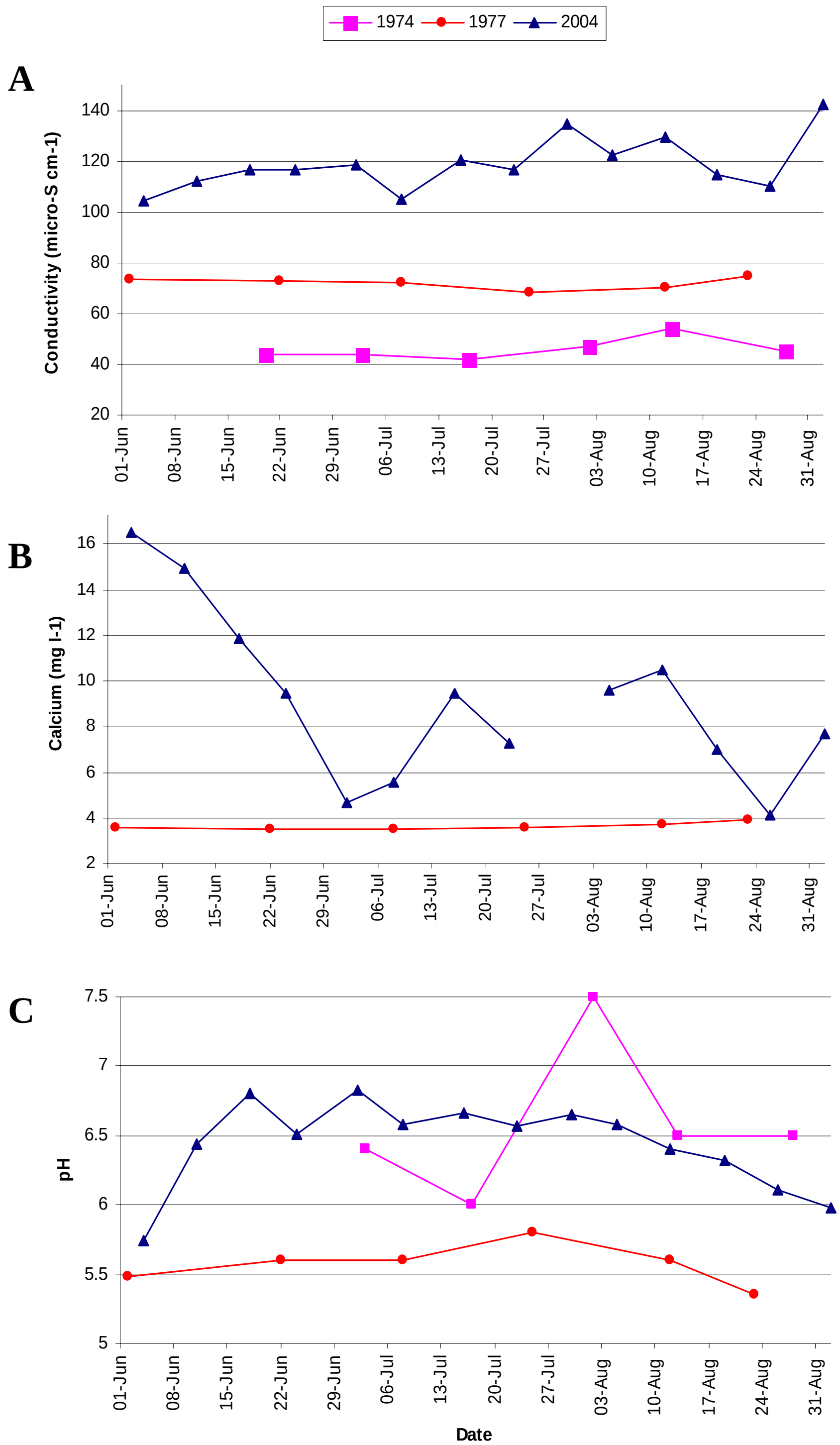


Figure 11. A) Conductivity (µS cm^{-1}); B), calcium (mg L^{-1}); and C) pH estimates for Springfield Lake, Halifax Regional Municipality, for the sampling periods 20 June to 28 August 1974 (square), 02 June to 23 August 1977 (circle), and 04 June to 02 September 2004 (triangle).

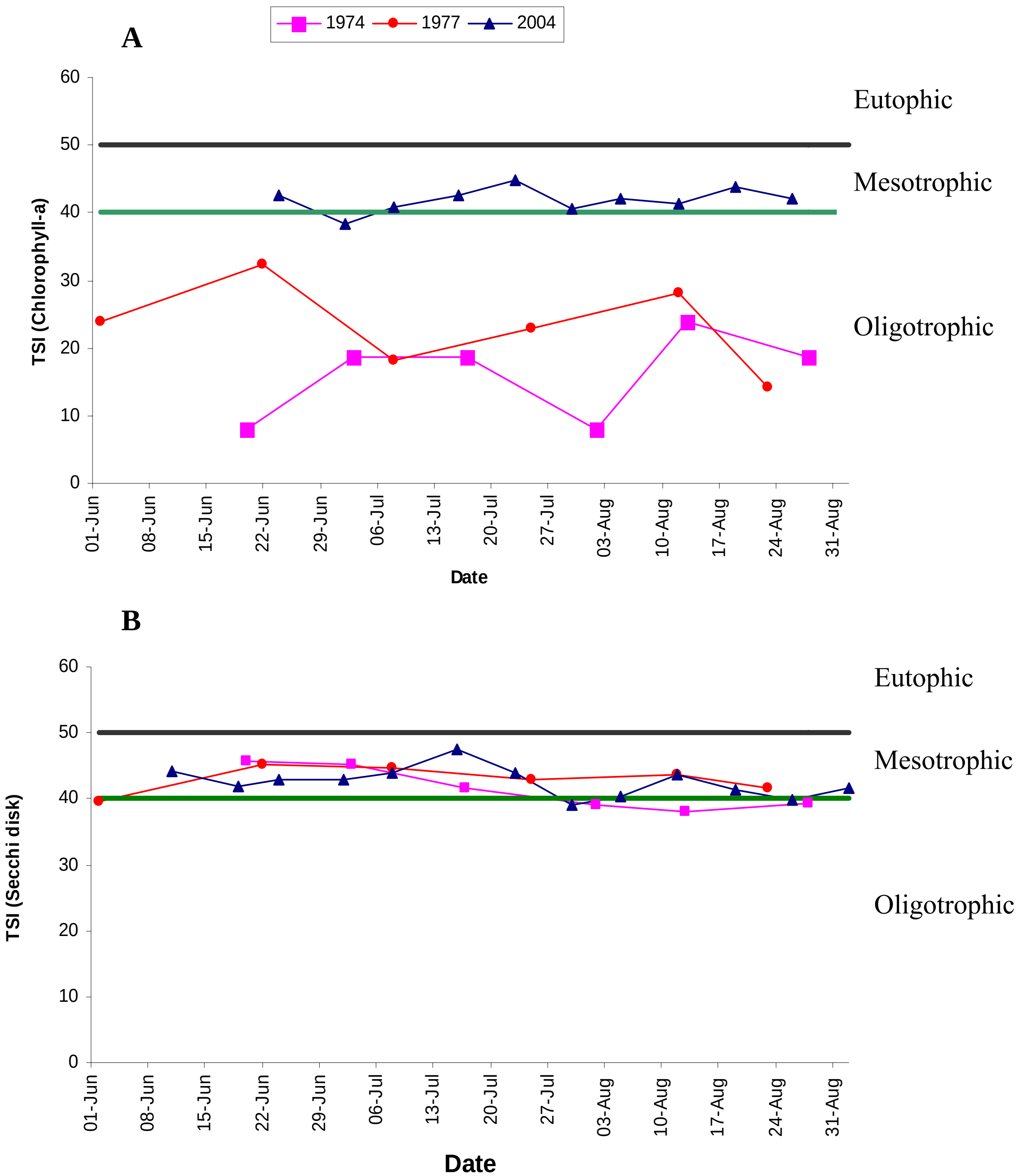


Figure 12. Trophic State Index (*TSI*) values calculated using: A) chlorophyll-*a* (µg l^{-1}), and B) Secchi disk depth (m) measurements from Springfield Lake, Halifax Regional Municipality, for each sampling day for the study periods of 20 June to 28 August 1974 (square), 02 June to 23 August 1977 (circle), and 04 June to 02 September 2004 (triangle). The green line marks the transition from oligotrophic (0-40) to mesotrophic (40- 50) and the black line marks the mesotrophic to eutrophic (50-60) transition according to Carlson (1977).

Contemporary benthic macroinvertebrate diversity

The water column was well mixed with respect to temperature at all five sites when the benthic samples were collected (**Table 6)**. Oxygen saturation at all sites showed stratification, with a DO saturation at 2 m of 36%; indicative of a hypoxic environment (**Table 7**). Nine taxa were observed among the benthic samples (**Table 7**): Hirudinea (leeches), Oligochaeta (worms), Chironomidae (midges), Helicopsychidae and Hydroptilidae (caddisflies), Sphaeriidae (mussels), Valvatidae and Hydrobiidae (snails). The Family term Bivalvia is used here to refer to small, translucent bivalves that I could not identify to family. Community composition varied among sites (**Figure 13**) and no single taxon was present in every sample, though Chironomidae, Sphaeriidae, and Valvatidae were in at least one sample at all sites. Diversity and biotic indices were relatively consistent among sites (**Table 8**).

Hydrobiidae and Bivalvia were the most common Families, each comprising more than 25% of the total community (all samples combined). All sites were dominated by either Hydrobiidae (Sites 1 and 2) or Bivalvia (Sites 4 and 5) with the exception of Site 3 which was dominated by Chironomidae (**Figure 13**).

The number of taxa per sample, S (averaged from two samples) ranged from 6 to 7. Site 2 had the greatest number of taxa (8) and Sites 3 and 4 both had 6 taxa present. The Simpson's Diversity Index, *D*, for each site (averaged from two samples) ranged from 0.61 to 0.79. Sites 2 and 4 had the lowest diversity, and Site 5 the highest. These values indicate that there is a 60-80% chance that two species randomly taken from this community would be in the same Family. The Shannon-Wiener Diversity indices, *H*, ranged from 1.14 to 1.67 and as above; Site 4 had the lowest *H* and Site 5 had the highest. Evenness, *J*, at the sites ranged from 0.70 (Site 2) to 0.89 (Site 5), on a scale of 0 to 1 where 1 is perfectly even.

Table 6: Water temperature (^{o}C) and dissolved oxygen (DO) saturation profiles at five benthic macroinvertebrate sampling sites in Springfield Lake, Halifax Regional Municipality. Site 1 was measured on 27 October. Sites 2 to 5 were measured on 29 October 2004.

	Water Temperature (^{o}C)					**Per cent DO Saturation**				
Site	1	2	3	4	5	1	2	3	4	5
Surface	10.5	9.1	9.4	9.2	9.3	118.5	105.6	108.7	102.2	106.5
1 m	10.5	9.1	9.4	9.1	9.3	61.9	44.2	77.1	50.2	77.9
2 m	10.5	9.1	9.4	9.0	9.3	35.0	35.5	35.9	36.4	35.5
3 m	10.5	-	9.4	-	9.3	30.5	-	29.8	-	29.4

Table 7: Number of individual benthic macroinvertebrates observed per Family at five sites (two replicates per site) in Springfield Lake, Halifax Regional Municipality, for 27 and 29 October 2004.

Order	Family	1A	1B	2A	2B	3A	3B	4A	4B	5A	5B	Total
Hirudinea	Hirudinea	-	1	1	1	4	2	-	-	1	8	18
Oligochaeta	Oligochaeta	-	1	-	-	1	-	1	-	-	-	3
Diptera	Chironomidae	1	3	-	4	1	5	1	1	4	16	36
Trichoptera	Helicopsychidae	-	-	1	-	-	-	-	-	-	-	1
Trichoptera	Hydroptilidae	-	1	-	2	-	-	-	-	-	2	5
Bivalvia	Sphaeriidae	3	1	2	23	-	2	-	4	3	3	41
Bivalvia	Bivalvia	-	-	2	9	6	-	11	28	6	17	79
Gastropoda	Valvatidae	1	3	-	19	1	1	2	4	1	4	36
Gastropoda	Hydrobiidae	11	2	17	20	-	-	15	11	3	12	91
Totals		16	12	23	78	13	10	30	48	18	62	310

Table 8: Index values for: number of species per taxa (*S*), Simpson's Diversity Index (*D*), Family Biotic Index (*FBI*), Shannon-Wiener Diversity Index (*H*), evenness (*J*), Belgian Biotic Index (*BBI*), Biological Monitoring Working Party score system (*BMWP*), and average score per taxon (*ASTP*), based on macroinvertebrates collected in Springfield Lake, Halifax Regional Municipality, on 27 and 29 October 2004.

Index	1A	1B	2A	2B	3A	3B	4A	4B	5A	5B
S	4	7	5	7	5	4	5	5	6	7
D	0.48	0.82	0.44	0.77	0.78	0.66	0.61	0.6	0.78	0.8
FBI	6.5	7	6.4	7.3	8.5	7.4	6.9	7.5	7.3	7.2
H	0.92	1.82	0.92	1.60	1.31	1.22	1.12	1.15	1.62	1.72
J	0.66	0.94	0.57	0.82	0.81	0.88	0.70	0.71	0.90	0.88
BBI	6	5	5	5	5	3	7	7	5	6
BMWP	11	21	12	23	12	11	12	14	17	23
ASTP	2.8	3	2.4	3.3	2.4	2.8	2.4	2.8	2.8	3.3

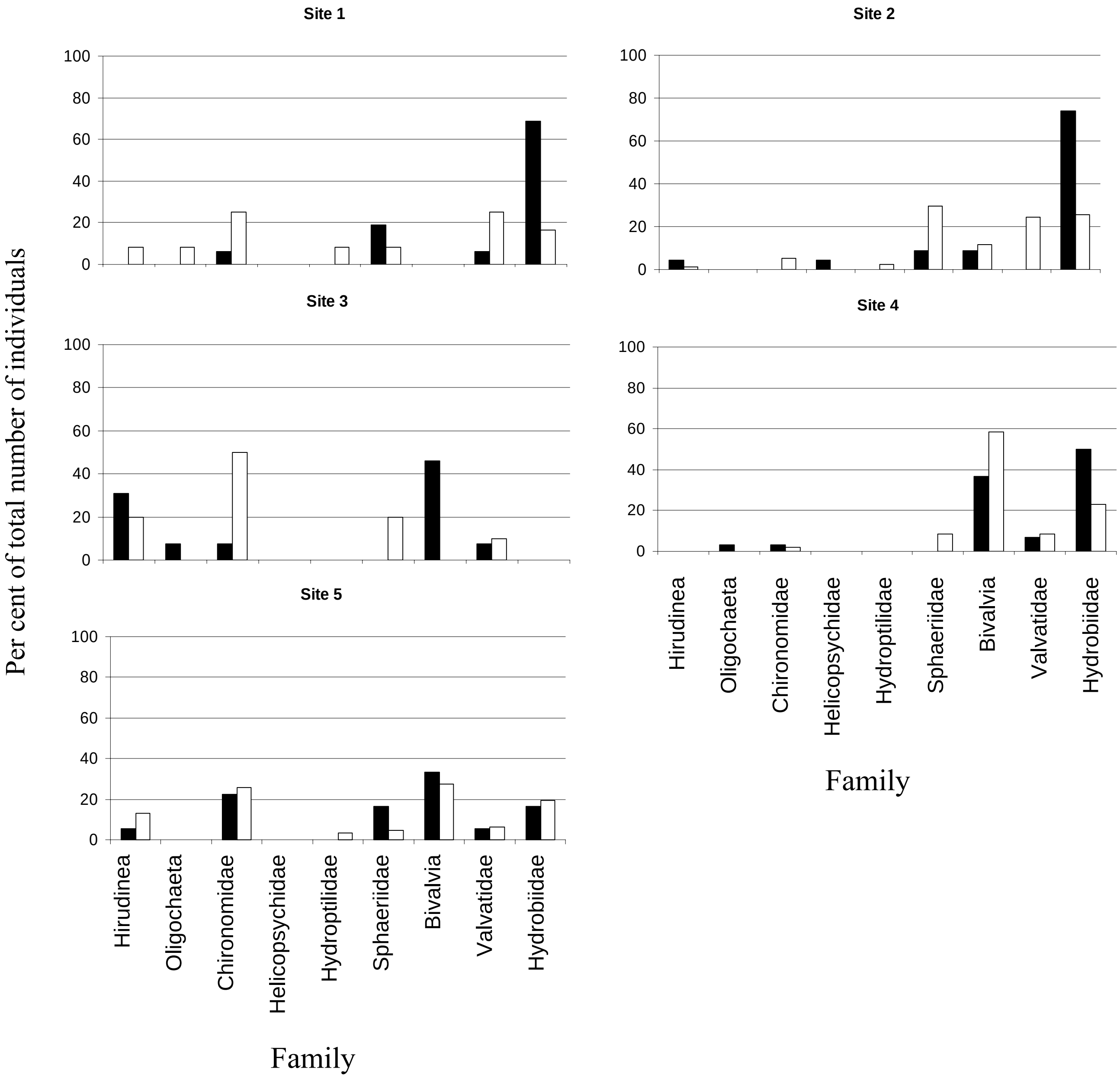


Figure 13. Per cent composition (Family) of the benthic community based on two samples (black vs white) at each site for Springfield Lake in 2004.

The *BMWP* scores ranged from 11.5 (Site 3) to 20 (Site 5). The corresponding *ASTP* scores indicated that water at all sites was suffering from probable severe pollution (*ASTP* = 2.6 to 3.1). The *FBI* scores imply that the lake has poor (Sites 1, 2 and 4) or very poor (Sites 3 and 5) water quality and the *BBI* values show the lake to be slightly (Site 4) to moderately polluted (Sites 1, 2, 3, and 5).

Historical benthic macroinvertebrate diversity and temporal comparison

Hyalellidae and Chironomidae were the most common Families in 1997. All sites were dominated by either Hyalellidae (Sites 1 and 2) or Chironomidae (Sites 3, 4 and 5). Hyalellidae was not found in the 2004 samples, and Chironomidae was present at all sites but only dominant at Site 3.

The number of taxa per sample, *S* (averaged from two samples) was 2 to 3 times larger in 1997 than in 2004 ($p<0.001$) and ranged from 14 (Site 1) to 18 (Site 2) (**Table 9**). The Simpson's Diversity Index, *D*, for each site (averaged from two samples) ranged from 0.51 to 0.76. Site 5 had the lowest diversity, and Site 2 the highest. These values indicate that there is a 50-80% chance that two species randomly taken from this community would be in the same Family; similar to 2004 ($p = 0.65$). The Shannon-Wiener Diversity Index, *H,* values ranged from 1.62 to 2.43, and these were also not different from 2004 ($p=0.80$). Evenness, *J*, at the sites ranged from 0.61 (Site 5) to 0.84 (Site 2) and again there was no difference relative to 2004 ($p=0.4$).

Table 9: Historical (Hynes 1998; Kirsh 1999) and contemporary (this study) index results for each benthic macroinvertebrate sampling site in Springfield Lake, Halifax Regional Municipality. Indices used: number of species per taxa (*S*), Simpson's Diversity Index (*D*), Shannon-Wiener Diversity Index (*H*), evenness (*J*), Biological Monitoring Working Party score system (*BMWP*).

Site	**1**		**2**		**3**		**4**		**5**		**Mean**	
Index	**1997**	**2004**	**1997**	**2004**	**1997**	**2004**	**1997**	**2004**	**1997**	**2004**	**1997**	**2004**
S	14	7	18	8	15	6	15	6	14	7	15.2	6.8
D	0.7	0.65	0.76	0.61	0.52	0.72	0.71	0.61	0.51	0.79	0.64	0.68
H	2.13	1.37	2.43	1.26	1.73	1.27	2.07	1.14	1.62	1.67	2.00	1.34
J	0.81	0.8	0.84	0.70	0.64	0.85	0.76	0.71	0.61	0.89	0.73	0.79
BMWP	34	12	48	17.5	35	11.5	28	13	39	20	36.8	14.8

The *BMWP* scores in 1997 ranged from 28 (Site 4) to 48 (Site 2) and similar to 2004 the corresponding *ASTP* scores indicated that water at all sites was suffering from probable moderate pollution (*ASTP*= 4.00- 4.88; **Table 10**). The *FBI* scores imply that the lake has fairly poor (Site 1), poor (Sites 2, 3, and 4) or very poor (Site 5) water quality. This is slightly better than 2004 when all sites were indexed as either poor or very poor in terms of water quality. The 1997 *BBI* values show the lake to be moderately polluted at all sites, whereas in 2004 the lake was slightly polluted at one site, and moderately polluted at the others.

Table 10: Summary of the Family Biotic Index (*FBI*), Belgian Biotic Index (*BBI*), and Average Score Per Taxon (*ASTP*) values for 1997 (Hynes 1998; Kirsh 1999) and 2004 benthic macroinvertebrate sampling in Springfield Lake, Halifax Regional Municipality, and the corresponding water quality interpretations. *FBI*: FP = fairly poor; P = poor; VP = very poor. *BBI*: M = moderately polluted; S = slightly polluted. *ASTP*: M = probable moderate pollution; S = probable severe pollution.

Site	**1**		**2**		**3**		**4**		**5**		**Mean**	
Index	**1997**	**2004**	**1997**	**2004**	**1997**	**2004**	**1997**	**2004**	**1997**	**2004**	**1997**	**2004**
FBI	6.2	6.8	6.6	6.9	6.9	8.0	7.1	7.2	7.4	7.3	6.8	7.2
Water quality	FP	P	P	P	P	VP	P	P	VP	VP	P	P
BBI	6	5.5	6	5	5	4	5	7	5	5.5	5.4	5.4
Water quality	M	M	M	M	M	M	M	S	M	M	M	M
ASTP	4.25	2.9	4.80	2.85	4.38	2.6	4.00	2.6	4.88	3.1	4.46	2.81
Water quality	M	S	M	S	M	S	M	S	M	S	M	S

Discussion

According to the chlorophyll-*a* based *TSI*, the trophic state of Springfield Lake over the last 30 years has changed from predominantly oligotrophic to highly mesotrophic. When based on chlorophyll-*a* measurements this change has been significant; when based on Secchi disk depth there has been no change. There are intrinsic problems with using Secchi depth values to calculate *TSI*, as it can be ambiguous (Megard *et al*. 1980); at low chlorophyll-*a* levels Secchi depth can be controlled by light-absorbing particles other than chlorophyll-*a* (Lorenzen 1980) and Secchi disk observations are subject to observer bias. For these reasons, chlorophyll-*a TSI* values are to be given more weight than Secchi depth *TSI* values (Carlson 1980). As each increase of seven units on the *TSI* scale represents a doubling in algal biomass (Carlson 1980) it can be inferred that chlorophyll-*a* has more than quadrupled in Springfield Lake since 1974, a fact that is consistent with the mean chlorophyll-*a* in 1974 (0.3 ± 0.2 μg l^{-1}) increasing to 3.2 ± 0.6 μg l^{-1} in 2004. Further, this diagnosis is consistent with the decrease in dissolved oxygen saturation and increase in nutrients.

Dissolved oxygen in Springfield Lake in 2004 is of particular interest in terms of the eutrophication of the lake. Dissolved oxygen saturation in 2004 was significantly stratified (**Figure 3**) and at depths of 1, 2, and 3 m saturation was significantly lower than at the surface, and this was not the case in 1977. The water column was not thermally stratified in either 1977 (**Figure 4**) or 2004 (**Figure 3**) which indicates that something is causing the stratification of oxygen observed in 2004. The effect of water temperature on oxygen saturation (Wetzel 1975) was removed by using per cent saturation for comparisons and water temperature was not different between years. Therefore, differences in oxygen saturation cannot be attributed to temperature differences or insufficient water mixing. The water column was well oxygenated in 1977 and poorly oxygenated in 2004. If wind mixing is sufficient to mix temperature in the top 3m of the water column in 1977 and 2004 then it is reasonable to conclude that in 2004 oxygen from the surface is also mixed down (as in 1977) but is being depleted at a greater rate. There is some evidence that wind events are having an effect on the oxygen stratification, when periods of high wind result in a well-mixed water column and periods of weak or average winds do not exert enough stress to mix down the water column, but this relationship was not entirely clear in the 2004 data. Oxygen in the water column is used up by plant and animal respiration and primarily by bacterial respiration during the breakdown of organic matter (Wetzel 1975). Thus the simplest explanation for the dissolved oxygen stratification in 2004 is the decomposition of the algae and plant matter.

The significant increase in algal biomass, as represented by chlorophyll-*a*, is entirely consistent with the oxygen stratification observed in 2004. Phosphorous is most frequently the limiting nutrient to plant growth in freshwater (Carlson 1977) and there is no time series of phosphorous available in my study. However, the one observation from 2004 was within the observed historical range, indicating that, at best, phosphorous has not decreased in the Lake. Furthermore, lakes rich in total nitrogen can be shown to be rich in phosphorous (Downing and McCauley 1992), which suggests that phosphorus in the Lake should be increasing with the observed increase in nitrogen. Ramstack *et al*. (2004) compared pre-colonisation (1750-1800) to post-colonisation water quality and found that urban lakes showed increases in total phosphorous. Phosphorous and nitrate loading has also been linked with hypoxia (Hagy *et al*. 2004) presumably as a result of their relationship with primary production. Plants also release nutrients into the water when they decompose, providing nutrients for new plant growth. When extraneous sources of nutrients are added to this cycle, nutrient loading and excess plant growth occurs, i.e. eutrophication (Wetzel 1975). Hall *et al*. (1999) studied human impacts on prairie lakes over 1776-1996 using paleolimnological techniques and found that human land use was a greater determinant of algal community changes than climactic factors such as temperature or evaporation.

Given that chlorophyll-*a* has increased significantly it was unexpected that no change was observed in either turbidity or Secchi depth. It is especially surprising for Secchi depth, as the relationship between Secchi depth and chlorophyll-*a* is well documented (Carlson 1977; Brezonik 1978; Lorenzen 1980). It is possible that the increase in chlorophyll-*a*, while significant relevant to historical levels, was not significant enough to affect light attenuation and water clarity; an interpretation consistent with the relatively constant light transmittance values I observed in 2004 (**Figure 9d**).

The historical increase in chlorophyll-*a* is concurrent with the observed nitrate increase. This nutrient increase over time likely has two sources, natural and anthropogenic. As the algae and plants in the Lake decompose they release nutrients into the water. As well, runoff from residential properties and erosion from development increase nutrients in the lake. It is not clear how nitrate concentration has changed over the last 30 years, as the historical measures were of nitrate and nitrites. Contemporary nitrate levels are a great deal higher than nitrate/nitrite measurements in 1974 and 1977, though they are well within the range of nitrate (0 to 10 mg l^{-1}) commonly observed in unpolluted fresh water (Wetzel 1975) and in Appalachian regions of fresh water springs (0.06 to 14 mg l^{-1}; Van Everdingen 1991), and nitrate is subject to high temporal and spatial variability (Wetzel 1975). Nitrite is short-lived in water (Wetzel 1975) so its contribution to the 1974 and 1977 values can likely be ignored. Based on the QEII nitrate/nitrite results, it is reasonable to conclude that nitrate ion concentrations in the lake have increased along with other ions such as calcium and ammonium. Contemporary ammonium ion concentration was high as well, though there is no historical baseline available for comparison. Given the increase and high levels of calcium, ammonium, and nitrate ions it is not surprising that conductivity increased with time as well. Conductivity increased significantly between 1974 and 1977 and again between 1977 and 2004 and there is some indication in the 2004 data for a delayed response to rain events, where some measurements (i.e., conductivity, calcium) increased shortly following periods of high rainfall and subsequent runoff into the lake.

There are no reliable data in this study to demonstrate an increase in chloride over the past three decades, though it is unreasonable to assume no change in that ion. The Ramstack *et al*. (2004) comparison of presettlement and modern times in Minnesota lakes shows increases in chloride were most strongly correlated with the percentage of watershed area devoted to residential or commercial land use and road surfaces. These results were ascribed to road salting, but salting is banned on the roads around Springfield Lake, though not on the highway at the southern end of the lake that passes though the watershed (C. Taggart, Springfield Lake Watch, Middle Sackville, Nova Scotia., pers. comm.). The observed increase in calcium however is likely due to runoff from residential properties containing lime. The increase in calcium entering through runoff can influence lake acidity, though this was not apparent in the data (**Figure 11c**).

The change in pH over the past 3 decades is one of interest, as it was expected that pH would decrease. Springfield Lake lies in pyretic bedrock, which can lead to high levels of sulphate, and therefore low pH, in lakes (Kerekes *et al.* 1986). In 1974 the Lake was relatively neutral, and in 1977 it was more acidic, reaching as low as 5.3. In 2004 the Lake returned to a pH of 6.5. The decrease in pH between 1974 and 1977 is consistent with Watt and Ray (1979) who conducted a study on the long term trends in acidification of 16 Halifax lakes from 1955 to 1979 and observed declining pH in all lakes studied. Kerekes *et al*. (1986) considered atmospheric hydrogen sulphide to be the primary cause of acidification in the surface waters of Nova Scotia lakes. The increase to a more neutral pH between 1977 and 2004 is also consistent with observations by Watt and Ray (1979) in the above mentioned study. Four lakes that were undeveloped in 1955 and developed in 1979 showed increases in pH. This is consistent with the rise in pH between 1977 and 2004 being explained by anthropogenic factors. As well, Kerekes et al. (1986) posit that calcium increases are able to decrease hydrogen ion concentrations, and therefore increase pH in lakes naturally high in sulphates. Perhaps more importantly, increases in pH have been found to reflect high levels of algal productivity (Ramstack *et al.* 2004) as the algae take up carbon dioxide during photosynthesis. Increases in pH can also be attributed to concentrations of other nutrients. As nitrate, ammonium, and calcium ion concentrations have all increased significantly during the period in which pH has been increasing, it is likely these are contributing to the change in pH.

For per cent dissolved oxygen saturation, calcium ion concentration, turbidity, and chlorophyll-*a*, the coefficient of variation in the 2004 data was greater than in 1974 or 1977. This could be an effect of the higher temporal resolution of the trends in 2004. However, Ramstack *et al.* (2004) observed that human activities can affect the range of limnological variation in urban areas particularity with respect to nutrients, presumably related to point-source pollution events (runoff, fertilization practices, etc.)

Lake trophy can influence the benthic macroinvertebrate community (Staicer *et al.* 1994b) but the above trend of eutrophication in Springfield Lake was not necessarily reflected in the benthic macroinvertebrate data in measures of diversity. However, the magnitude of change that was observed in the physical and chemical variables was over 30 years, and the benthic macroinvertebrate comparison is only over 7 years. Even though little change was observed, the biological indices described as having poor water quality and suffering from probable moderate pollution in 1997 (Hynes 1998) remained in 2004.

Most species diversity indices are based on the underlying principle that undisturbed communities will have a high species richness and diversity (Metcalfe 1989). Based on the 2004 measurements, Site 5 had the highest diversity and Site 2 the lowest (**Figure 13**). This is an interesting reversal of the 1997 observations where Site 2 had the highest diversity and Site 5 had the lowest. Site 5 is located near sewer Lift Station number 8 (**Figure 1**), which overflows municipal sewage into the lake in times of high rainfall (greater than 70mm; unpublished data, Halifax Regional Municipality, Environmental Management Services, 21 Mount Hope Ave., Dartmouth, N.S.). Given the occasional source of nutrients that this may provide to the general area, it is reasonable that Site 5 has more taxa and greater diversity. Over-all there was no significant difference between the diversity measures used and it is possible that the community has not been noticeably disturbed, or is under any more stress than in 1997. However, there are no benthic macroinvertebrate data prior to extensive urban development, when the benthic community was arguably under less stress.

Though it can be measured with myriad indices, diversity is essentially a measure of the number of species present (Hill 1973). Hence, the total number of taxa (*S*) is generally used as a simple expression of diversity (Barton and Metcalfe-Smith 1992). *S* in 1997 was greater than in 2004. However, a greater *S* in 1997 may reflect the differences in sampling more than a difference in the benthic community, and this can be traced to the species-area relationship (Hill 1973). Sampling in 1997 used the kick-net method (Hynes 1998) and covered a larger area than the Ekman samples in 2004. Simpson's Diversity Index (*D*) is sensitive to the more abundant species and is best used to determine dominance concentration (Hill 1973) as *D* will decrease with increasing diversity, but there was no difference in *D* between 1997 and 2004. The Shannon-Weiner Diversity Index (*H*), which increases with increasing diversity, was larger in 1997 than 2004, though the 2004 samples had a slightly greater evenness (*J*) than did the 1997 samples. It should be noted that both *J* and *H* are designed for species-level data and I am using them here at the Family level.

Barton and Metcalfe-Smith (1992) compared benthic indices between control and impacted (agricultural and industrial) sites and found that neither *H* nor *D* could successfully separate control and impact sites. In fact, they found that no index satisfactorily discriminated between impact and control sites. Some indices were affected by sampling method, and they determined that assessments of benthic biota depend on both the sampling method and the indices used. Of all the indices examined, Barton and Metcalfe-Smith (1992) concluded that, in general, the more taxonomically demanding indices gave better results and most accurately reflected ecological degradation resulting from organic pollution. For my study it is not possible to measure the accuracy of the indices used, however there was little difference among sites, indicating uniformity in the Lake and it is likely that all 5 sites suffer comparable anthropogenic impacts, as indicated by the similar *FBI*, *ASTP*, and *BBI* indices below.

The Belgian Biotic Index (*BBI*) assigns a value based on the number of the most pollution-sensitive Family in a sample (De Pauw and Vanhooren 1983; **Appendix Table 3**). In both 1997 and 2004, the most pollution sensitive Family was Trichoptera and the most pollution tolerant Family was Chironomidae. Pollution sensitive families like Trichoptera made up less than 1% of the total community in 1997, and only 2% in 2004. Pollution tolerant Families like Chironomidae and Sphaeriidae made up 33% of the total community in 1997, and 25 % in 2004. The most glaring difference between the biota in 1997 and 2004 was that the 1997 samples contained no bivalves and only one gastropod, whereas Sphaeriidae bivalves and Valvatidae gastropods were found at every site in 2004. This difference is not easily explained by sampling discrepancy as the two methods should adequately sample these two families. The dominant taxa in 1997 were Hyalellidae (amphipods) and Chironomidae (Diptera), and in 2004 the dominant taxa were Hydrobiidae gastropods and Bivalvia. This could be an effect of pH, as macroinvertebrate richness is negatively associated with acidity (Schell and Kerekes 1989). More acidic lakes have greater proportions of Diptera and Trichoptera, as seen in 1997, while clams and gastropods, which were abundant in 2004, are absent at pH levels of less than 5 (Schell and Kerekes 1989). The increase in calcium levels may be one explanation for why molluscs were observed at low levels in 1997, and were omnipresent in 2004 - molluscs require calcium (Staicer *et al*. 1994b).

According to the *FBI*, *BBI*, and *ASTP*, Springfield Lake in 2004 had poor water quality and was suffering from moderate to severe pollution. These estimates are slightly worse than what the same indices indicated for 1997: poor water quality and moderate pollution. Biotic indices are based more on the pollution tolerance of taxa present in samples than on the sample sizes or diversity, so the differences in sampling between 1997 and 2004 should not influence these results. Thus, it is reasonable to conclude that water quality in Springfield Lake, as reflected in the benthic macroinvertebrate community, has neither changed nor improved since 1997

Where is Springfield Lake headed?

This study did not address climate change over the past thirty years and its effect on the variables measured. The mean air temperature over the course of the study period did not increase but the effects of the climate on water temperature, hydrology and watershed and in-lake process are likely to be complex (Schindler *et al*. 1996). In a lake as shallow as Springfield (maximum depth 5 m), increased evaporation may be a factor that can affect the other variables (e.g. conductivity).

The time it takes a lake to recover from eutrophy can be long; from the time nutrient input is reduced to the time a lake is declared recovered can take more than a decade (Edmondson and Lehman 1981; Carpenter *et al.* 1998). In some cases, nutrient reduction is not sufficient, as observed by Hall *et al.* (1999) in a lake where water quality did not improve over the course of 17 years, despite a 3-fold decrease in phosphorous input. Recovery in shallow lakes can be especially complicated; the balance in a shallow lake ecosystem is difficult to recover once it is lost (Phillips *et al.* 1999).

The trend over the past 30 years, as far as it can be resolved, shows Springfield Lake to be approaching a eutrophic state. The present anthropogenic influences on the lake are doubtless enough to maintain the lake at this level, or push it further into eutrophy. If nothing is done to mitigate this, Springfield Lake will continue toward a eutrophic and possibly hypereutrophic state.

However, and perhaps optimistically, Springfield Lake offers a unique opportunity for effective, efficient, and timely management as it is a headwater lake. All water entering Springfield Lake does so either through springs, streams, or runoff from the watershed. There is no influence from other lakes, rivers, or input upstream; the people living within the watershed have a large degree of control over what enters the lake. Though the recovery of eutrophic shallow lakes has proved difficult, it has also proved possible (Annadotter *et al.* 1999; Madgwick 1999; Phillips *et al.* 1999), and the ability of to control point-source pollution is an indisputable advantage. Control of runoff is essential, but any management initiative must also address the entire watershed, as any lake management focused solely on a lake and its shoreline is destined to be naïve and ineffective (Dillon and Rigler 1975). Incorporation of the shoreline is important because human modification of shoreline can lead to increased erosion and runoff, and has also been shown to impact aquatic organisms, for example, altering the spatial distribution of fish (Scheuerell and Schindler 2004). Consequently, a focus incorporating the entire watershed is imperative because the terrestrial landscape and the lake are intrinsically connected through drainage (Dillon and Rigler 1975; Johnes 1999) and this relationship is even more important in Springfield Lake where the watershed is the sole source of input. Any change in the watershed affects the lake and the processes operating in the lake, such as the way the lake uses nutrients (Edmondson and Lehman 1981). Consequently, Springfield Lake, being a headwater lake, is ideal from a management standpoint.

As Springfield Lake is not highly eutrophic, and it is a headwater lake, it is probable that proper management can restore it to a less eutrophic condition. This will require active stewardship by watershed residents, particularly those with shoreline property. Measures should be taken to reduce the amount of nutrient-rich runoff and erosion that enters the lake. As well, comprehensive water quality sampling should continue throughout subsequent years if the success or failure of mitigation measures is to be known. This will not only confirm the results of the current study, it will show the effectiveness of any management measures.

Conclusion

Springfield Lake has changed from relatively oligotrophic and well-oxygenated in the 1970s to mesotrophic and near hypoxic at depth in 2004. There is an obvious trend toward eutrophication with time in the physical, chemical, and biological data. Measures of the benthic macroinvertebrate community in 1997 and 2004 indicate that the water quality in Springfield Lake is increasingly poor and suffering from moderate pollution. Without active management it is likely that Springfield Lake will continue on its current course and become eutrophic.

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Appendix

Table 1: Carlson's (1977) Trophic State Index (after Carlson and Simpson 1996)

***TSI* score**	**Chlorophyll-*a*** $\mu g\ l^{-1}$	**Secchi Depth** m	**Total Phosphorous** $\mu g\ l^{-1}$	**State**
0 - 30	<0.95	> 8	< 6	Oligotrophic
30 - 40	0.95 – 2.6	8- 4	6-12	
40 - 50	2.6 – 7.7	4 - 2	12 - 24	Mesotrophic
50 – 60	7.3 - 20	2 -1	24 -48	Eutrophic
60 – 70	20-56	0.5 – 1	48 – 96	
70 - 80	56 - 155	0.25 – 0.5	96 - 192	Hypereutrophic
> 80	>155	<0.25	192-384	

Table 2: Taxon-specific tolerance values used for the Family Biotic Index (FBI) and the Biological Monitoring Worker's Party (BMWP; after Kirsh 1999).

Order	**Family**	**FBI Value**	**BMWP Value**
Hirudinea	Hirudinea	10	0
Oligochaeta	Oligochaeta	8	0
Diptera	Chironomidae	6	2
Trichoptera	Helicopsychidae	3	0
Trichoptera	Hydroptilidae	4	6
Bivalvia	Sphaeriidae	8	3
Bivalvia	Bivalvia	8	3
Gastropoda	Valvatidae	8	3
Gastropoda	Hydrobiidae	6	3

Appendix (cont.)

Table 3: Standard table to determine BBI index. Standard Units (SU) are the number of systematic units observed of a particular faunisitic group (after De Pauw and Vanhooren 1983)

Faunistic groups	II	Total numbers of systematic units present				
		0-1	2-5	6-10	11-15	16 or more
		Biotic Index				
Plecoptera or Ecdyonuridae	Several SU	-	7	8	9	10
	Only 1 SU	5	6	7	8	9
Cased Trichoptera	Several SU	-	6	7	8	9
	Only 1 SU	5	5	6	7	8
Ancylidae or Ephemeroptera	More than 2 SU	-	5	6	7	8
	2 or less SU	3	4	5	6	7
Aphelocheirus or Odonata or Gammaridae or Mollusca (except Sphaeridae)	All SU mentioned are present	3	4	5	6	7
Asellus or Hirudinea or Sphaeridae or Hemiptera	All SU mentioned are present	2	3	4	5	-
Tubificidae or Chironomidae	All SU mentioned are present	1	2	3	-	-
Eristaline (=Syrphidae)	All SU mentioned are present	0	1	1	-	-

Table 4: Water quality interpretations of FBI and ASTP values (after Kirsh 1999)

FBI Value	Water Quality	ASTP Value	Water Quality
0 – 3.75	Excellent	> 6	Clean water
3.76 – 4.25	Very Good	5 -6	Doubtful quality
4.26 – 5.00	Good	4-5	Probable moderate pollution
5.01 – 5.75	Fair	<4	Probable severe pollution
5.76 – 6.50	Fairly Poor		
6.51 – 7.25	Poor		
7.26 – 10.00	Very Poor		