Lake
Simcoe
Environmental
Management
Strategy

Implementation Program 2005

Lake Simcoe Water Quality Update 2000 - 2003
Technical Report Imp. B.20

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FOREWORD

This report is one of a series of technical reports prepared in the course of the Lake Simcoe Environmental Management Strategy (LSEMS) Implementation Program. This program is under the direction of the LSEMS Steering and Technical Committees, comprised of representatives of:

- Lake Simcoe Region Conservation Authority;
- Chippewas of Georgina First Nations
- Department of Fisheries and Oceans
- Ministry of Agriculture and Food;
- Ministry of the Environment;
- Ministry of Municipal Affairs and Housing;
- Ministry of Natural Resources;
- Regional Municipalities of Durham and York;
- County of Simcoe;
- Cities of Barrie, Kawartha Lakes and Orillia;
- Towns of Bradford West Gwillimbury, Innisfil and New Tecumseth; and
- Townships of Oro-Medonte and Ramara.

The Lake Simcoe Environmental Management Strategy (LSEMS) studies were initiated in 1981 in response to concern over the loss of a coldwater fishery in Lake Simcoe. The studies concluded that increased urban growth and poor agricultural practices within the drainage basin were filling the lake with excess nutrients. These nutrients promote increased weed growth in the lake with the end result being a decrease in the water's oxygen supply. The "Final Report and Recommendations of the Steering Committee" was released in 1985. The report recommended that a phosphorus control strategy be designed to reduce phosphorus inputs from rural and urban sources. In 1990 the Lake Simcoe Region Conservation Authority was given overall coordination responsibilities for Phase I Implementation, as outlined in the LSEMS Cabinet Submission and subsequent agreement (Recommendation E.1). At the completion of Phase I Implementation, a report was produced. "LSEMS Implementation Program - Summary of Phase I and Recommendations for Phase II, 1995" outlined the activities and progress during Phase I and presented recommendations for Phase II of the LSEMS Implementation Program. The LSEMS Implementation Program is currently in its third phase of implementation (2001 - 2006).

The goal of the LSEMS Implementation Program is to improve and protect the health of the Lake Simcoe watershed ecosystem and improve associated recreational opportunities by:

- Restoring a self sustaining coldwater fishery,
- Improving water quality,
- Reducing phosphorus loads to Lake Simcoe, and,
- Protecting natural heritage features and functions.

The LSEMS Implementation Program will continue to initiate remedial measures and control options designed to reduce phosphorus inputs entering Lake Simcoe, monitor the effectiveness of these remedial measures and controls and evaluate the overall response of the lake to this program. Through cost sharing programs, environmental awareness by the public and further studies, the goal of restoring a naturally reproducing coldwater fishery in Lake Simcoe by improving water quality can be reached.

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SUMMARY

Prompted by concerns over the eutrophication of Lake Simcoe, the Ontario Ministry of the Environment initiated a lake-wide water quality monitoring program in the early 1980s. A number of indicator variables, including total phosphorus (TP), chlorophyll *a*, Secchi disk visibility, dissolved oxygen (DO) and phytoplankton biovolume have been consistently measured since that time, at between 8 and 12 in-lake sampling stations, in order to assess changes in the nutrient status of the lake. Water quality is also routinely monitored at 3 (4 pre-1997) municipal water intake stations, and at the lake outflow near the Atherley Narrows. The purpose of this report is to update previous reports on the water quality of Lake Simcoe with the most currently available data from the
Ontario Ministry of the Environment, and to extend the existing data record to 2003.

Despite large increases in population and urban area within the watershed over the past 2 decades, average TP levels were generally lower in 2000-2003 compared to 1980-1983 at the majority of lake stations, although declines were only significant at the high TP sites of C1 and C6 in Cook's Bay and K39 in Kempenfelt Bay. Similarly, TP concentrations at the outflow decreased slightly between 1983-86 (13.2 ± 2.2 µg/L) and 2000-2003 (10.2 ± 0.7 µg/L), and were consistently lower than at any lake station. Significant declines in TP also occurred in nearshore water sampled at the municipal water intakes at Beaverton (0.35 µg/L/year), Keswick (0.34 µg/L/year) and Sutton (0.24 µg/L/year) over the same time period. In contrast, chloride (Cl) concentrations in the lake outflow and at the municipal water intakes are currently almost double levels in the early 1980s, reflecting increased urbanization of the watershed. Rates of Cl increase were similar (0.65-0.78 mg/L/year) across the intake and outflow stations and indicate a well-mixed lake.

Chlorophyll *a* concentrations declined significantly at only 1 lake station (C6), likely due to high variability in annual average values, but temporal patterns in chlorophyll *a* were generally similar across the lake. Maximum concentrations of chlorophyll *a* and phytoplankton biovolume occurred at stations having the highest TP, and these sites also had the lowest water clarity (as indicated by Secchi disk visibility). Significant declines in phytoplankton biovolume and increases in water clarity occurred at the majority of stations over the 20 year period of record, and Secchi disk depths were on average 40-80% greater in 2000-2003 compared to 1980-1983. Inter-annual patterns in biovolume and water clarity were highly coherent among lake stations, indicating that these changes were consistent across the lake and support the concept of a well-mixed system. Zebra mussels, which became established in Lake Simcoe around 1995, may have contributed to decreased phytoplankton biovolume and related increases in water clarity, which began during the mid-1990s.

Average rates of DO depletion (18m-bottom zone; normalized to 4°C) in 2000-2003 (average 1.25 ± 0.21 g O_2/m^3/month) were similar to rates in 1980-1983 (average 1.26 ± 0.19 g O_2/m^3/month), but were lower than values in the 1990s. Although late summer, volume-weighted DO levels (18m-bottom) increased slightly over the period of record, DO concentrations in 2000-2003 were as low as 3.0 mg O_2/L (2001) at the end of the summer stratified period, and were consistently below the LSEMS objective of 5 mg/L.

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INTRODUCTION

Phosphorus (P) is an essential element for all organisms. It is an important plant nutrient and is the principal limiting factor for biotic production in freshwater environments. Excessive P concentrations in freshwaters stimulate the growth of algae and aquatic macrophytes which can have negative effects on aquatic ecosystems. Increased algal growth may result in decreases in water clarity and potential taste and odour issues. Decomposition of dead and decaying plant matter in the deepest parts of lakes consume oxygen, which may limit the volume of deep, cold water habitat available for popular sport fish such as lake trout and whitefish which require temperatures in the range of 7-12°C (Evans *et al*., 1996). For example, DO levels of 7 mg/L or higher are reportedly necessary for "minimal impairment of activity" in salmonids and 5 mg O_2/L has been associated with a 50% reduction in maximum potential activity in lake trout (Gibson and Fry, 1954; Davis, 1975). Lake Simcoe does not currently support a naturally reproducing coldwater fishery, and the goals of the LSEMS Program include reducing P loads to the lake and restoring self-sustaining, naturally reproducing coldwater fish populations.

The purpose of this report is to update previous publications on the water quality of Lake Simcoe (*e.g*. Nicholls, 2001a) with the most currently available data from the Ontario the of Environment. Trends and patterns in 5 trophic indicator parameters of water quality, namely total phosphorus (TP), chlorophyll *a*, phytoplankton biovolume, Secchi disk visibility and DO concentration have been reported on, and the previous data record extended to 2003. Additional chemical parameters, including chloride (Cl), and hypolimnetic iron (Fe) are also discussed as they provide information on the extent of urbanization and potential for sediment-P release, respectively.

METHODS

The Ontario Ministry of the Environment currently monitors water quality in Lake Simcoe at 8 in-lake sites and 3 municipal water intakes (Figure 1), although up to 12 in-lake sites and 4 municipal water intakes have been monitored for varying periods over the past 2+ decades. Water quality (untreated) at the 3 municipal drinking water intake stations (Beaverton, Keswick, Sutton) is assessed on an approximately weekly basis throughout the year; municipal drinking water is obtained from Lake Simcoe *via* pipes located at varying depths and distances offshore

(Table 1). The 8 in-lake stations that are currently sampled (C1, C6, C9, K39, K42, K45, E51 and S15) are visited approximately twice monthly through the ice-free season (May-October), and composite water samples from the euphotic zone (lower depth determined as 2.5 times the Secchi disk visibility; see below) are analyzed for phytoplankton biovolume and concentrations of TP, chlorophyll *a* and other chemicals using standard methods (OMOE, 1983). In addition, DO profiles with depth and water clarity are assessed at each site concurrent with water quality sampling. Water clarity is evaluated by lowering a black and white Secchi disk through the water column and its visibility is recorded as the water depth (meters) at which the disk is no longer visible to an observer at the water surface. Dissolved oxygen is measured with a hand-held YSI probe at 1 m depth intervals to the lake bottom in order to evaluate changes in oxygen with depth and over time.

Table 1. Depth and distance off-shore of municipal water intake pipes.

	Depth of intake pipe (m)	Distance off-shore (m)
Sutton	4.0	180
Beaverton	4.6	986
Keswick	8.5	365

Average values for each parameter, excluding DO were calculated over the ice-free season at each in-lake site, whereas annual (June 1 – May 31 hydrologic year) averages were calculated at the outflow and municipal intake stations. Outlier data points were identified following Nicholls (2001a). Monthly dissolved oxygen depth profiles were produced for 2000-2003, inclusive at the deepest lake site, located in Kempenfelt Bay (K42), consistent with earlier LSEMS reports (*e.g.* Nicholls, 1998; 2001a). In addition, depth profiles of TP and Fe are presented at station K42 in Kempenfelt Bay in order to evaluate the potential for P release from sediment under anoxic conditions.

Dissolved oxygen depletion rates in the hypolimnion at station K42 in Kempenfelt Bay were determined for the years 2000-2003, inclusive, by least-squares regression of mean hypolimnetic dissolved oxygen concentration *versus* calendar date over the stratified period (June-October) (Evans *et al.*, 1996). Rates were volume-weighted and standardized to a temperature of 4°C, consistent with earlier reports (*e.g.* Nicholls, 2001a).

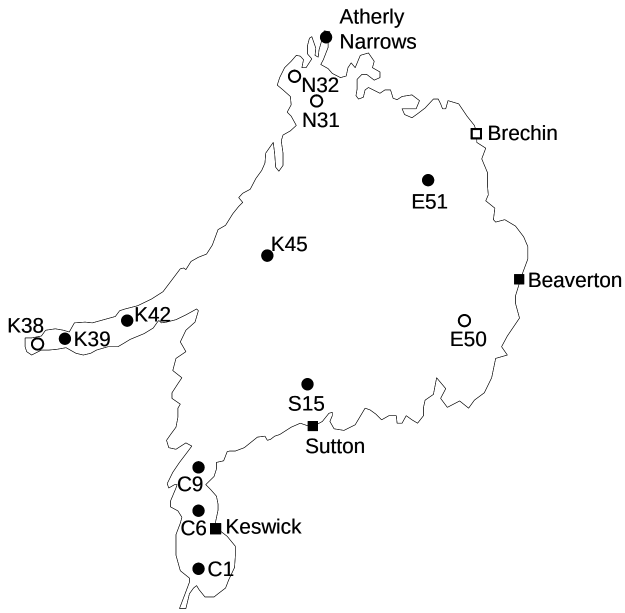


Figure 1. Lake water quality monitoring stations (•) and municipal water intake sites (■). Open symbols (ο, □) indicate sites that are not currently sampled, but for which historical data exist.

Data treatment and statistics

In order to determine whether different lake stations exhibited similar patterns in water quality, monthly or annual (May-October) averages were converted to Z-scores using equation (1), and plotted in units of standard deviation relative to the long-term mean.

$$\text{Z-score} = \frac{(\text{value} - \text{mean})}{\text{standard deviation, SD}} \tag{1}$$

Pearson correlation analysis of Z-scores was used to determine whether different lake stations exhibited similar temporal patterns. The Pearson product moment correlation coefficient (r) varies from -1 to 1 and reflects the extent of a linear relationship between two data sets, where $r = 1$ indicates a perfect positive relationship and $r = 0$ indicates no linear relationship. Temporal trends in water quality parameters over the entire data record were evaluated for statistical significance ($p < 0.05$) by least-squares regression (mean annual concentrations against years). Bonferroni corrected p values were calculated and are included as a footnote in the tables. The Bonferroni correction is a method of adjusting the p value used to assess significance, in order to reduce the probability of Type 1 errors (i.e. rejecting the null hypothesis when it is actually true; Legendre and Legendre 1998). However, since there is debate over the validity of this procedure in ecological studies (Cabin and Mitchell, 2000; Moran, 2003), we elected to assess significance based on the unadjusted values. Data were tested for deviations from normality and homogeneity of variance. Based on these tests, the phytoplankton biovolume data were $\log_{10}$ transformed for the analysis. Relationships between average TP, chlorophyll *a* and algal biovolume were examined using Pearson and Spearman rank correlations.

RESULTS AND DISCUSSION

Total phosphorus

Average total P concentrations over the ice-free period (May-October) during the most recent years of record (2000-2003) ranged from a high of 22.7 ± 9.8 µg/L at C1 in Cook's Bay to a low of 13.1 µg/L at the easternmost (E51) and southernmost (S15) main lake stations. Average TP concentrations in 2000-2003 were generally lower than earlier values, and significant ($p < 0.05$) declines in TP were noted at C1, C6 and K39 over the past 20 years of monitoring (Table 2).

Table 2. May-October average (± 1 standard deviation) total phosphorus concentrations (µg/L) at currently monitored in-lake stations in Lake Simcoe.

Station	*Average TP ± SD (µg/L) (1980-83)*	*Average TP ± SD (µg/L) (2000-03)*	*Change in TP (µg/L/year); significance**
C1	29.7 ± 4.7	22.7 ± 9.8	-0.41, p = 0.02
C6	22.6 ± 3.0	16.5 ± 2.1	-0.31, p = 0.01
C9	18.0 ± 3.5	14.4 ± 1.1	-
K39	20.4 ± 6.0	13.4 ± 1.1	-0.29, p = 0.01
K42	17.4 ± 3.0	13.4 ± 2.2	-
K45	16.5 ± 3.7	13.4 ± 1.3	-
E51	13.1 ± 2.0	13.1 ± 0.3	-
S15	na	13.1 ± 1.7	-

**The Bonferroni corrected* p *value given 8 comparisons is 0.006; trends were evaluated over the entire data record (1980-2003).*

Nicholls (2001a) similarly noted significant declines in TP at stations C1, C6 and K39 (as well as C9 and N31) ranging from 0.4 µg/L/year (K39) to 1.0 µg/L/year (C6) over the period ending in 1999. It should be noted that significant declines in TP only occurred at stations located close to shore and stream confluences, in the highly agricultural and urbanized Cook's Bay (C1, C6) and Kempenfelt Bay (K39), respectively, where TP levels are consistently higher than at other stations (Table 2). In contrast, while TP levels at the main lake sites (K45, E51, S15) and bay outlets (K42, C9) are currently slightly lower than at the onset of monitoring, trends over time were not statistically significant, in part due to large variation in TP between years (Figure 2).

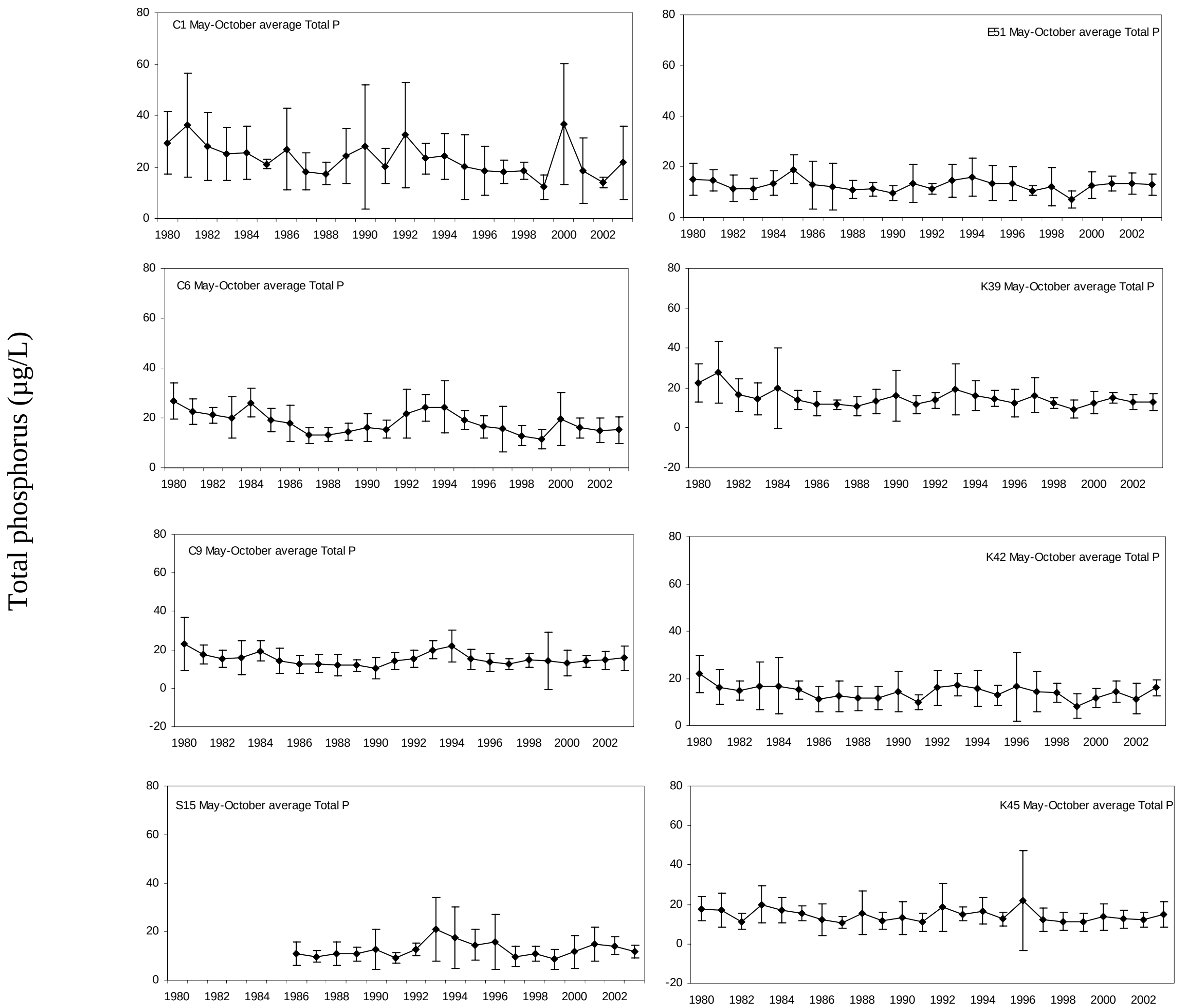


Figure 2. Average (May-October) total phosphorous concentration (µg/L; ± 1 standard deviation) at currently monitored stations in Lake Simcoe.

Consistently high TP levels at station C1 are related to extremely high concentrations of P (generally >100 µg/L) in tributaries of the Holland River, and in discharge from the Bradford Pump House (Scott *et al.*, 2001; Winter *et al.*, 2002). However, TP concentrations generally decline with distance northward from shore, and are only slightly higher at station C9 (located near the mouth of Cook's Bay) compared to other lake stations (Table 2). Johnson and Nicholls (1989) suggested that close to half of the P input to Cook's Bay is retained in sediment, which may explain the rapid decrease in TP with distance from the south shore inflows (Figure 1). In general, TP levels across lake stations decrease in the order of Cook's Bay (C1, C6, C9) > Kempenfelt Bay (K39, K42) > main lake stations (E51, S15, K45; Figure 1).

Although TP concentrations differed between stations, inter-annual patterns in TP were coherent across the lake. Only station C1 exhibited a unique temporal pattern of TP. Correlations between lake stations excluding C1, ranged from r = 0.31 (E51 *vs*. K39) to r = 0.81 (C6 *vs*. C9). In general, TP concentrations were highest in the early 1980s, declined through the mid-1980s, peaked around 1994/95 and then declined again to a minimum in 1999 (Figure 3). Average concentrations in the most recent period of record (2000-2003) are similar to the long-term average (Figure 3). Large inter-annual variations in P concentration, which are coherent across lake stations, suggest the influence of large-scale factors such as climate. Certainly, year-to-year variations in stream flow (which is a function of climate) result in large fluctuations in total P loading to the lake *via* the tributaries, and tributary export is the most variable component of the total P budget for the lake (Nicholls, 2001b). Comparisons between annual phosphorus loadings and lake concentrations will be included in a report planned for release in 2005.

Table 3. Annual (June-May) average (± 1 standard deviation) total phosphorus concentrations (µg/l) at municipal water intake stations. Rate of decline computed from 1982 to 2002.

Station	*Average TP ± SD (µg/L) (1982-1984)*	*Average TP ± SD (µg/L) (2000-2002)*	*Rate of TP decline (µg/L/year); significance**
Sutton	17.2 ± 1.0	11.8 ± 0.7	-0.24; p = 0.003
Beaverton	21.4 ± 4.1	15.4 ± 2.6	-0.35; p = 0.008
Keswick	19.8 ± 2.0	13.2 ± 1.5	-0.34; p = 0.01

**The Bonferroni corrected* p *value given 3 comparisons is 0.02.*

Similar to the main lake stations, TP concentrations at the 3 municipal intake stations were highly variable between years, but declined significantly over the entire period of record (1982/84 – 2000/02) (Figure 4; Table 3). Nicholls (1995) reported similar significant declines in

TP at the Sutton and Beaverton stations between 1982 and 1992. Although annual rates of change were small (0.24-0.35 µg/L/year), TP concentrations at the intakes declined by between 5 µg/L and 7 µg/L from 1982 to 2002 (Table 3).

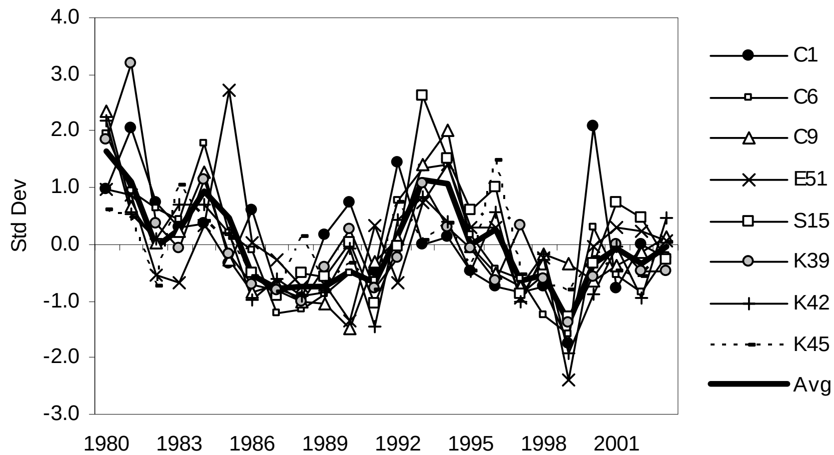


Figure 3. Standardized average (May-October) TP concentrations (Z-score) at the currently monitored in-lake stations in Lake Simcoe. The y-axis is in units of standard deviation relative to the long-term mean; *e.g.* a value of 2 indicates that the TP concentration in that year was 2 standard deviations higher than the 20-year mean. 'Avg' indicates the average Z-score for all lake stations excluding C1, which exhibited a distinct temporal pattern.

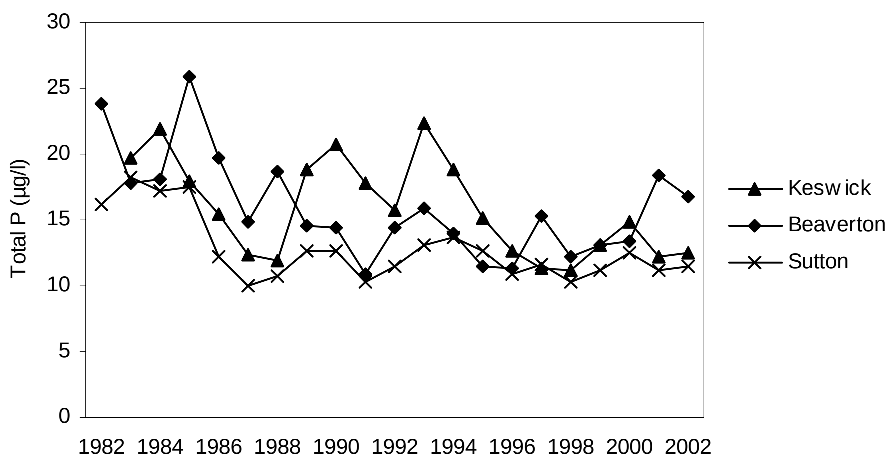


Figure 4. Annual average TP concentrations (µg/L) at the Keswick, Beaverton and Sutton municipal water intake stations.

There was no significant change in TP concentration at the lake outflow over the 1983-2003 monitoring period, although average concentrations in 2000-2003 (10.2 ± 0.7 µg/L) were slightly lower than in 1983-1986 (average 13.2 ± 2.6 µg/L) (Figure 5). Higher TP levels were reported at a nearby outflow station in the early 1970s (~20 µg/L) and early 1980s (~15 µg/L) (Nicholls, 1995), and if these early values are considered then the decline in TP at the outflow has been significant. Total P concentrations at the lake outflow are always lower than at the main-lake stations possibly due to cumulative retention of P within sediment during passage of water through the lake system (Johnson and Nicholls, 1989).

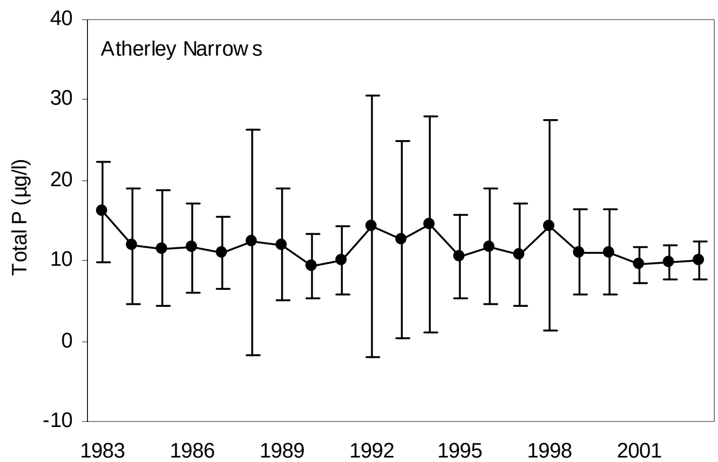


Figure 5. Annual average TP concentration (±1 standard deviation; µg/L) at the Lake Simcoe outflow near the Atherley Narrows.

Table 3. May-October average (± 1 standard deviation) chlorophyll *a* concentration (µg/L) at currently monitored in-lake stations.

Station	*Average chl* a *± SD (µg/L) (1980-83)*	*Average chl* a *± SD (µg/L) (2000-03)*	*Change in chl* a *(µg/L/year); significance**
C1	5.5 ± 2.5	5.7 ± 2.9	-
C6	3.5 ± 0.5	3.0 ± 1.3	-0.05; p = 0.04
C9	2.4 ± 0.5	3.1 ± 0.7	-
E51	1.8 ± 0.3	2.1 ± 0.4	-
S15	2.1 ± 0.6	2.2 ± 1.0	-
K39	3.0 ± 0.8	3.0 ± 0.6	-
K42	2.2 ± 0.5	3.0 ± 0.7	-
K45	1.8 ± 0.2	2.5 ± 0.5	-

**The Bonferroni corrected* p *value given 8 comparisons is 0.006; trends were evaluated over the entire data record (1980-2003).*

Chlorophyll* a*, phytoplankton biovolume and water clarity

Similar to TP, chlorophyll *a* levels decreased in the order of Cook's Bay (C1, C6, C9) > Kempenfelt Bay (K39, K42) > open-lake stations (E51, S15, K45; Table 3), and maximum chlorophyll *a* levels always occurred at the most near-shore bay sites (C1 and K39). There was a positive correlation between average TP and average chlorophyll *a* among lake stations (Table 4). With the exception of station C6, average chlorophyll *a* concentrations did not change significantly over time, and were similar or slightly higher in 2000-2003 compared to 1980-1983, in part due to the large amount of variation in annual values (Figure 6). Similar to TP, temporal patterns in average chlorophyll *a* were coherent among stations across the lake (Figure 7), and correlations between stations ranged from r = 0.23 (E51 *vs*. K39) to r = 0.78 (K39 *vs*. K42).

Table 4. Pearson and Spearman Rank correlations between 1980 to 2002 average chlorophyll *a* (μg/L), TP (μg/L), phytoplankton biovolume (mm^3/m^3) and Secchi disk visibility (m) for currently monitored open-lake stations.

	Correlation coefficients	
Variables	*Pearson*	*Spearman Rank*
Chlorophyll *a* & TP	0.99	0.88
Biovolume & Chlorophyll *a*	0.93	0.98
Biovolume & TP	0.93	0.95
Secchi depth & Biovolume	-0.91	-0.73
Secchi depth & TP	-0.96	-0.80

Phytoplankton biovolume declined significantly at the majority of sites (Figure 8) from highs of 334 – 1387 mm^3/m^3 in 1980-1982 to lows of 179 - 390 mm^3/m^3 in 2000-2002 (Table 5). A consistent pattern was observed across sites, which included peaks in 1985, 1990 and 1994, a decline between 1994 and 1998, and relatively stable populations in the final period of 1998-2002 (Figure 8). The decline in biovolume which began in the mid 1990s may be related to the establishment of zebra mussels in the lake in 1995. While year-to-year changes in phytoplankton biovolume did not appear to mirror changes in chlorophyll *a* (Figures 7 & 8), a positive correlation between chlorophyll *a* and phytoplankton biovolume was observed between 1980 to 2002 averages for the open lake stations (Table 4). Phytoplankton biovolume was similarly highly correlated with average TP concentration, with maximum algal densities occurring at sites with high TP.

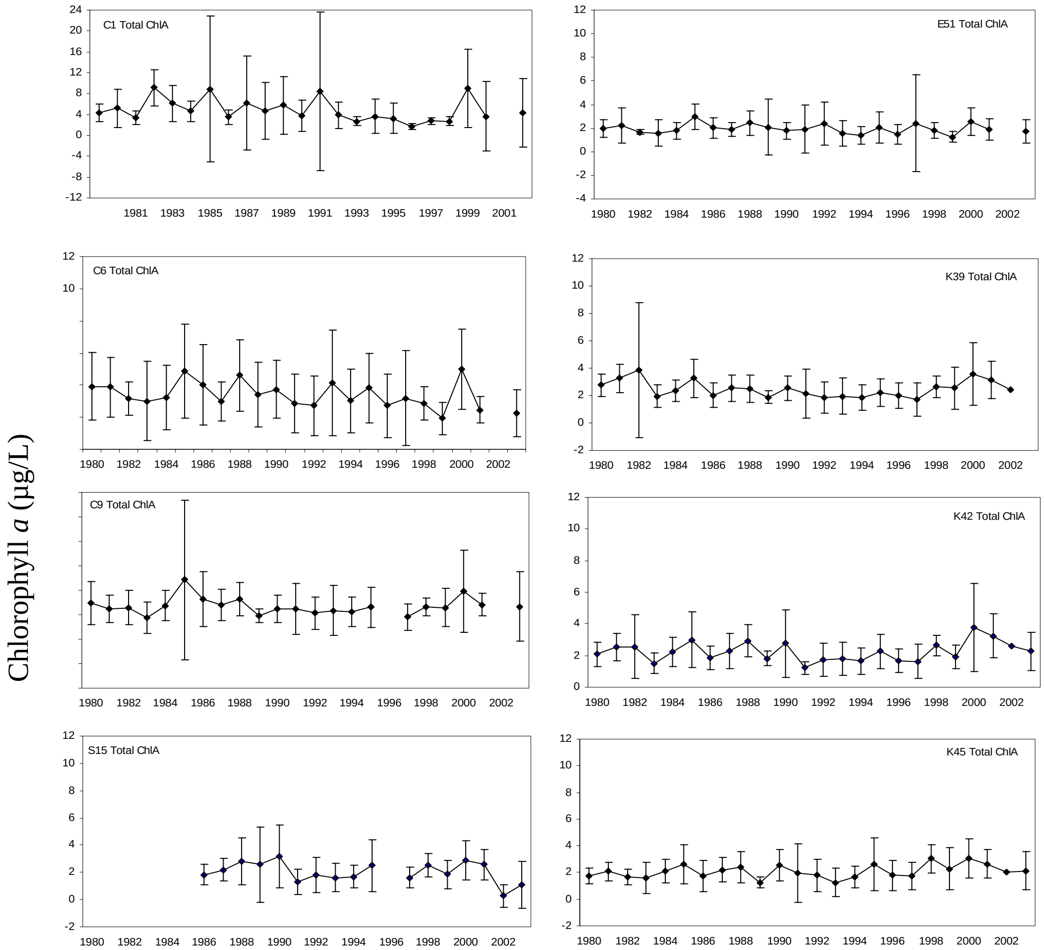


Figure 6. Average (May-October) chlorophyll *a* concentration (µg/L; ±1 standard deviation) at currently sampled stations in Lake Simcoe.

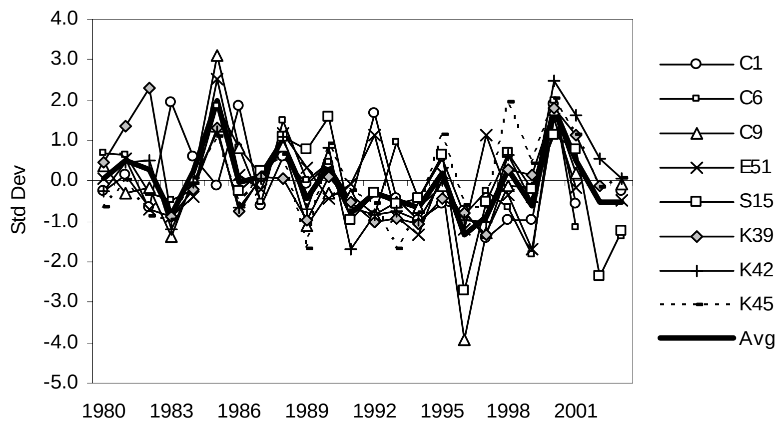


Figure 7. Standardized average (May-October) chlorophyll *a* concentrations (Z-score) at in-lake stations in Lake Simcoe. The y-axis is in units of standard deviation relative to the long-term mean; *e.g*. a value of 2 indicates that the chl *a* concentration in that year was 2 SDs higher than the 20-year mean. 'Avg' indicates the average Z-score for all lake stations excluding C1, which exhibited a distinct temporal pattern.

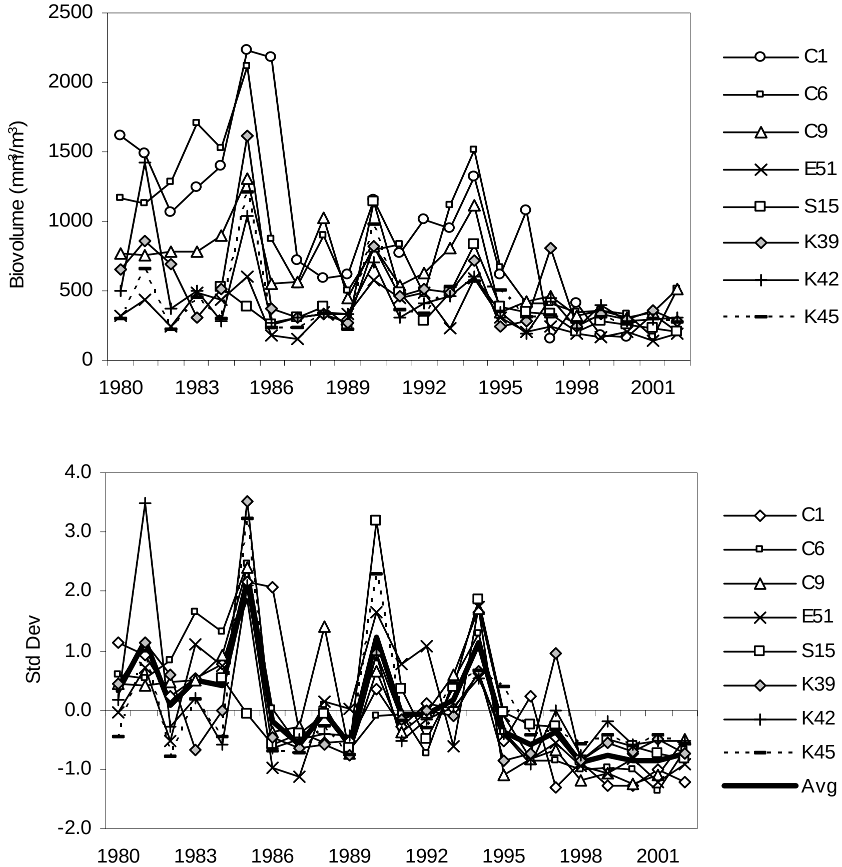


Figure 8. Average (May – October) algal biovolume (mm^3/m^3) (upper) and standardized average (Z-scores) algal biovolume (lower) at main lake stations in Lake Simcoe, 1980-2002. 'Avg' is the average Z-score of all stations excluding C1.

Table 5. May-October average (± 1 standard deviation) phytoplankton biovolume (mm^3/m^3) at currently monitored stations in Lake Simcoe from 1980 to 2002.

Station	*Average biovolume ± SD (mm^3/m^3) (1980-82)*	*Average biovolume ± SD (mm^3/m^3) (2000-02)*	*Change in log-transformed biovolume (mm^3/m^3/yr); significance**
C1	1387 ± 287	238 ± 88	-0.04; p < 0.001
C6	1191 ± 83	335 ± 173	-0.03; p < 0.001
C9	769 ± 11	390 ± 112	-0.02; p < 0.001
E51	334 ± 95	179 ± 30	-0.02; p = 0.02
S15	na	229 ± 63	-
K39	739 ± 110	312 ± 42	-0.01; p = 0.03
K42	762 ± 576	297 ± 9	-0.01; p = 0.03
K45	388 ± 235	277 ± 20	-

**Biovolume data were log_{10} transformed for the analyses. The Bonferroni corrected p value given 8 comparisons is 0.006; trends were evaluated over the entire data record (1980-2002).*

Significant increases in water clarity were observed over the 20+year monitoring period, and Secchi disk visibility increased markedly at all sites beginning in the early 1990s (Figure 9). Secchi disk depths were fairly consistent between 1980 and 1991, and ranged from 1.9 ± 0.3 m (C1) to 4.5 ± 0.4 m (K45) compared with 2.5 ± 0.1 m (C1) to 6.9 ± 0.8 m (E51) in 2000-03.

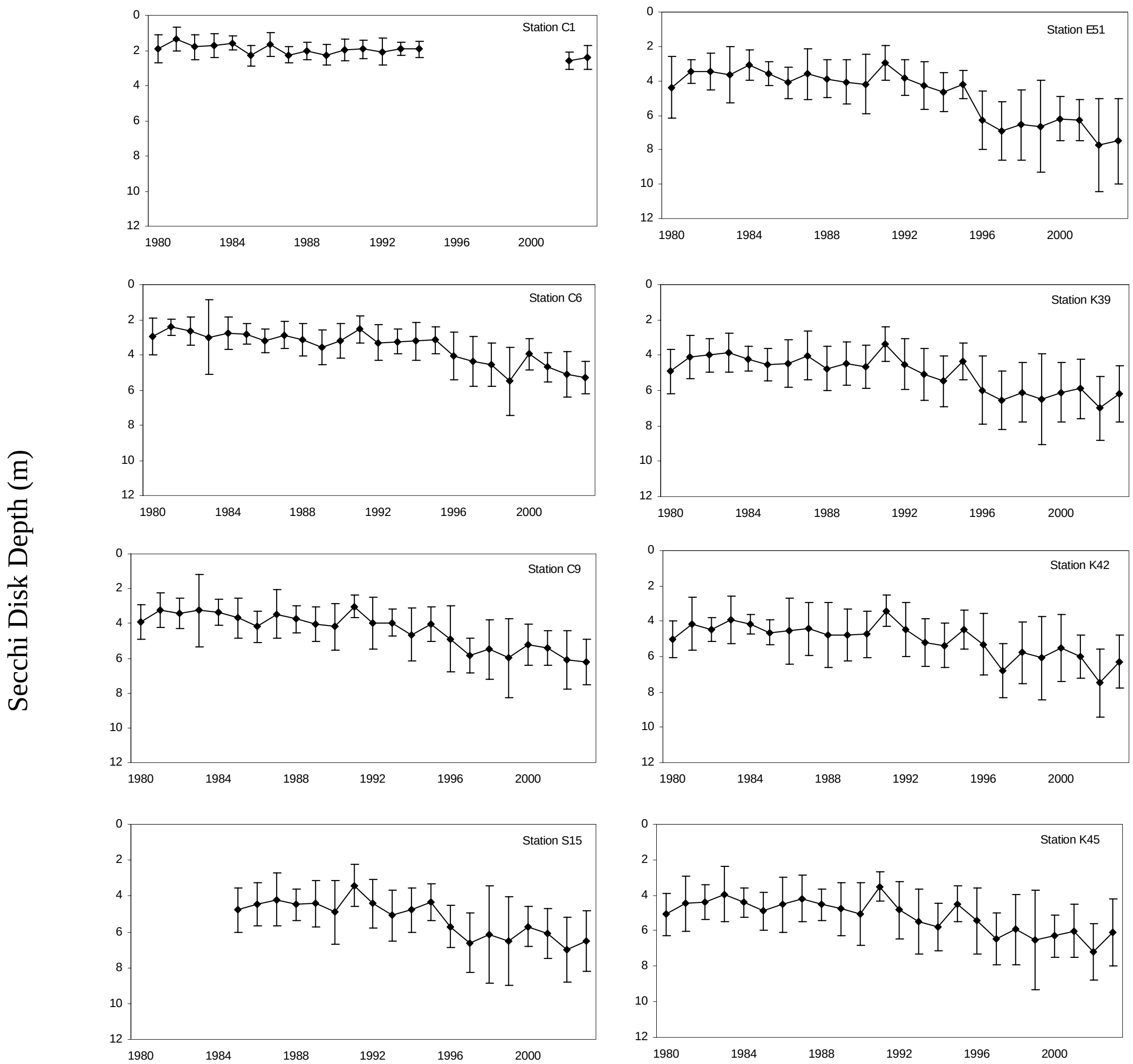


Figure 9. May-October mean (±SD) Secchi disk visibility (m) at currently monitored stations in Lake Simcoe.

With the exception of C1, temporal patterns of Secchi disk depth were highly coherent among lake stations, and correlation coefficients between sites (*r* values) ranged from 0.86 (C6 *vs*. K45) to 0.97 (K45 *vs*. K39; Figure 10).

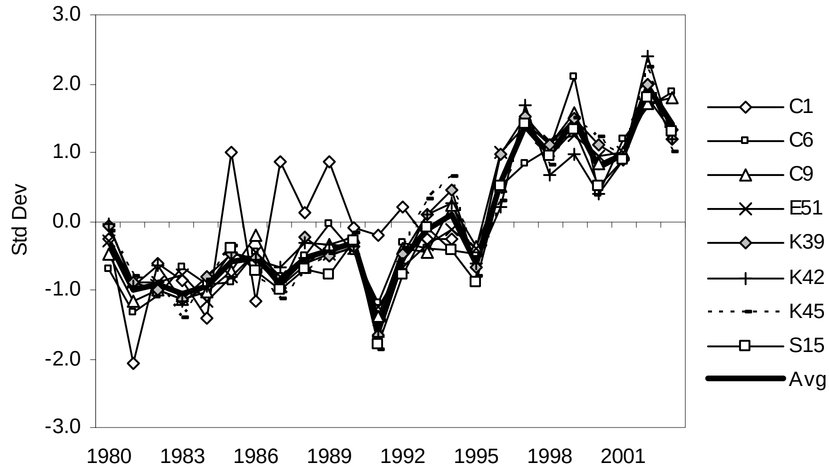


Figure 10. Standardized average Secchi disk visibility (Z-score) at in-lake stations. 'Avg' indicates the average Z-score for all lake stations excluding C1, which exhibited a distinct temporal pattern. Positive values are calculated for years in which the Secchi disk depth was *greater* than the long-term (i.e. 20 year) average, and therefore indicate increased water clarity whereas negative values indicate lower-than-average Secchi disk visibility.

Increased water transparency in Lake Simcoe has been previously associated with the establishment of zebra mussels in the lake around 1995 (Nicholls, 1998). Certainly, algal biovolume appeared to be both higher and more varied pre-1995, and trends toward increased water transparency (Secchi disk depth) began at most stations in the early to mid 1990s (Figure 10). Zebra mussels clear the water by filtration of particles from the surrounding water, and are expected to have a greater impact at near-shore shallow sites than at deeper sites due to the higher ratio of zebra mussel substrate to overlying water volume in shallow zones (Nicholls, 1998). However, highly coherent temporal patterns of Secchi disk visibility among lake stations that vary greatly in depth (3 m at C1 to 39 m at K42) suggest that the lake is relatively well mixed, and that zebra mussels may affect the transparency of water across the lake. Nicholls (2001a) analyzed water clarity data from 4-year periods pre- and post-zebra mussel invasion using step trend detection and found significant increases in water clarity at all stations post-zebra mussel invasion.

Dissolved oxygen

One of the major goals of the LSEMS program is to restore lake water quality to the extent that a self-reproducing lake trout population may be established, and an interim end-of-summer DO objective of 5 mg/L (volume-weighted, 18m-lake bottom zone) was adopted to improve cold water fish habitat. Monthly trends in DO were similar in 2000-2003 compared to previous years (*e.g*. Nicholls, 2001a), and concentrations in the deep water zone at station K42 declined from initial values of 10-12 mg/L at spring overturn to a minimum of 1.8 mg/L in October 2001 (Figure 11). Corresponding rates of DO depletion ranged from 1.4 to 2.2 g O_2/m^3/month at ambient temperature and 1.0 to 1.5 g O_2/m^3/month when normalized to 4°C. Depletion rates in the most recent period of record are similar to average values in the early 1980s, but are lower than rates in the mid 1990s (Figure 12). Nicholls (2001a) similarly noted that DO depletion rates had increased through the 1980s, but had generally decreased through the 1990s.

The most direct way to assess changes in deep-water DO availability relative to the LSEMS objective is to compare end-of-summer, volume-weighted DO concentrations in the 18m-bottom zone across the entire data record. In the most recent 2000-2003 period, late summer bottom zone DO levels ranged from 3.8 mg/L in 2001 to 4.6 mg/L in 2003, and were consistently below the LSEMS objective (Figure 13). In fact, deep water DO levels declined to below 5 mg/L by late summer in almost every year of the 20-year record (Figure 13). However, there was a small but significant increase in minimum deepwater DO concentration over time (0.08 mg O_2/L/year; $R^2 = 0.20$, $p = 0.04$). While we cannot attribute causality, these results may indicate some improvement in deepwater DO concentrations as a consequence of decreased phytoplankton biovolume. There was a significant negative correlation between annual average phytoplankton biovolume and end-of-summer DO at station K42 ($r = -0.5$). Average volume-weighted temperatures in bottom waters in many years were near the upper limit of the optimal thermal habitat range for lake trout (8-12°C), white fish and herring (7-11°C) (Evans *et al*., 1996; Figure 13).

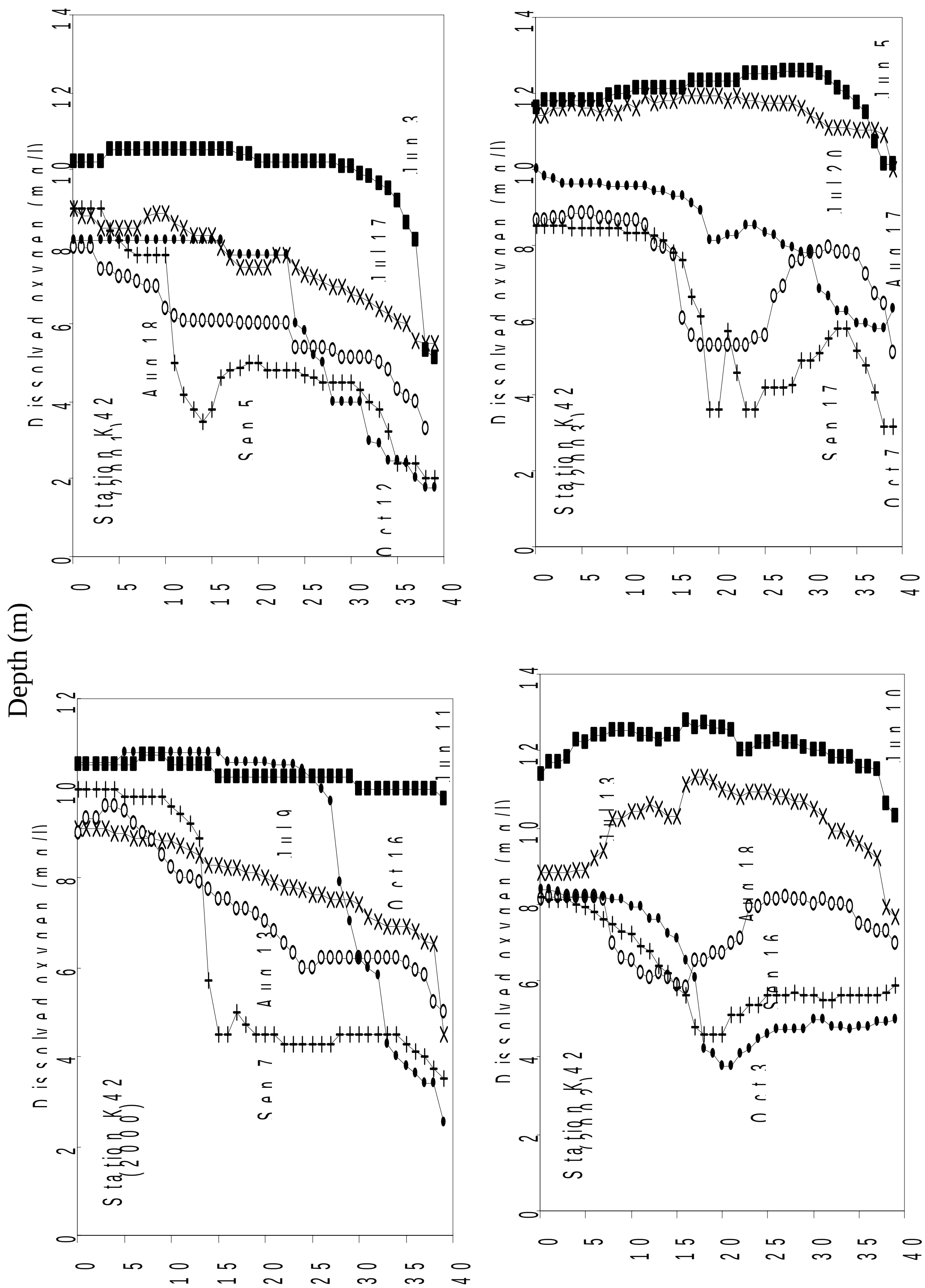


Figure 11. Progressive decrease in dissolved oxygen concentration (mg/l) over the summer-fall periods of 2000 – 2003. Dissolved oxygen concentrations with depth (1 m intervals) are presented for the deepest region of Lake Simcoe in Kempenfelt Bay (Station K42, depth = 39m).

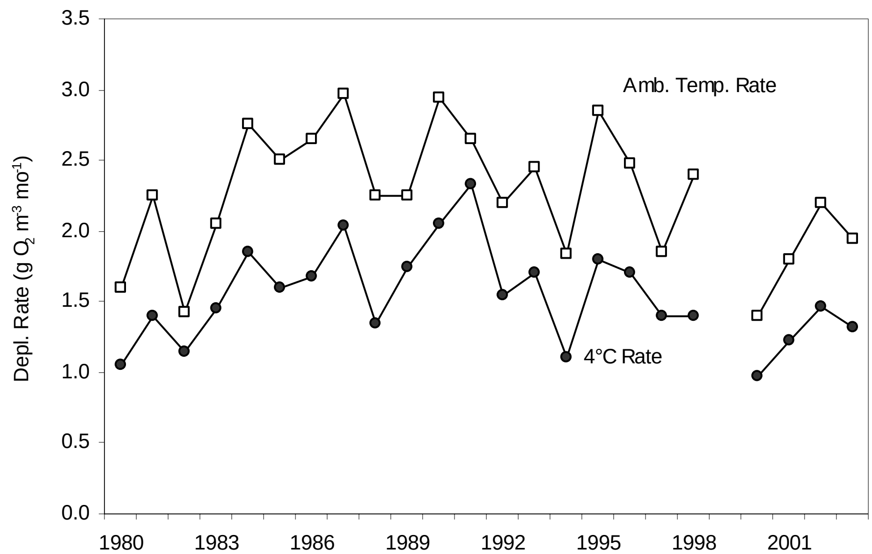


Figure 12. Volume-weighted dissolved oxygen depletion rate in the 18m-bottom zone of Lake Simcoe at station K42 for the stratified periods (typically mid-June to mid-September) of 1980-2003. Rates at ambient (measured) temperature and after normalization to 4°C, respectively, are presented for comparison. Values for 1999 are not presented due to instrument failure and anomalously high values during that year (Nicholls, 2001a).

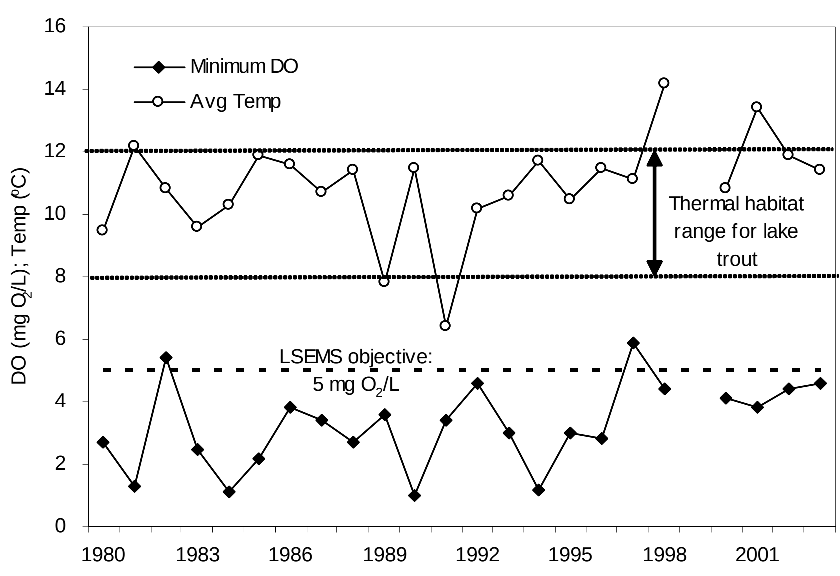
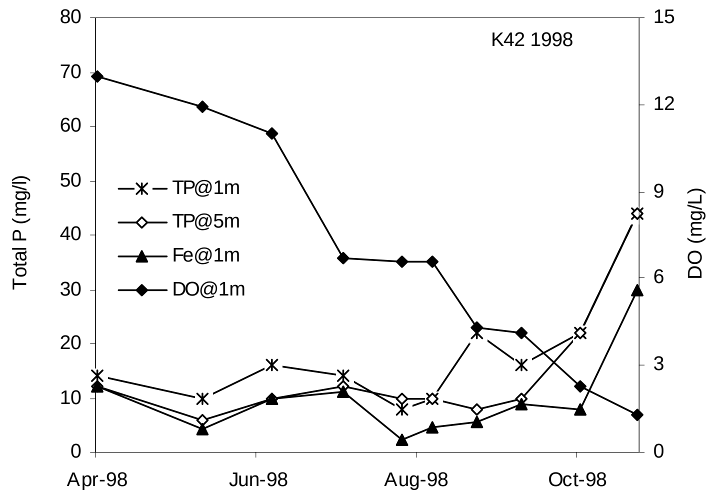


Figure 13. Minimum (end-of-summer) volume-weighted DO concentration and average temperature in the 18m-bottom zone of Lake Simcoe at station K42.

Potential for P release from sediment

Under oxygen-limiting conditions, P (as well as iron) is released from sediment, and can be an important internal source of P in lakes with high P levels in sediment (Rossi and Premazzi, 1991; Penn *et al*., 2000). These inputs can persist following reductions in external loading and effectively delay the recovery of lake water quality. Nicholls (1995) suggested that release of P from sediment could contribute to TP levels in Lake Simcoe, and could be particularly significant at deep regions of the lake that become oxygen limited, such as Kempenfelt Bay. In order to investigate whether P release from sediment was occurring in Lake Simcoe, TP and Fe concentrations with depth were examined at station K42 in 2 years (1998, 1999) for which depth-specific water chemistry data were available.

Total P concentrations in bottom waters increased over the period of summer stratification to a maximum of ~50 µg TP/L, coincident with declining DO (Figure 14). Iron levels also increased over the stratified period, suggesting that higher TP levels were due to redox-related release from sediment. Nicholls (1995) reported end-of-summer TP concentrations in the range of 40 to 100 µg/L, 1 m above the bottom at deepwater stations K42 and K45 when DO levels were below 2 mg/L. Importantly, the TP level in bottom waters increased to 50 µg/L in October 1999 even though DO levels remained relatively high at 4 mg/L (Figure 14). If a net increase of 35 µg/L (i.e. 50 µg/L minus initial level of 15 µg/L) is assumed, and extrapolated to the lake volume corresponding to the 35-39m zone (38 x 10^6 m^3; MNR bathymetry data), a total net P release of ~1350 kg (1.35 metric tones) is calculated. While 1.35 tonnes represents only a small fraction (~1%) of the total P pool in the lake (~125 tonnes TP), it is similar in magnitude to the average annual P load from individual sewage treatment plants (Scott et al., 2001). Furthermore, the relative importance of internal P loading will increase over time as external loads to the lake are reduced.



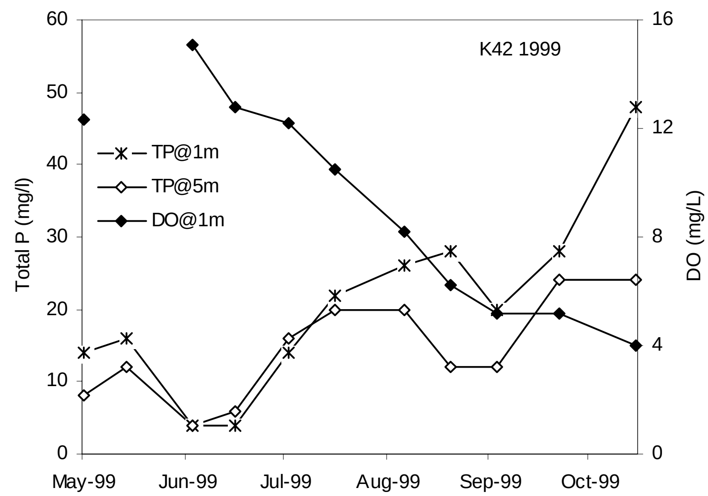


Figure 14. Progressive increase in TP and Fe concentrations and loss of DO at depths of 1m and 5m above the lake bottom; Station K42 1998 - 1999.

Chloride

Chloride levels in Lake Simcoe have increased consistently over the past 20 years, at rates of 0.65- 0.78 mg Cl/L/year at the municipal water intakes (Figure 15). A similar rate of Cl increase occurred at the lake outflow (0.78 mg/L/year) and the current concentration at Atherley (~34 mgCl/L) is 3 times the level measured in the early 1970s (~11 mg Cl/L; Nicholls, 1995). While current Cl levels in Lake Simcoe are unlikely to pose a biological threat, they are a useful indicator of the extent of urbanization in the watershed, and demonstrate that substantial runoff from urban surfaces is entering the lake. In addition, consistent rates of increase of Cl across lake stations indicate a relatively well-mixed system, which bodes well for lake water quality modeling.

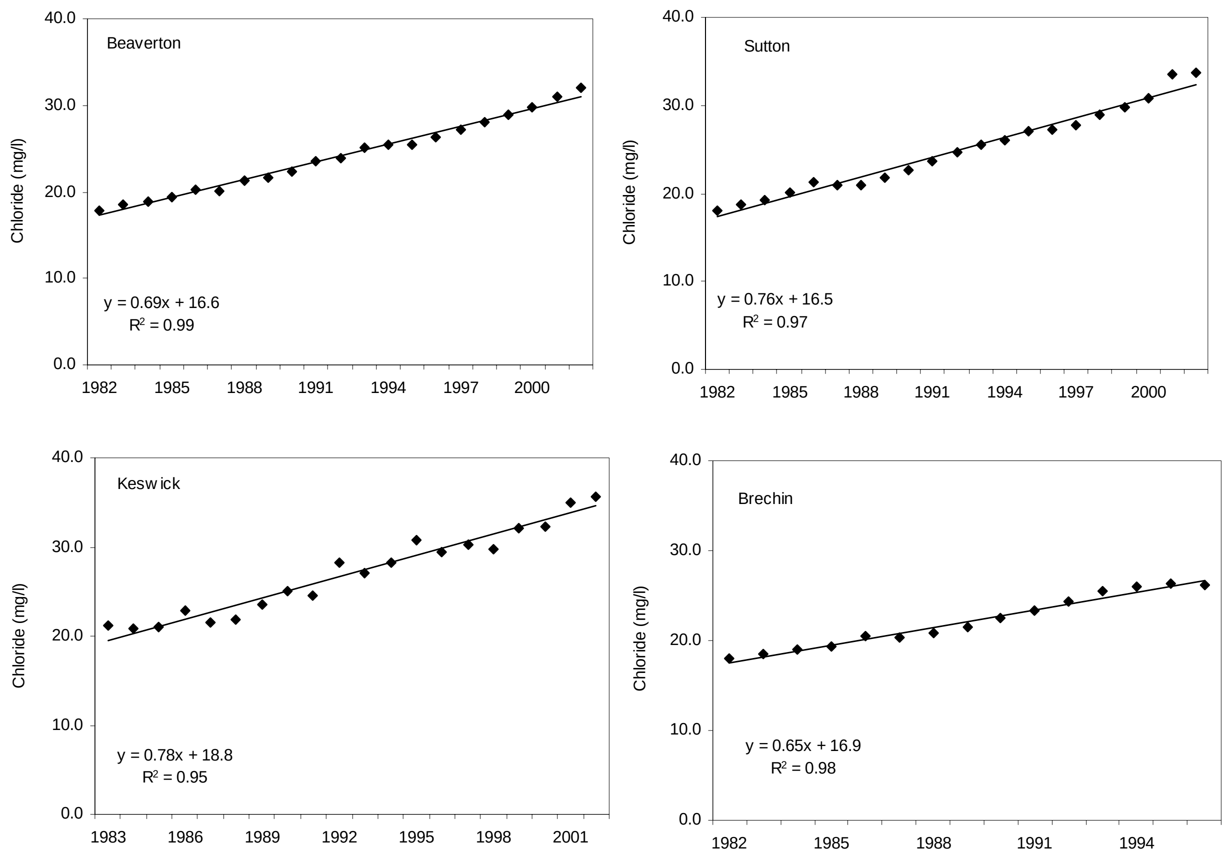


Figure 15. Increase in annual average chloride concentration (mg/L) at municipal water intakes on Lake Simcoe.

Summary

In summary, changes in water quality in Lake Simcoe have been varied over the past 20+ years of monitoring. While there were definite increases in water clarity and significant declines in phytoplankton biovolume across the lake, TP decreased significantly at only 3 of 8 lake stations, and at all 3 municipal water intake plants. Chlorophyll *a* levels at lake stations were highly variable within years and only declined at 1 of the 8 stations over the ~20 year monitoring period, contrary to marked decreases in phytoplankton biovolume. While variability in TP and chlorophyll *a* levels at individual stations may have obscured long-term trends, patterns over time were generally coherent among stations. There were strong correlations between TP, chlorophyll *a*, phytoplankton biovolume and water clarity among stations. Volume-weighted, end-of-summer DO levels in the 18m-bottom zone continue to be lower than the LSEMS objective of 5 mg/L, and indicate reduced habitat volume for cold water fish.

Further work is recommended to understand the drivers of large inter-annual changes in water quality and the relationships (or lack thereof) between climatic variations, P inputs to the lake, and resultant water quality. Results presented in this report highlight the importance and necessity of long-term data sets for identifying trends in water chemistry, and for recognizing the influence of factors such as climate and invasive species, which may obscure expected changes in water quality due to changes in P loading to the lake.

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Mark and Vanessa Ledlie did the lake sampling from 2000 to 2003, and measured temperature, dissolved oxygen and Secchi disk visibility. Staff of the Laboratory Services Branch, Ontario Ministry of the Environment performed chemical analyses. Lynda Nakamoto and Lucja Heintsch processed phytoplankton samples and maintained the phytoplankton database, and Ron Ingram entered temperature, dissolved oxygen and Secchi depth data from 2000. Keith Somers gave advice on statistical methods. Shaun Watmough, Peter Dillon, Wolfgang Scheider, Ken Nicholls, Ron Griffiths and Lynda Nakamoto provided helpful comments on the draft of this report.

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LAKE
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APPENDIX A

LSEMS Implementation Program
Technical Reports and Bulletins

LAKE
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LSEMS Implementation Program
Technical Reports and Bulletins

Technical Reports

- ☐ Imp A1 **Harrington and Hoyle Limited**. **1992** *Lower Holland River Erosion Control Study*

- ☐ Imp A2 **Peat, G. and M. Walters. 1994** *Lake Simcoe Tributary Monitoring Data Report: 1982-1986*

- ☐ Imp A3 **Beak Consultants Limited. 1995** *Development and Implementation of a Phosphorus Loading Watershed Management Model for Lake Simcoe*

- ☐ Imp A4 **LSEMS. 2001** *LSEMS Phase II: Annual Water Balance and Phosphorus Loading for Lake Simcoe 1990-1998*

- ☐ Imp A5 **LSEMS. 2001** *Phosphorus Loading to Lake Simcoe, 1990-1998 Highlights and Preliminary Interpretation in Historical and Ecosystem Contexts*

- ☐ Imp B1 **Limnos Limited. 1987** *The Benthic Alga* Dichotomosiphon tuberosus *in Lake Simcoe-1986*

- ☐ Imp B2 **Beak Consultants Limited. 1987** *The Predictability of Hypolimnetic Dissolved Oxygen Depletion in Lake Simcoe (Part I)*

- ☐ Imp B3 **Cumming-Cockburn and Associates Limited. 1987** *Estimated Outflow from Lake Simcoe at Atherly, 1982-86*

☐ Imp B4 **Limnos Limited. 1988** *Aquatic Plants of Cook's Bay, Lake Simcoe 1987*

☐ Imp B5 **Limnos Limited. 1988** *Duckweed Harvest from Holland River*

☐ Imp B6 **Limnos Limited. 1988** *Assessment and Control of Duckweed in the Maskinonge River, Keswick, Ontario*

☐ Imp B7 **Johnson and Nicholls. 1989** *The History of Phosphorus, Sediment and Metal Loading to Lake Simcoe from Lake Sediment Records*

☐ Imp B8 **Beak Consultants Limited. 1987** *The Hypolimnetic Oxygen Dynamics in Lake Simcoe: Part 2-Evaluation Using Time Trend and Model Simulation Techniques*

☐ Imp B9 **Limnos Limited. 1990** *Lake Simcoe Hypolimnion Aeration–An Assessment of the Potential for Direct Treatment*

☐ Imp B10 **Hopkins, G.J. and L. Webb. 1990** *Lake Simcoe Nearshore Water Quality Monitoring at Water Supply Intakes, 1982-1989 Data Report*

☐ Imp B11 **Limnos Limited. 1991** *Status in 1990 of the Dominant Benthic Alga* Dichotomosiphon tubersus *in Lake Simcoe*

☐ Imp B12 **Cumming-Cockburn and Associates Limited. 1991** *Estimated Monthly Flows and Exports of Total Nitrogen and Phosphorus from Lake Simcoe* at Atherly

☐ Imp B13 **Nicholls, K. H. 1991** *Water Quality Trends in Lake Simcoe 1972-1990 and the Implications for Basin Planning and Limnological Research Needs*

☐ Imp B14 **Hydroflux Engineering. 1992** *Hydrodynamic Computer Model of Major Water Patterns in Lake Simcoe*

- ☐ Imp B15 **Beak Consultants Limited. 1993** *Hypolimnetic Oxygen Dynamics in Lake Simcoe: Part 3–Estimation of Phosphorus Loadings and Evaluation of Empirical Oxygen Models for Lake Simcoe*

- ☐ Imp B16 **Beak Consultants Limited. 1993** *Hypolimnetic Oxygen Dynamics in Lake Simcoe: Part 3–Model Confirmation and Prediction of the Effects of Management*

- ☐ Imp B17 **Nicholls, K. H. 1995** *A Limnological Basis for a Lake Simcoe Phosphorus Loading Objective*

- ☐ Imp B18 **LSEMS. 1998** *Lake Simcoe Water Quality Update with Emphasis on Phosphorus Trends*

- ☐ Imp B19 **LSEMS. 2001** *Lake Simcoe Water Quality Update: LSEMS Phase II Progress Report, 1995-1999*

- ☐ Imp B20 **Eimers, M.C., Winter, J.G. 2005** *Lake Simcoe Water Quality Update 2000 - 2003*

Technical Bulletins

- ☐ What Are Macrophytes? How Are They Affecting Lake Simcoe? September 2003