Reservoir Water Quality Treatment Study

Includes Water Quality Assessment and Modeling for the reservoirs Fanshawe Lake, Wildwood, and Pittock

Prepared by

Gertrud Nürnberg *and* Bruce LaZerte

Freshwater Research

3421 Hwy 117, Baysville, Ontario, P0B 1A0

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1. Executive Summary

In this report, limnological characteristics and phosphorus mass balances of Fanshawe Lake, Wildwood and Pittock Reservoir are based on historical and 2004 monitoring data. Most of the data were made available by the Upper Thames River Conservation Authority (UTRCA), but additional sources were queried, like the Ontario Ministry of the Environment (MOE), and several specialists from universities. The data were amended with additional monitoring efforts in summer 2004, by UTRCA and the University of Toronto.

Several characteristics determine the **limnological functioning of Fanshawe Lake**. (1) It is a run-of-the-river reservoir with an average water residence time of less than 10 days (range from 5 to 20 days between 1954 and 2004, a 51 year-period). (2) It has two bottom outlets for hydro-electrical power generation and a low flow valve, in addition to the high-flow surface outlet. (3) It is located in a major South Ontario agricultural area in the Thames River water system, so that it receives agriculturally polluted water from upstream reservoirs and is the source for several downstream lakes.

The **overall water quality of Fanshawe Lake** can be summarized as eutrophic to hyper-eutrophic. Total phosphorus (TP) concentrations and indicators of algal biomass (chlorophyll and Secchi transparency) decrease significantly from upstream to downstream locations within the lake. This trend has been found in the study year 2004 for all three water quality variables and was verified for Secchi with data from earlier studies within the 1990 to 2003 period. Although it is well-mixed in the shallow upstream section, it occasionally thermally stratifies close to the dam in the summer. During these periods, oxygen depletion (hypoxia and anoxia) and elevated phosphorus concentrations occur in the deep water layer, indicating phosphorus release from the sediments as internal phosphorus load. There are various indications that Fanshawe Lake is phosphorus limited, while nitrogen is in surplus.

There is evidence of the invasion by the zebra mussel. Although this event is problematic with respect to the current bottom fauna, it means higher transparency and water clarity, as observed in the last years.

Fanshawe Lake's water quality varies from summer to summer. To quantify occurrences of algal blooms and increase the years of observations with algal biomass indicators (there is no "bloom journal" and there are only five years of reliable chlorophyll data) a novel indicator of algal blooms was developed. This variable, Low-Nitrate-Days (LND) is based on the observation that in Fanshawe Lake high chlorophyll (indicator of algal blooms) coincides with periods of low nitrate during the summer.

The **bloom indicator LND** (days/year) is defined as the period of time during summer and early fall, when nitrate concentration of the outflow is below about 1 mg/L. There are 37 years between 1966 and 2004 with such information in Fanshawe Lake, and LND ranges from 0 to 175 with a long-term average of 62 days/year. There is a small significant decrease with time, suggesting that water quality was worse in the past and is slowly improving.

To determine the important causes of the observed variability of water quality, physical characteristics, including annual and seasonal flows, and their relationships with water quality

indicators and LND were investigated. Simply put, the results show that during high-flow years the water quality is relatively good, while during low-flow years it is poor. Also, higher phosphorus concentrations and poor water quality in the reservoir are correlated with the lower phosphorus loadings of low-flow years.

A **mass balance study** involving 23 years of data between 1975 and 2004 attempts to sort out the importance of phosphorus loading from various sources, in particular external versus internal load. More than 95% of the external phosphorus load reaches Fanshawe Lake via the main inflow, the North Thames River (median 15.9 $g/m^2/yr$), while other inputs like precipitation, the tributary, Wye Creek, and immediate runoff are insignificant. Internal load, as the phosphorus released from anoxic sediments, was calculated to be at least 22% on average (computed from comparison of annual input with export), but could be as high as a third of median external load. Because of the bottom outlet, much of this internal load leaves Fanshawe Lake without much contact and fertilizing influence on its surface waters. However, it can be expected to have a fertilizing effect on its downstream waters and increase eutrophication of these parts of the Thames River.

Because **Pittock and Wildwood Reservoirs** have been known to have significant algal blooms during some summers, their limnology and trophic state were investigated and mass balances computed. But much less data were available for these reservoirs and their gauged inflows represent only a small part of total flows (59% for Pittock and 28% for Wildwood). Therefore, their evaluation is much less certain than that of Fanshawe Lake.

The available TP, chlorophyll and Secchi disk data suggest that Wildwood Reservoir is less eutrophic than Pittock, which is hyper-eutrophic with respect to Secchi disk transparency. There are nitrate concentrations of the outflow available to compute LND for 24 years in Wildwood, and 22 years of Pittock. The average duration of LND predicted blooms is similar, 79 days in Wildwood and 83 days in Pittock, approximately 20 days longer than in Fanshawe Lake (62 days).

Mass balance analysis, including the application of the known inflow phosphorus concentration to the un-gauged portion of the reservoirs, reveal that Wildwood Reservoir likely acts as a phosphorus sink by retaining some of its phosphorus loading from external sources (7% on average), while Pittock Reservoir adds a certain amount (20% of external load on average) by releasing it, indicating internal P loading. As Wildwood is upstream of Fanshawe Lake, its phosphorus retention is beneficial for Fanshawe Lake (and other downstream parts of the Thames River). Contrarily, there are no reservoirs located downstream of Pittock Reservoir that could be adversely affected by its phosphorus export. However, downstream parts of the South Thames River will be impacted.

The **recommendations for the restoration** of Fanshawe, Pittock and Wildwood Reservoirs are based on the restoration goals and decision criteria assembled by the Project Management Team (Appendix I). When choosing treatment options, the reservoirs' position in agricultural watersheds, their bottom outlets, the flow dependencies, and the internal phosphorus load have to be taken into account.

Most of the variability in LND of all investigated reservoirs is due to water flow. Therefore, operational changes to accommodate this relationship would be beneficial. The evaluation of treatment options, summarised in Table 6-1, reveals hypolimnetic withdrawal as the most feasible and promising option for Fanshawe Lake, as well as the other reservoirs. This means operational changes, so that the summer flows would be augmented and the discharge of phosphorus-rich water from the bottom outlet maximized. Such treatment is based on enhanced bottom water withdrawal. It has been found to successfully reduce phosphorus concentration in the surface and bottom water, decrease algal biomass and eventually decrease hypolimnetic oxygen depletion. This treatment would also benefit Wildwood, as it is stratified and has high phosphorus concentration in the bottom. It would benefit Pittock probably to a lesser degree, because it is mixed more frequently.

Unfortunately, increasing TP exports from these reservoirs may adversely affect downstream water quality. In particular, increasing Wildwood's exports could negatively impact Fanshawe Lake. The importance of the downstream effects depends on the acceptability of the water quality and the anticipated recreational usage of the affected parts of the river. To determine relative importance and for prioritization, the watersheds should be investigated as a whole, including the rivers and reservoirs, as they influence each other. Therefore, several future activities are proposed that should benefit the water quality of the whole area under the jurisdiction of the UTRCA.

The **proposed future activities** include:

- Monitoring to determine eutrophication and support the LND – algal bloom relationship
- Public outreach: Involving volunteers of the public for educational purposes and to assist the monitoring effort (recording Secchi disk transparency in reservoirs and deeper river sections, keep journal for algal blooms)
- Modeling the whole system with help of all available and future water quality and flow data
- The creation of an "Upper Thames River Master Plan" (UTRMP) with the mission to
    - Identify and remediate pollution hotspots
    - Identify and implement further restoration possibilities (on land and water)

2. Introduction

2.1. Purpose of this Report

Fanshawe Lake is a reservoir on the North Thames River constructed by the Upper Thames River Conservation Authority (UTRCA) in 1950 (filled to permanent pool height in 1952) to aid with flood control. Since then many other purposes have been achieved, including the operation of a hydroelectric plant for 400 households, flow augmentation and recreation. Recreation has become an important asset as the lake was used for sculling as early as 1952 and has been the National High Performance Rowing Centre for the Canadian Women's Olympic Rowing Team since the eighties. In addition, Fanshawe Lake's proximity to major population centres in Southwest Ontario warrants its frequent use for fishing, sailing, as well as day and overnight camping in adjacent park land owned by the UTRCA (Figure 2-1).

Unfortunately, "poor water quality has been a concern for the past two decades with relatively frequent episodes of blue-green algae and elevated bacterial concentrations", according to the UTRCA (RFP, June 2004), despite 25 years of upstream diffuse source pollution control.

Two other reservoirs owned by the UTRCA are important for recreational purposes as well, and show similar signs of eutrophication. These are Wildwood Reservoir, upstream of Fanshawe on Trout Creek just above the Town of St. Mary's, and Pittock Reservoir (Gordon Pittock) on the Thames River, just above the Town of Woodstock.

In this report, only those results are included that are reasonably certain and useful for choosing treatments. First, limnological characteristics of Fanshawe Lake are presented as evident from historical and 2004 monitoring data. Evaluation of the other two reservoirs is included but is necessarily more cursory because of a lack of data. Besides the limnological characterization, mass balances for the nutrient phosphorus were computed and water quality modelled. Based on these analyses and exchanges with the Project Management Team, recommendations, including treatment options and future activities, are presented that will most likely improve water quality in the reservoirs.

2.2. General Characteristics of Fanshawe Lake

Fanshawe Lake is located in south-west Ontario, at the approximate location of 43°2' 20" North 81°11'5" West, just north of the City of London. It is situated in a rural area and its watershed is mainly agricultural with some parks and recreational facilities immediately around the lake. It is a "run-of-the-river" reservoir of the North Thames River, which represents the main inflow and outlet with a dam and spillway. During the summer most water leaves the lake at its downstream end via outlets at about 8 -10 m depth for a hydro-electrical facility and a low flow valve. This valve allows the bottom water from the reservoir to be sprayed into the downstream stilling area, hence increasing oxygen content and decreasing reduced gases to the receiving stream.

Day use camp

Fishing near the dam

Hydroelectric outflow near Fanshawe Dam

Figure 2-1. Fanshawe Lake vistas of July 12, 2004, showing rowing, sailing, fishing, picnicking, hydro electric power generation, and flood control

Photos by G. Nürnberg

The watershed/lake area ratio is large, as is typical for run-of-the-river reservoirs. The morphometric characteristics (Table 2-1, especially the morphometric index) are typical of a polymictic reservoir, i.e. the water mixes from time to time; it is probably always mixed at the shallower upstream part (Figure 2-2), but may occasionally stratify in the summer at the deeper section close to the dam. On such occasions, dissolved oxygen (DO) concentration may decrease to hypoxic (defined here as below 5.5 mg/L) and even anoxic (defined here as below 2 mg/L, when measured by immersed probes).

All morphometric data are based on hypsographic information provided by UTRCA and are listed in Appendix A. Volumetric variables and flows are based on the years 1954 to 2004 and therefore are slightly different from those used in previous studies.

Table 2-1. Morphometry and hydrology of Fanshawe Lake

Altitude at average pool[1] (m above sea level)	262.4
Watershed area, A_d (km^2)	1,447.4
Surface area[1], A_o (ha)	272.6
Area-Ratio, A_d/A_o	532
Maximum depth (m)	12.1
Mean depth[1], z (m)	4.82
Morphometric index, $z/A_o^{0.5}$	2.93
Volume[1] (10^6 m³)	13.146
Outflow volume[1] (10^6 m³ per yr)	560
Water residence time[1], τ (volume/outflow)	0.026 years or 9.5 days
Annual flushing rate[1], $\rho = 1/\tau$ (per yr)	38.4
Annual water load[1], $q_s = z/\tau$ (m/yr)	205

[1]Longterm average 1954-2004

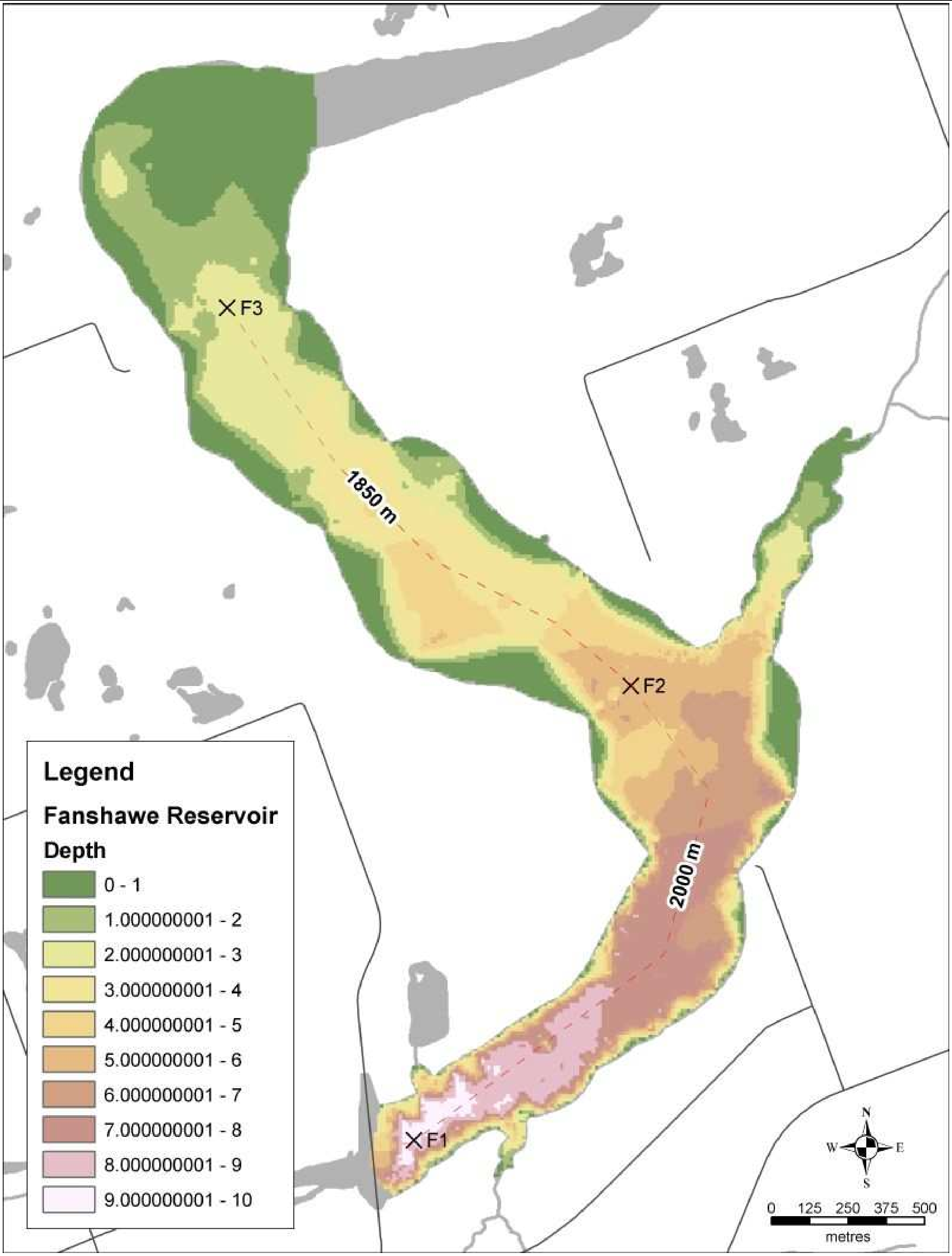


Figure 2-2. Fanshawe Lake bathymetry (provided by UTRCA) and sampling sites

2.3. Sources of Water Quality Data and 2004 Data Collection

Reservoir water quality data including nutrients, algal biomass (chlorophyll concentration) and Secchi disc transparency (for turbidity and algal biomass) and temperature and dissolved oxygen profiles are available for several years between 1988 and 2004 and some early data for 1973. In addition, there are nutrient data for the in- and outflows. All data were inspected and are made available separately from this report on file. Reservoir data were summarized to represent euphotic zone (i.e. surface layer) average summer concentration.

To complete previous efforts, supplemental data collection was done in the summer of 2004. In particular, total phosphorus (TP), chlorophyll, Secchi, and temperature and oxygen profiles were sampled five times between July 19 and September 17, 2004 at three locations in Fanshawe Lake (Figure 2-2), to determine any temporal and spatial trends. In addition, TP samples were taken at three different depths at the dam, to determine whether concentration may increase with depth, which would indicate P release from sediments and internal P loading. Field sampling was conducted by UTRCA staff; TP and Chlorophyll were analyzed in a commercial lab. The data for this sampling effort in 2004 are listed in Appendix C.

To further assess the probability of sediment P release, sediment samples were taken at the three sampling locations and analysed for TP, metals and organic content by the laboratory of Prof. Cyr of the University of Toronto. See data report in Appendix H.

In Wildwood and Pittock reservoirs, euphotic zone samples were taken at two locations each on July 22 and August 19, 2004 and analysed for TP and chlorophyll.

Statistical analysis was used to decide whether a pattern was likely “real” or due to chance alone. Usually linear regression analysis was performed and three statistics are reported: (1) the sample size, n, (2) R^2 that represents the proportion of the variability explained, and (3) the significance level p. In testing correlations and regressions, generally a level of 95% or $p=0.05$ or better was applied. Only, when sample size was small, i.e. seven or less, a level of 90% or $p=0.10$ was deemed marginally significant. As a measure of central tendency the average was computed unless the sample was small and possibly biased; in that case a median was used.

3. Water Quality

It is important to evaluate Fanshawe Lake's past and present water quality, so that a status quo is determined against which potential future treatment effects can be evaluated. Also, if present and historical water quality can be explained, future conditions involving various restoration techniques can be predicted.

Algal growth in lakes and reservoirs is usually limited by the supply of phosphorus so that blooms increase with increasing phosphorus concentrations in the water (e.g. Nürnberg 1996). Changes in the mass of phosphorus entering a lake or reservoir from the watershed (external loading) or lake sediments (internal loading), will change the average concentration of phosphorus within the lake and consequently the abundance of algae. Furthermore, the actual summer algal biomass may be limited by light because of increased turbidity or increased mixing depth in certain reservoir sections, and algal cells may be flushed out faster than they can reproduce.

Summer averages of water quality variables are often used to characterize annual trends in lakes and reservoirs. Usually, summer is also the season that is most important to lake users. To facilitate comparison of water quality between lakes and reservoirs, a classification with respect to "trophic state" has been applied by many limnologists. Threshold values of the most important trophic state indicators are listed in Table 3-1 (Nürnberg 1996).

3.1. Trophic State and Water Quality of the Euphotic Zone

A comparison of the euphotic zone summer averages at the dam in Fanshawe Lake (Table 3-2) with the threshold values of the trophic state classification (Table 3-1), reveals that all variables indicate eutrophic conditions for the reported years, except for the year 1989. That year showed extreme values for TP, chlorophyll and Secchi, representative of hyper-eutrophic conditions. In contrast, the subsequent year 1990 and the most recent year 2004 showed better water quality that was borderline mesotrophic-eutrophic. The first year for which water quality data are available (1973) can be classified as eutrophic based on TP and Secchi values, despite an uncharacteristically low chlorophyll average. The early chlorophyll analysis in the seventies by the MOE involved subtraction of pheophyton that led to underestimation of the chlorophyll values (Ken Nicholls, MOE, pers. comm. 1996). Therefore the 1973 chlorophyll average is not included in any subsequent analysis presented here.

Total nitrogen (TN) data, available for three years (1989 to 1991) were very high at 4.5 to 6.7 mg/L and are off the classification scheme indicating hyper-eutrophic conditions. This is characteristic of agricultural drainage and as a result, Fanshawe Lake is clearly not nitrogen limited, and nitrogen management would not lead to increased water quality. Instead, Fanshawe Lake shows indications of phosphorus limitation as the TN:TP weight ratio is usually very high (140 on average, never below 14), far higher than 7 - 16 above which phosphorus limitation is expected. In addition, there is a good relationship between phosphorus and chlorophyll (Section 3.5) and the bioavailable fraction of phosphorus, SRP, is usually below the detection limit during the growing season.

There are no long-term trends apparent for TP and chlorophyll summer averages in the euphotic zone; however, Secchi transparency has improved in the last two years to values that were never recorded before. The increased transparency coincides with the colonization of Fanshawe Lake by the Zebra Mussel (*Dreissena polymorpha*), which has been increased in abundance since its first observation in 1999. The hypothesis that increased water clarity is due to the Zebra Mussel in Fanshawe Lake is supported by the lack of such apparent increase in the other two reservoirs that have not been invaded yet.

Table 3-1. Trophic state categories based on summer water quality

	Oligotrophic	Mesotrophic	Eutrophic	Hyper-eutrophic
Total Phosphorus (mg/L)	< 0.010	0.010 - 0.030	0.031 - 0.100	> 0.100
Total Nitrogen (mg/L)	< 0.350	0.350 – 0.650	0.650 – 1.200	>1.200
Chlorophyll (μg/L)	< 3.5	3.5 - 9	9.1 - 25	> 25
Secchi Disk transparency (m)	> 4	2 - 4	1 - 2.1	< 1

Table 3-2. Water quality characteristics in the euphotic zone at the dam (F1) and in mid reservoir (F2) for all available summers

Sites:	F1 at Dam			F2
Year	TP mg/L	Chl µg/L	Secchi m	Secchi m
1973	0.082	7.7 *	1.10	
1980	0.052		0.90	
1988	0.060	21.0	1.22	
1989	0.104	73.1	1.06	
1990	0.037	10.0	1.64	1.05
1991	0.060	19.4	0.92	
1999			1.01	1.03
2001			1.60	1.33
2003			2.10	1.00
2004	0.036	11.8	2.24	1.21

* Possibly an underestimate of early Chlorophyll analysis by MOE

Table 3-3. Average water quality in the euphotic zone at three sites in summer 2004

2004	F1	F2	F3	Average
Distance from Dam, m	125	2,125	3,975	
TP, mg/L	0.036	0.069	0.116	0.074
Chl, µg/L	11.8	37.5	45.0	31.4
Secchi, m	2.24	1.21	0.91	1.45

Past Secchi data also reveal a spatial trend whereby the dam site usually has higher transparency, indicating better water quality, than the mid-reservoir station (F2, Table 3-2, Figure 3-1). Additional sampling in 2004 at three locations along the reservoir supports this observation (Table 3-3), as all three trophic state characteristics improve from hyper-eutrophic conditions at the inflow site F3, to eutrophic at F2, and almost mesotrophic conditions at the dam.

In fact, the regression of Secchi disk transparency on location (expressed as distance upstream of dam Figure 2-2) is significant ($p<0.05$, $n=110$), indicating an average transparency decrease of 0.114 m per km upstream. However, it explains only a small proportion of the variability (4%, $R^2=0.04$), because of dependencies on water flow as presented in Section 3.5.2)

Such drastic differences in water quality of a run-of-the-river reservoir are well-known and have been observed by the authors in two reservoirs before (Lake Mitchell, South Dakota and Brownlee Reservoir, Idaho). These spatial differences have to be considered when setting water quality and treatment goals.

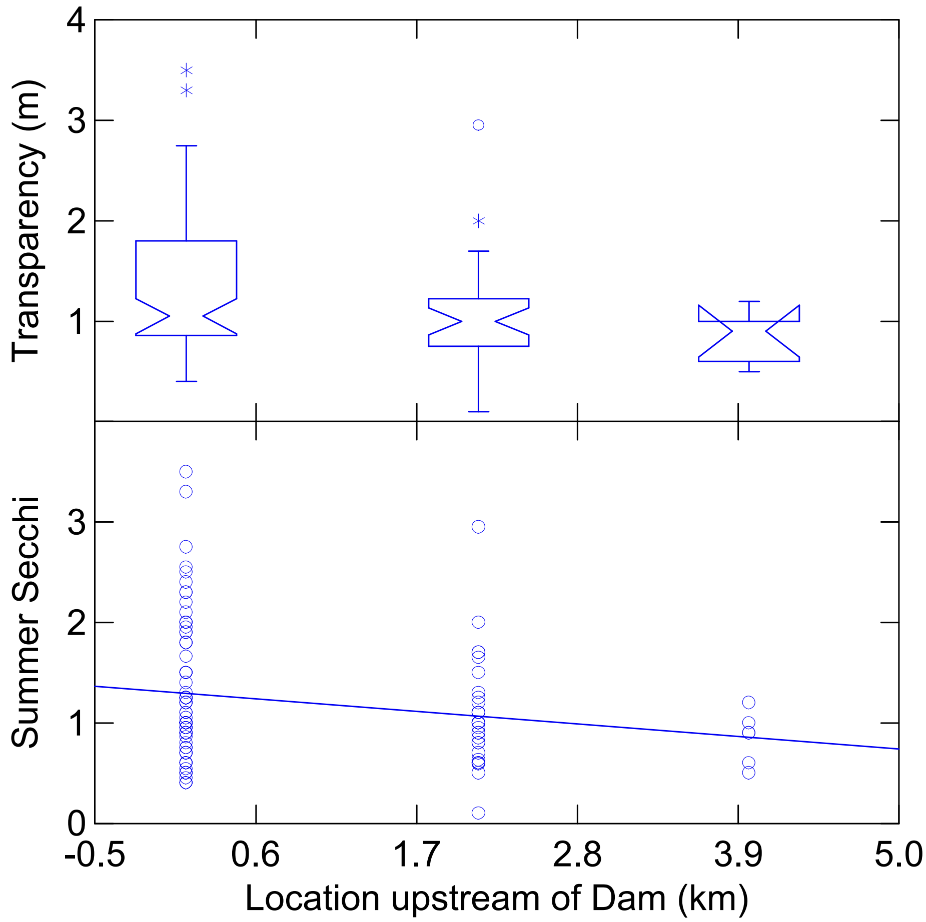


Figure 3-1. Summer Secchi disk transparency values along sampling sites

Note: Location "F3" at 3.9 km close to inflow was only sampled in 2004.

Lower panel: Individual data points for all stations and regression line. Upper panel: Medians and non-parametric confidence bands. The horizontal lines are upper hinges or 75th percentile, and lower hinges or 25th percentile, respectively. The narrow "waist" represents the median, the vertical line the range, except that star and circle represent outliers. The slanted lines off the median represent 95% non-parametric confidence bands.

3.2. Water Quality of the Deeper Water

Since Fanshawe Lake is relatively shallow with a high flushing rate (38 times per year, Table 2-1) and has a bottom outlet, temperature stratification is weak even at the deeper dam site. The temperature in the summer is often high, above 20 ºC throughout the water column, and only intensive monitoring reveals whether there is permanent stratification or not. Such data are available for one year, 1999, when thermo-sensors were left suspended at four different depths from May to November close to the dam in Fanshawe Lake to record temperatures every 4 hrs throughout he day. According to these data, the reservoir was weakly stratified from around June 10 until the beginning of September. The temperature during that period was 22 ºC at the bottom and up to 26 ºC at the top. Pockets of 24 ºC appeared in August at a deeper depth indicating occasional mixing events. Stratification was evident even before surface outflow was employed (July 9 to October 26) for maintenance operation of the pen stock.

Accordingly, despite polymixis and fast flushing, periods of thermal stratification and anoxia and hypoxia can exist in Fanshawe Lake and were quantified as anoxic and hypoxic factors as described in Appendix B. As expected, the actual values are relatively low, except in 1989 (Table 3-4) and indicate that the reservoir's weak stratification and short residence time (9.6 days average, Table 2-1) keeps oxygen depletion low despite high nutrient and probably high organic loads.

Table 3-4. Measures of anoxia (anoxic factor) and hypoxia (hypoxic factor)

	Factors (days/summer)	
	Anoxic	Hypoxic
1989	49	73
1990	17	42
1999	10	28
2000	not many data, but large	
2001	10	93
2002	5	25
2003	10	16
2004	2	18

However, when anoxia does occur, the shallow depth and high bottom temperatures provide good conditions for internal phosphorus loading as phosphorus release from anoxic sediments. This is evident by comparing phosphorus concentrations from the well-mixed euphotic layer with those sampled at a depth of about 8 m or 1 m above the bottom near the dam (F1). While euphotic TP was about 0.036 mg/L in 2004, the hypolimnetic concentration was 0.063 mg/L on average, with the last sample of Sept 17 as high as 0.079 mg/L (Appendix C). Similarly, the average bottom TP concentration of 42 available dates between 1988 and 1991 was elevated at 0.082 mg/L, with the 1989 average as high as 0.138 mg/L, as compared to far lower values in the euphotic layer

3.3. Algal Bloom Indicators

There are no quantitative records of the extent and duration of algal blooms, although such blooms are one of the main concerns that initiated this study. Chlorophyll concentration and Secchi disk transparency are indicators of algal biomass and extreme values indicate algal blooms.

When chlorophyll summer average concentration is above 10 µg/L, nuisance algal blooms of 30 µg/L chlorophyll or more can be occasionally expected according to Walmsley (1984). In Fanshawe Lake a similar relationship exists when medians are compared to individual values at the dam site (Figure 3-2). In an attempt to further quantify bloom frequency, Walker (1984) developed a model that predicts the frequency (% of summer) of nuisance algal blooms (at chlorophyll concentration above 30 µg/L) from summer average chlorophyll concentration (Chl). Multiplication by 1.55 turns the frequency to days per summer (assuming a 155 day summer period).

Bloom Frequency (d/summer) = 1.83 x (Chl - 10) x 1.55 **Equation 1**

In Fanshawe Lake, Equation 1 predicts that frequencies of nuisance algal blooms may have occurred between 0 and 100% (155 days) of the summer (Table 3-5). Unfortunately, there are only five years of reliable chlorophyll data for these predictions.

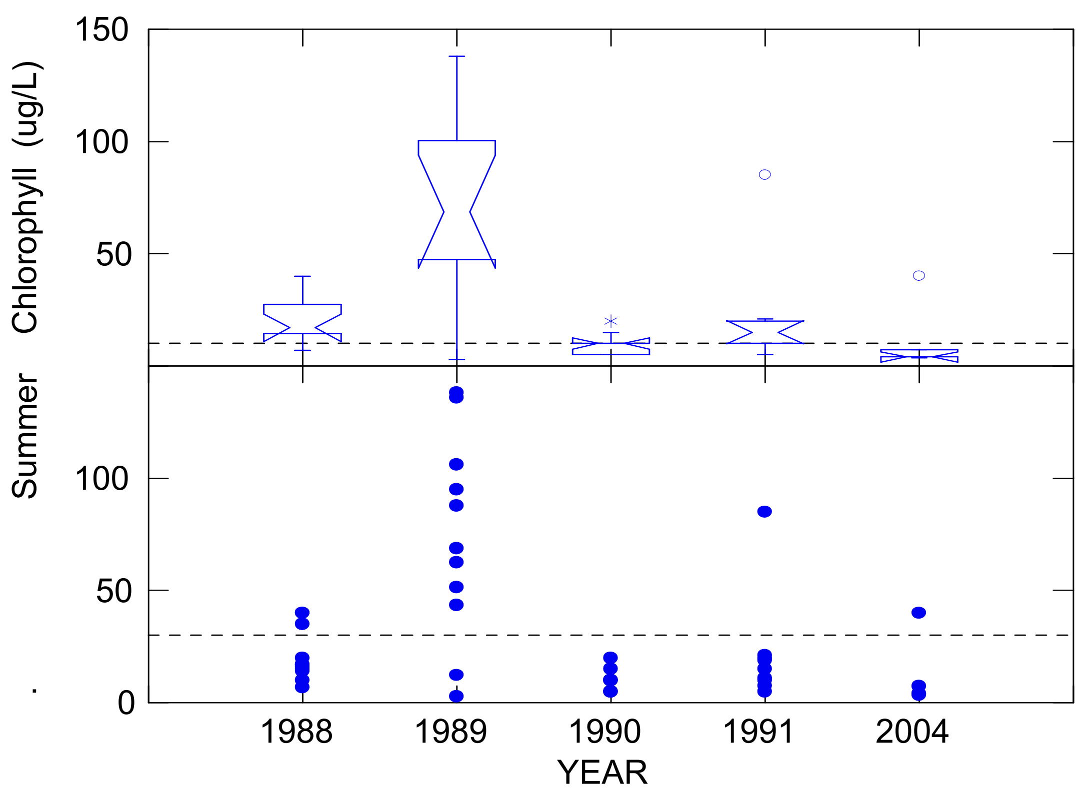


Figure 3-2. Chlorophyll concentration at the dam site.

Note that there are no data between 1991 and 2004. Broken line indicates 10 µg/L (upper panel, where "waist" indicates median) and 30 µg/L (lower panel of individual values).

Table 3-5. Algal bloom indicators at the dam

Year	Chl*	Bloom*	LND**
	μg/L	days per summer	
1988	21.0	30	73
1989	73.1	150	103
1990	10.0	0	0
1991	19.4	26	34
2004	11.8	5	28

*Based on average summer Chl concentration
**Low-Nitrate-Days, independent of Chl

As there is a lack of any long-term information on algal blooms in Fanshawe Lake, and there are only occasional observations and not more than five years of chlorophyll data, the available chemical data for two extreme years with respect to chlorophyll concentrations (1989 and 1990) were examined with the goal to find a pattern. Such a pattern emerged, as nitrate levels dropped to very low levels in the summer and fall of the hyper-eutrophic year 1989, while nitrate remained high in the high quality succeeding year in 1990 that did not have any bloom conditions. This observation was applied to other years as well. Nitrate is usually present in large quantities in Fanshawe Lake, but decreased in the late fall of several years, and as early as in July in 1989. The decrease of nitrate indicates an increase in algae, as they use nitrate for their growth. When nitrate approaches low levels, phytoplankton species compositions will shift to nitrogen-fixing bluegreen algae, as has been observed in other systems before. Therefore low nitrate not only indicates an increase in algal abundance in general, but possibly pinpoints the occurrence of bloom-forming bluegreen algae. To use this information in further analyses, a variable was created, the period of Low-Nitrate-Days (LND), to represent the potential period of bluegreen algal blooms. Values of LND were calculated as the period of days, where the nitrate concentration is at or below a threshold of about 1 mg/L NO_3-nitrogen. Since there are only four years of nitrate data available for Fanshawe Lake itself, the nitrate concentration in the outflow was used to determine LND. Data for outflow- and lake-based LNDs are listed in Appendix D.

Note: As there is no increased ammonia at the times of increased LND, and often total nitrogen may be decreased, it is not plausible to assume that low nitrate is simply a consequence of chemical reduction under hypoxic conditions. Instead, low nitrate appears to be a consequence of algal blooms.

Outflow-based LNDs are available for 37 years and there is a long-term trend apparent in Figure 3-3. The period of low nitrate usually starts in late summer and may reach far into fall but is highly variable between years. That means that LND covers a wide range, from no days to the entire summer and early fall (0 to 175 days). LND is high initially but decreasing from the sixties to the eighties, with occasional high-bloom years thereafter. In fact, the regression of LND on years is significant, although it only explains 19% of the variance ($n=37$, $R^2= 0.19$, $p<0.01$). As shown below, the other large part of the variance is due to hydrology (Section 3.5.2). For five years, when data are available, it follows the trend of Walker's "Bloom frequency" of Equation 1 as predicted from chlorophyll averages (Table 3-5) and LND is significantly positively correlated with Walker's frequency ($n=5$, $R^2=0.75$, $P<0.10$, Figure 3-4). Relationships with other water quality variables are presented in the following section (Section. 3.4).

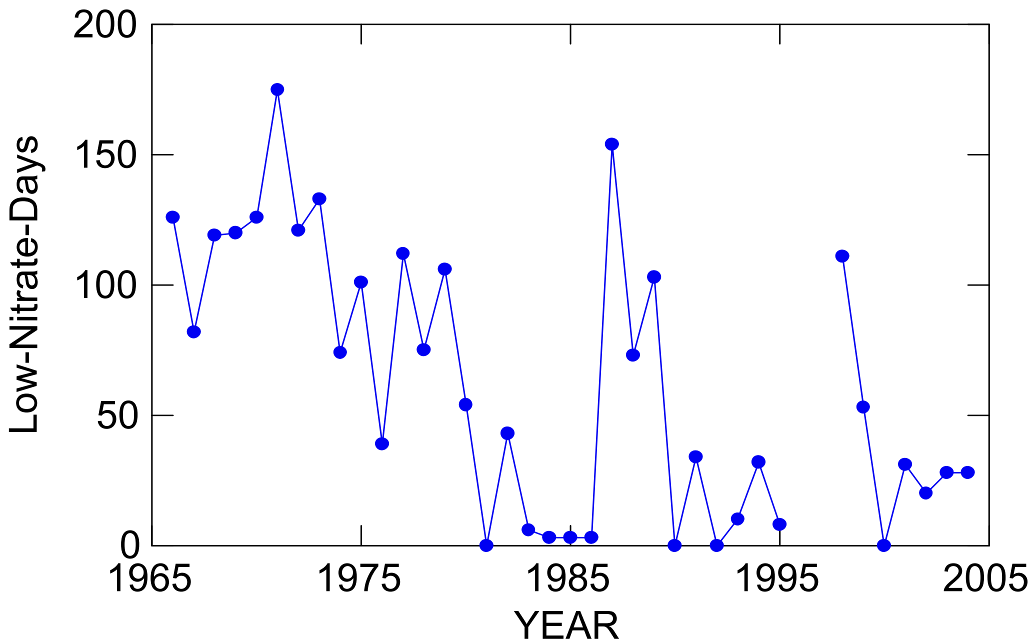


Figure 3-3. Algal bloom indicator. Low-Nitrate-Days (days) in Fanshawe Lake outflow.

No data for 1996 and 1997

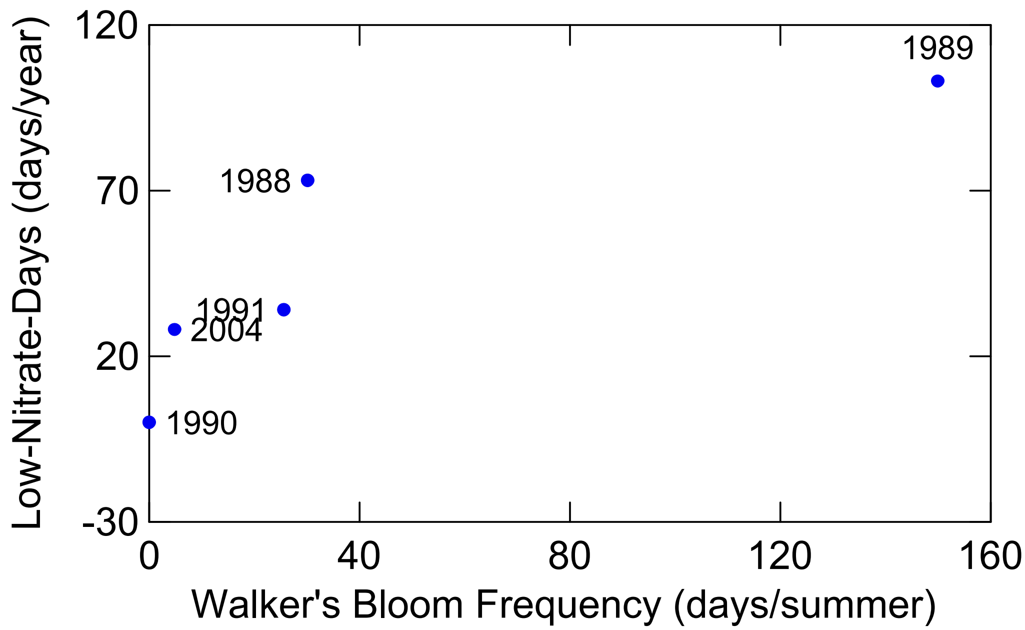


Figure 3-4. Low-Nitrate-Days compared to Walker's Bloom Frequency

3.4. Relationships among Water Quality Variables

If relationships between nutrients and algal biomass indicators are significant and positive, they point to the important variables that control algal blooms. Therefore, a regression analysis with respect to the water quality variables was conducted. Both, summer average and individual values at the dam site were used to corroborate results (Figure 3-5). Chlorophyll, as well as Secchi, is significantly correlated with TP, but not with TN for the individual data and the summer averages (Table 3-6). This result again indicates that nitrogen is not controlling algal biomass. Individual Secchi data are also correlated with chlorophyll, but annual averages are not (Table 3-6). The 2004 summer average is comparably more transparent, indicating the possible effect of the zebra mussel, as mentioned before. (However the regression does not improve by excluding 2004 from the analysis.)

Outflow-based LND is significantly correlated with summer average TP ($n=7$, $p<0.10$, Figure 3-6), but not with chlorophyll and Secchi (Table 3-6). The significant regression of LND on average TP is promising, as TP is the most important and reliable variable that indicates water quality in Fanshawe Lake. In comparison, representative chlorophyll estimates are much more difficult to achieve because of analytical difficulties and the large spatial variation of the pigment in the water. It is also possible that summer averages of chlorophyll and Secchi underestimate annual blooms in Fanshawe Lake because blooms possibly occur in the fall after the monitoring season. Furthermore, it could be that LND's represent bloom conditions for the whole reservoir, and not only for the less eutrophic site at the dam. At least, LND follows the same trend of chlorophyll, although in a curvilinear pattern, therefore the linear regression may not be the best model for its relationship with chlorophyll and the level of significance may be underestimated.

Overall, the significance levels of regressions between the water quality variables are quite high considering that the routine monitoring effort resulted in data for five to seven matching years only. The results affirm the quality of the available data and indicate that the observed trends and relationships accurately describe Fanshawe Lake's trophic state variables and their interactions. In particular, they corroborate the hypothesis that the level of algal blooms (as summer chlorophyll concentration) is controlled by TP concentrations. Furthermore, a comparison of the observed summer average Chlorophyll and Secchi with predictions based on various worldwide regression equations (Nürnberg 1996) shows reasonable agreement, indicating that Fanshawe Lake's water quality relationships is comparable to those of many other lakes and reservoirs (Appendix E). Future monitoring with respect to chlorophyll, nitrate concentration, and algal blooms may confirm that LND indeed indicates algal blooms. Despite some uncertainty with respect to the meaning of LND, we will use it here as a representation of algal bloom. This is the only way to investigate any possible long-term trends and relationships with other characteristics.

Table 3-6. Regression results for summer water quality relationships

Note: All values were log-transformed before analysis. Small sample size (5 to 10) represents May through September averages, while large sample size includes individual sampling dates; LND, Low-Nitrate-Days, are for the whole period including fall. n.s., not significant

Dependent	Independent	Sample Size n	R^2	Significance
Chlorophyll	TP	54	0.40	$p < 0.001$
Chlorophyll	TP	5	0.95	$p < 0.001$
LND	TP	7	0.51	$p < 0.10$
Secchi	TP	54	0.45	$p < 0.001$
Secchi	TP	7	0.45	$p < 0.10$
Secchi	LND	10	0.21	n.s.
LND	Chlorophyll	5	0.49	n.s.
Secchi	Chlorophyll	61	0.37	$p < 0.001$
Secchi	Chlorophyll	5	0.40	n.s.
Chlorophyll, LND, Secchi	Total Nitrogen			n.s.

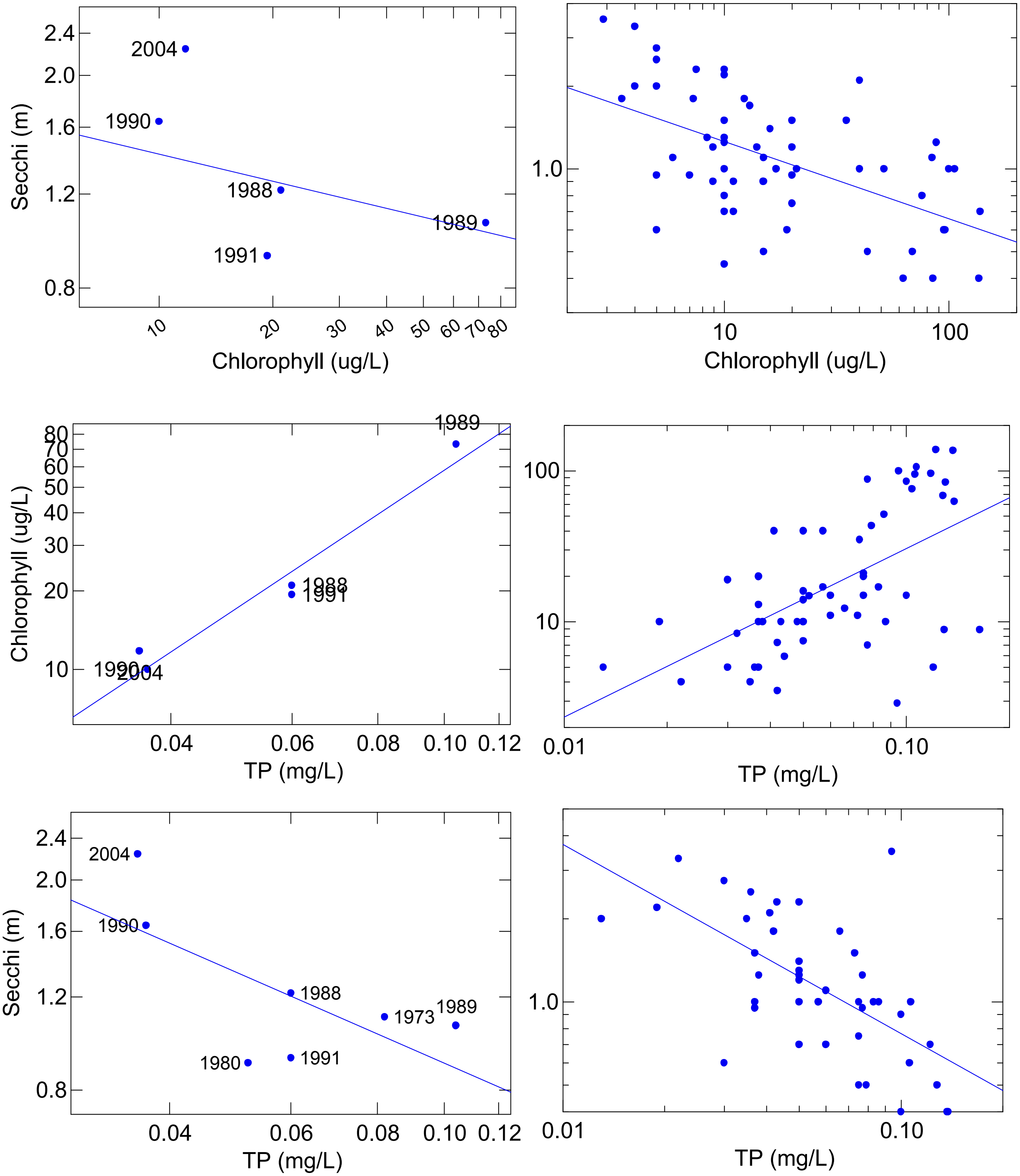


Figure 3-5. Relationships between water quality variables in Fanshawe Lake, for May through September in the euphotic zone at the dam site F1.

Note: Annual averages at left for indicated years, individual data at right; Regression lines are shown

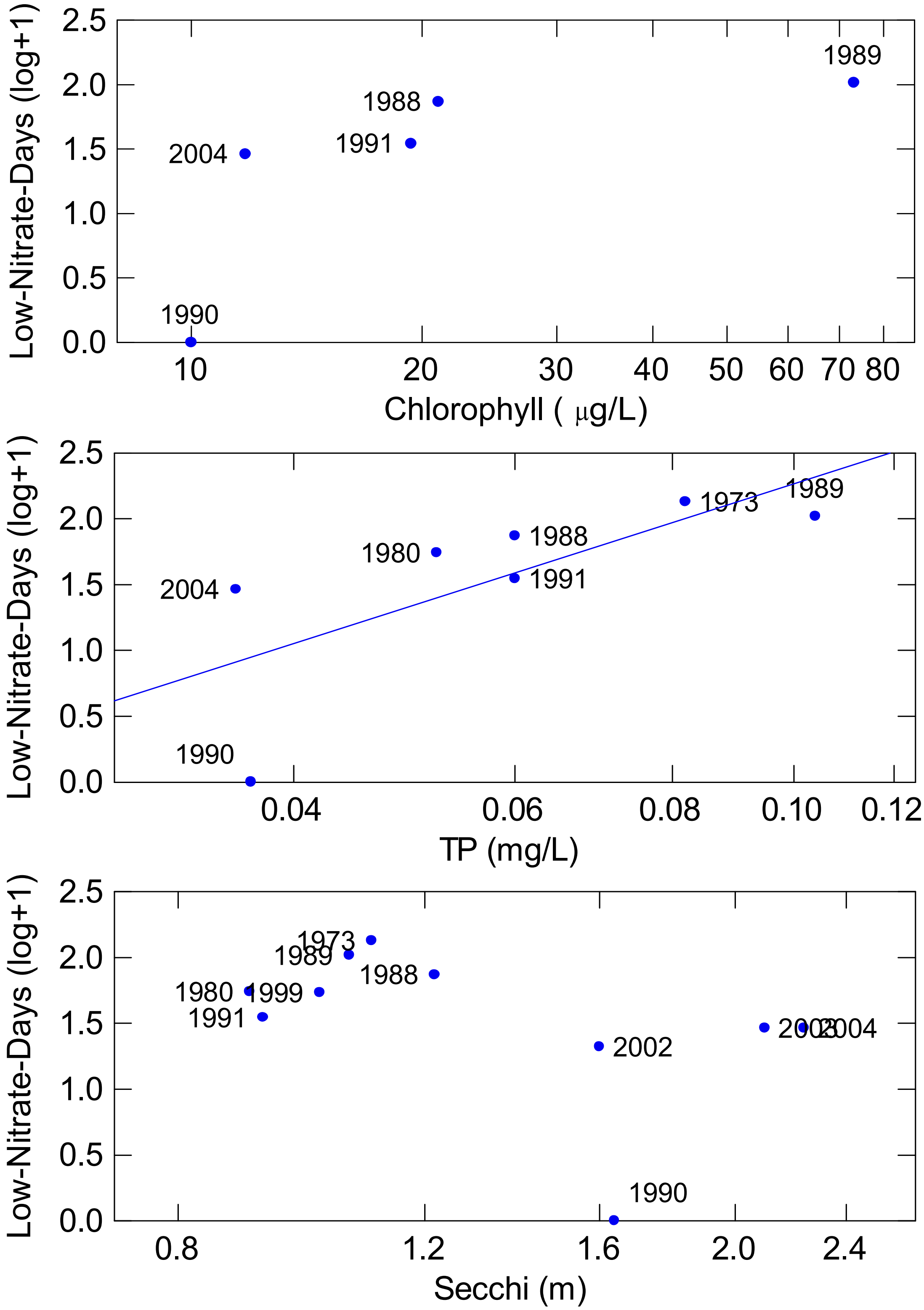


Figure 3-6. Comparison of Low-Nitrate-Days with summer average chlorophyll (top), TP (centre) and Secchi disk transparency (bottom)

3.5. Relationships between Water Quality Variables and Physical Characteristics

3.5.1. Physical characteristics

To determine the influence of physical lake characteristics, like flows, elevation and morphometry on water quality, the long-term trend of such variables was examined in detail. Only the most important results are presented here; some more are offered in Appendix F. The data used in this analysis are summarized in separate data files.

Inflows of the North Thames River at Plover Mills vary substantially from year to year and are highly significantly correlated with the Fanshawe Lake outflow especially for the May to September (summer) season (Figure 3-7) (n=51, $p<0.0001$, $R^2= 0.97$, log-transformed) and annual flows (n=51, $p<0.0001$, $R^2= 0.82$, log-transformed). Furthermore, the summer flows are virtually identical, as the regression line is not significantly different from the one-to-one line (regression equation: log outflow = -0.006 +1.002 log inflow). Therefore, any water quality variable dependent on either the in- or the outflow is correlated with both.

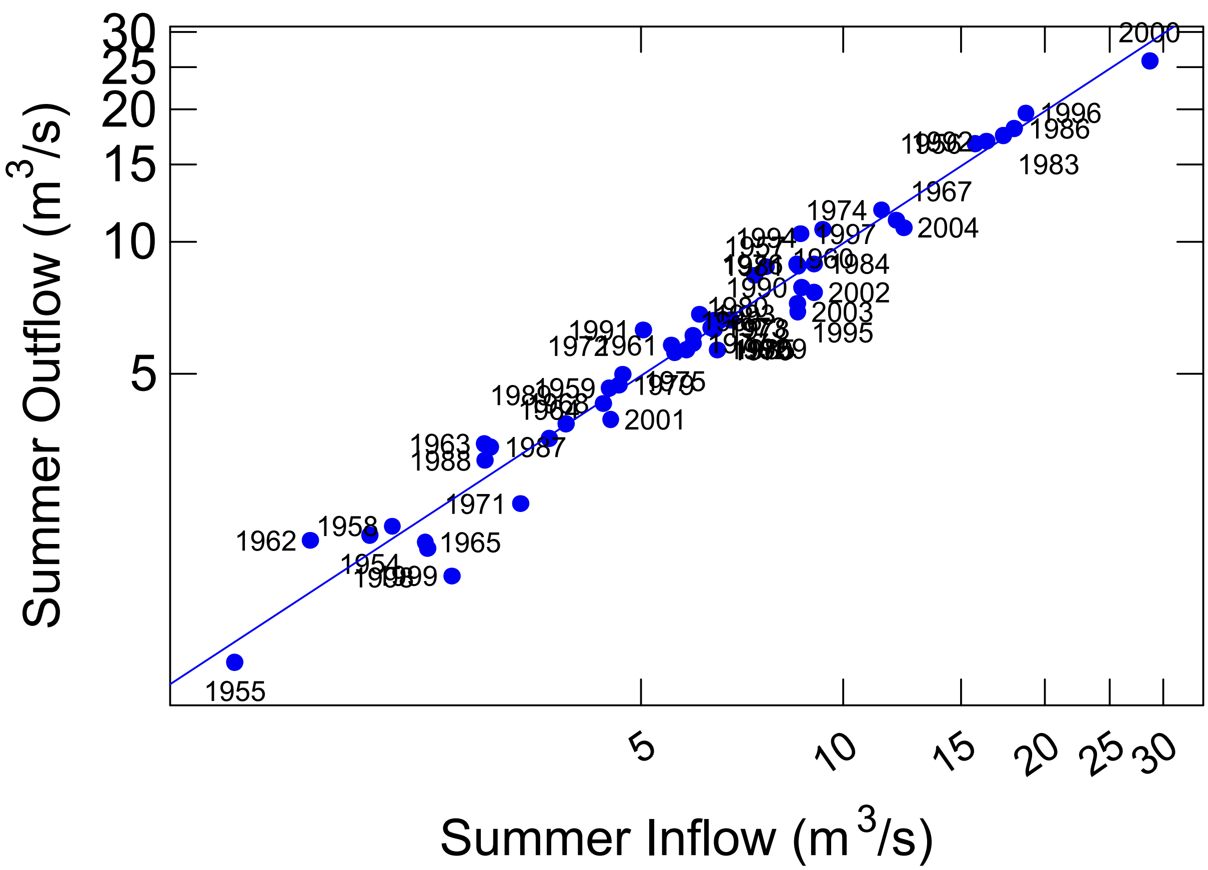


Figure 3-7. Summer (May to September) outflow versus inflow

After a highly variable beginning during the first years of operation, flows changed from relatively low flows in the mid sixties to higher ones after the seventies, which then were kept at a more or less constant level (Figure 3-8). This pattern is corroborated for both, summer and annual inflow averages by regression analysis, which indicates a significant increase with all years of operation, but no significant change after 1975.

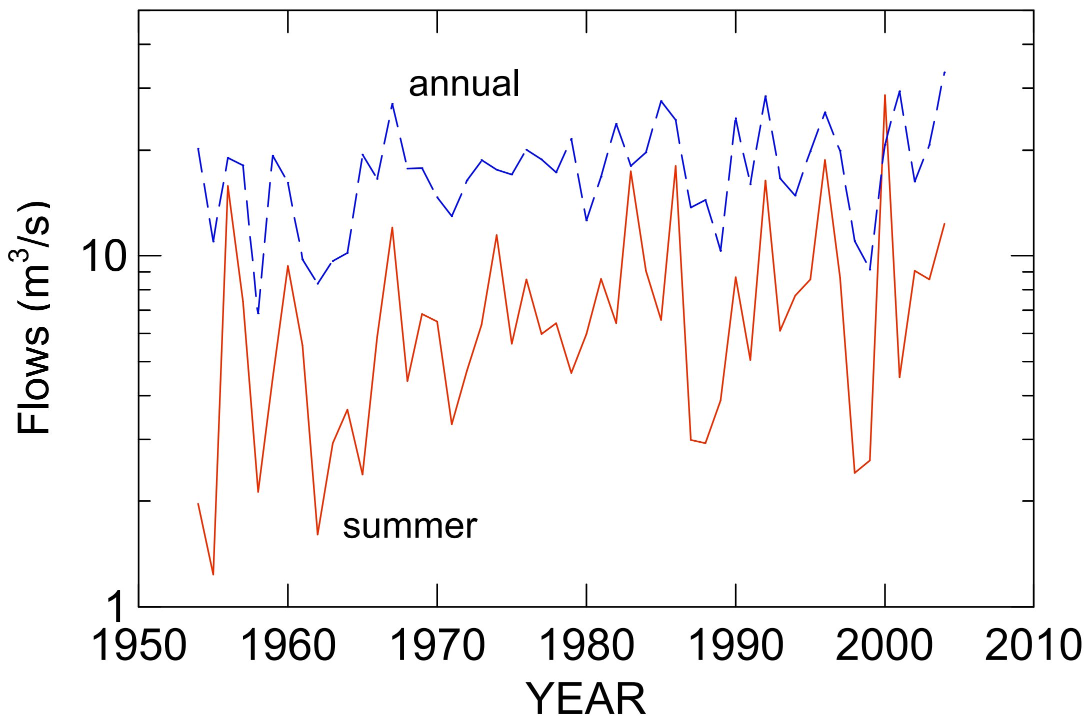


Figure 3-8. Annual and summer average flows of the main inflow

Similar to the flows, the TP loads also vary substantially from year to year (Figure 3-9). There is a slightly decreasing trend with time apparent, except for the most recent incoming loads. Since the flows do not show any trend after 1975, a concentration decrease should be the reason for any trend in load (Figure 3-10). However, load and concentration annual averages are based on few data, e.g. on four data points in 2004, and a more thorough sampling scheme would be necessary to determine if any trends are significant.

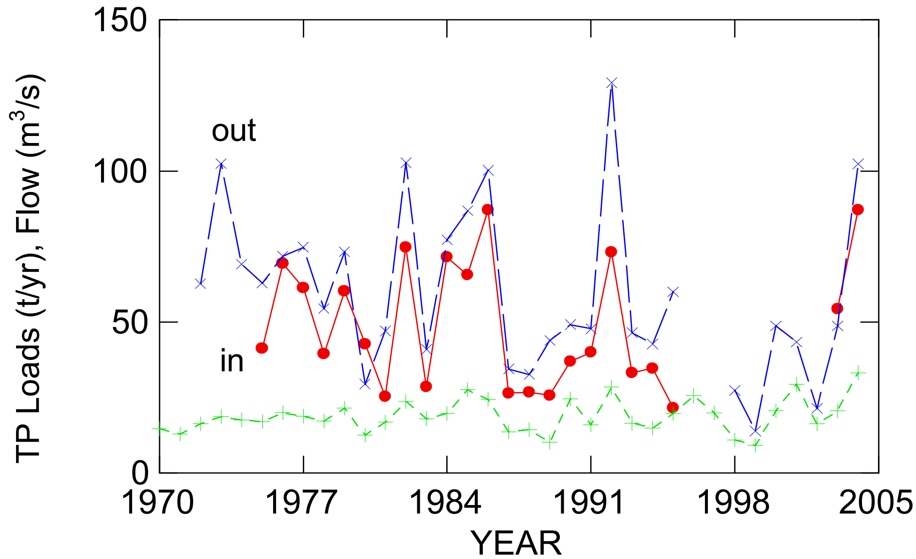


Figure 3-9. Comparison of annual TP load from the main inflow (filled circles, solid line) with the export (x, broken line). Annual inflow rate is shown at the bottom.

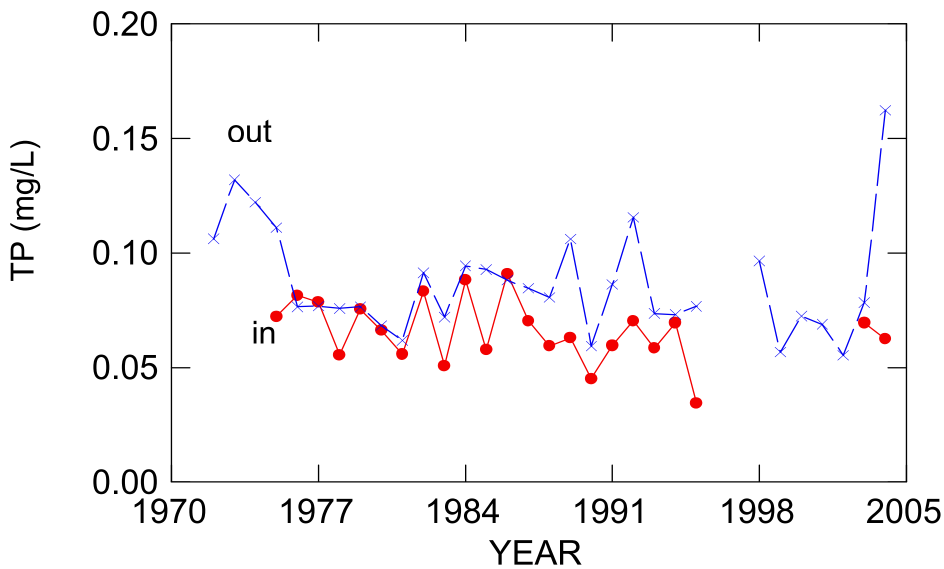


Figure 3-10. Comparison of annual TP concentration average of the main inflow (solid line) with that of the outflow (broken line).

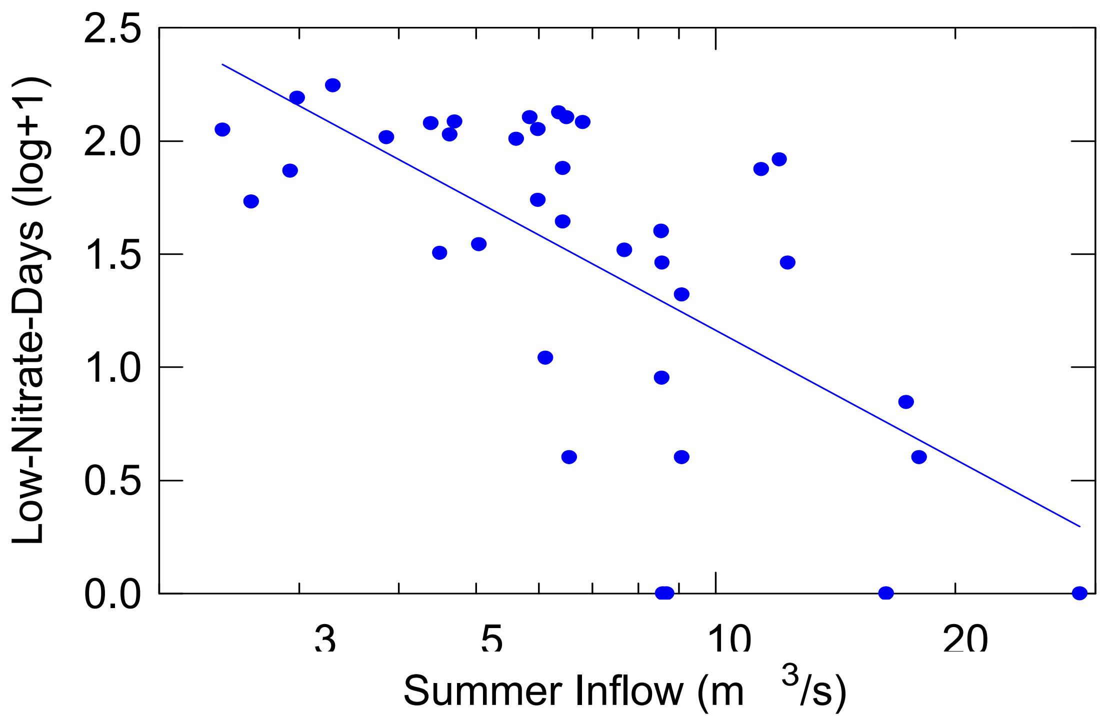
3.5.2. Relationships with flows and time

The observed large fluctuation in flows, nutrient loads and exports could influence water quality. To test this hypothesis, traditional water quality variables and Low-Nitrate-Days (LND, Section 3.3), were compared to various physical variables. (Tested were annual average flows as well as the 12 individual months, the four annual quarters, summer period as May through Sep, fall as Oct through Dec, and Jan through April.)

In fact, LND was significantly negatively correlated to in- and outflows for most of the time periods of the summer, including individual summer months. The highest regression coefficient was found for the average summer flows both for the inflow ($p<0.0001$, Figure 3-11, Table 3-7) and the outflow. In other words, the higher the summer flushing rate is, the better the water quality. The same tendency was apparent for the traditional variables of TP, Chlorophyll and Secchi, although the relationships sometimes were not significant, most likely because of low sample size (Table 3-7).

As described above in Section 3.3, there is also a significant decrease of LND with time. Therefore the influence of both variables, flows and years, was tested. In fact, the variable "year" improves the flow relationships significantly, so that for example, summer inflow's effect on LND is improved by 14% (from 43% to 57%, Table 3-7). The influence of "year" could mean decreasing phosphorus in inflow, that is marginally significant with years as well ($n=23$, $p<0.05$, $R^2= 0.19$).

Figure 3-11. Low-Nitrate-Days versus main summer inflow



But more summer flushing also means elevated nutrient loads, which are thus correlated with better water quality, expressed as LND (e.g. $n=23$, $R^2= 0.19$, $P<0.05$, for summer loads of the main inflow). In other lakes, high TP loads are associated with high lake-TP concentration and low water quality. But in a run-of-the-river reservoir like Fanshawe Lake, water quality increases with flushing (e.g. hypoxia decreased in a large Snake River reservoir, Nürnberg 2002) and nutrients, organic substances and algae may just be flushed out at higher flows (Soballe &

Kimmel 1987). In general, the average inflow TP concentration is more important than load in determining the phosphorus concentration in a run-of-the-river reservoir.

Table 3-7. Regression results for relationships between water quality and inflow

Note: All values were log-transformed before analysis; n.s., not significant

Dependent	Independent	n	R^2	Sign	Significance
TP	Summer inflow	7	0.48	-	$p < 0.10$
TP	Annual inflow	7	0.59	-	$p < 0.05$
Secchi	Summer inflow	10	0.53	+	$p < 0.05$
Secchi	Annual inflow	10	0.61	+	$p < 0.01$
LND	Annual inflow	37	0.20	-	$p < 0.01$
LND	Summer inflow	37	0.43	-	$p < 0.0001$
LND	Year	37	0.19	-	$p < 0.01$
LND	Summer inflow, Year	37	0.57	-, -	$p < 0.0001$
Inflow TP	Year	23	0.19	-	$p < 0.05$
Chlorophyll	Summer inflow	5	0.45	-	n.s.
Chlorophyll	Annual inflow	5	0.78	-	$p<0.05$

There are data available for four years that include one year with extremely low water quality, 1989, followed by a high water quality year, 1990 (Table 3-8). In 1990, high rain events led to high flow rates, high TP loads, but relatively good water quality. This can be explained by enhanced TP export via the bottom outlet. Also the volume-weighted annual and summer average inflow TP concentrations were comparably low in 1990 (Table 3-8).

Table 3-8. Four years in Fanshawe Lake as example of the complicated relationships.

Note: The water quality variables are based on May-Sep averages, all other variables are annual estimates. The incoming TP loading is a total load and includes the main inflow, the Wye Creek, precipitation and runoff from the immediate watershed.

	Water Quality Indicators				Climate and flow indicators				TP mass balance		Average inflow TP	
	TP	Chl	Secchi	LND	Precip	Flows (10^6 m^3)		tau	Loading (tonnes)		mg/L	
Year	mg/L	µg/L	m	d	mm	in	out	d	in	out	Annual	Summer
1988	0.060	21.00	1.22	73	922	477.8	427.6	11.3	28.48	32.71	0.059	0.051
1989	0.104	73.09	1.06	103	813	342.7	321.2	15.0	27.10	43.84	0.079	0.052
1990	0.037	10.00	1.64	0	1,300	818.2	729.6	7.1	40.28	49.18	0.048	0.041
1991	0.060	19.35	0.92	34	901	529.1	555.3	8.1	42.02	47.79	0.080	0.054

In general, flow rates and their year-to-year variation can have large impacts on water quality, independent of their impact on nutrient loads. They have to be taken into account when choosing treatment options.

4. Fanshawe Lake Phosphorus Mass Balance and Model

As described in the previous sections, algal biomass and blooms are controlled by the nutrient phosphorus and by water flows. In these interactions, higher phosphorus concentrations in the reservoir are correlated with more algae, but lower phosphorus loadings. As phosphorus is ultimately controlling the water quality in Fanshawe Lake, it is important to know the quantity and effect of the phosphorus input derived from various sources. In particular, it is important to assess the effect of external versus internal inputs in order to achieve restoration.

4.1. External Load and Export

External load to Fanshawe Lake was determined as the sum of the main inflow (the North Thames River at Plover Mills), the Wye Creek which flows into Fanshawe Lake about 500 m above the dam, inflow from the immediate watershed and precipitation directly unto the lake. The specific flows were multiplied by corresponding TP concentrations to arrive at loads (0).

External loads are highly variable from year to year because of variable flows. They range from 24 to 92 tonnes, with a median of 53 tonnes, or an areal load of 8.9 to 33 g/m^2/yr, with a median of 16 g/m^2/yr for a period of 23 years (Figure 4-1). Approximately 95% of external load arrives via the main inflow.

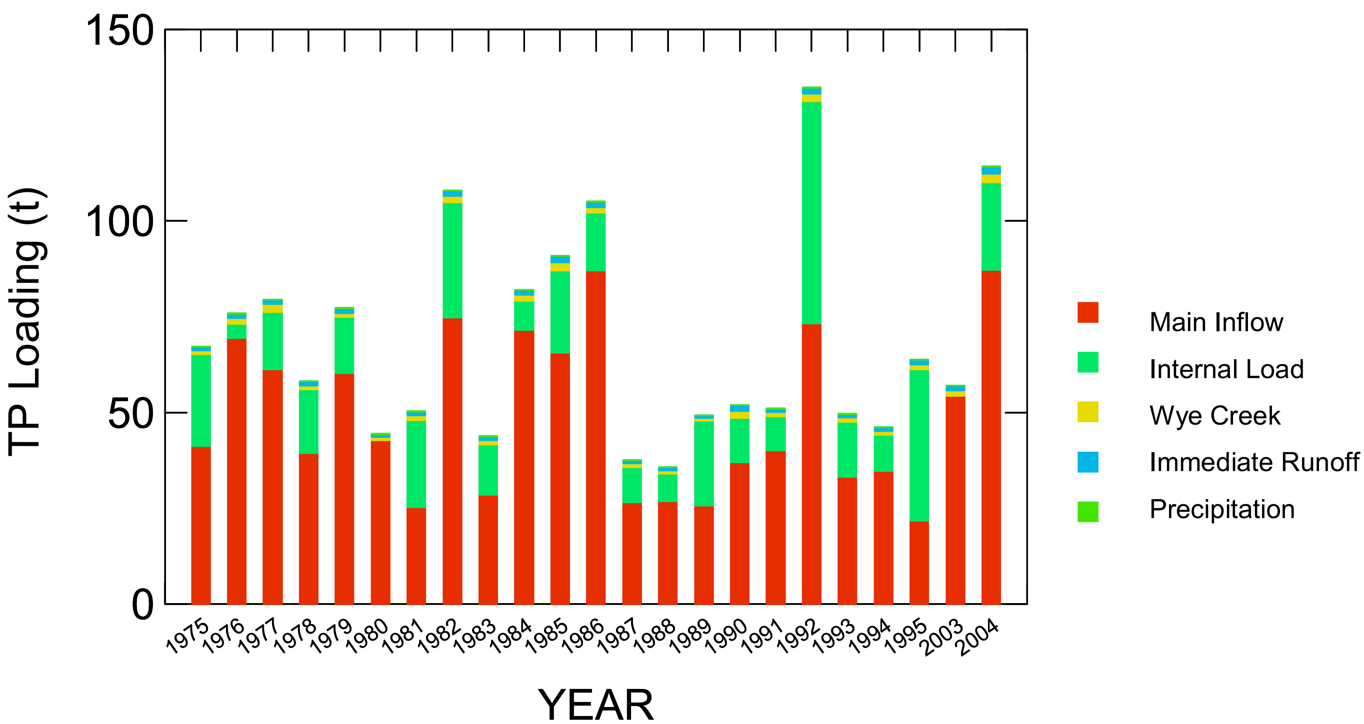


Figure 4-1. Comparison of TP loads from various sources.

The long-term average volume-weighted inflow concentration is 0.079 mg/L TP, and ranges from 0.035 to 0.115 mg/L. Approximately half of this is in a form that is considered to be biologically available as soluble reactive phosphorus (SRP) (on average 0.039 mg/L, ranging from 0.014 to 0.049). This is a large proportion of the external load and considerably influences algal growth.

Export was computed from outflow concentration and flow rates at the dam. They range from 14 to 129 tonnes with a median of 54 tonnes. The long-term median volume-weighted outflow concentration is 0.099 mg/L TP, and ranges from 0.050 to 0.196 mg/L for a period of 38 years. In- and outflows as well as load and export are quite similar, and usually increased inflow also means increased TP export in Fanshawe Lake.

4.2. Internal Load

In Fanshawe Lake TP export is even higher than the inflow load, since it happens mostly via the bottom outlet at 8-10 m depth. This hypolimnetic water has higher TP concentration than the euphotic zone, as discussed in Section 3.2, and export is enhanced (Figure 4-2, see Figure 3-9 and Figure 3-10 for annual averages). Because other sources (Wye Creek and precipitation) cannot account for the difference and also, because usually some of the incoming TP settles out during its travel through a reservoir, such enhanced export is indicative of internal phosphorus loading.

Internal load is the phosphorus load that is released from the sediments under anoxic conditions of the sediment surfaces. It originates from external inputs that settle and are transformed by geochemical processes in the sediments over time. The potential importance of internally derived phosphorus is higher than external load as it is in a form that is close to 90% biologically available (Nürnberg and Peters 1984), while the biologically available fraction of the external load was estimated as about 50%, based on SRP concentration in the inflow.

In Fanshawe Lake most of this internal load and net export of TP occurs during low flow years when total external loads are small (Figure 4-2). This is expected as reduced flushing rates increase anoxia and consequently internal load from the sediments (Section 3.2).

Net TP export is good for Fanshawe Lake as it indicates that the total mass of TP stored in the lake, including its sediments, has decreased. Also, much of the TP export comes from the hypolimnion; this internal load which otherwise would contribute to the lake surface TP concentration, enhancing algal blooms, is effectively withdrawn and sent downstream. The lake restoration technique of “hypolimnetic withdrawal” is based on this principal and enhances this effect (Nürnberg 1987). For restoration purposes, any improvement or extension of this process would be beneficial to Fanshawe Lake.

Quantitatively and over the long term, internal load is at least 3.7 g/m^2/yr (if mass balance data are reliable), which is the median difference between export and input as determined from simple mass balance computations for 32 years. It ranges from -5.5 to 18.8 g/m^2/yr over these years, where three years have negative values indicating a net retention of phosphorus in the lake.

In natural, slow flushing lakes without internal loading, a large proportion of TP, sometimes up to 90%, is typically retained every year due to settling processes. The exact percentage is dependent on flushing and depth, so that it can be predicted by the annual water load (i.e. the product of mean depth and annual average flushing rate; Nürnberg and LaZerte 2001). High retention means that the lake effectively traps phosphorus, which is good for downstream waters but also for the lake itself, as long as the phosphorus remains bound in the sediment. However, under more eutrophic conditions the sediment surface become oxygen depleted leading to phosphorus release from the bottom sediments as internal load. Once internal load takes place, observed retention, which is computed from mass balances as the difference of external load and export (so that R= (in-out)/in) is lower than predicted retention, as predicted retention does not include internal load.

Because of the high flushing rate in Fanshawe Lake, both, predicted (due to settling) retention and observed (including internal load) retention are small: a median of 7% for the predicted and of -22% for the observed. The observed net export in Fanshawe Lake can only be explained by an internal source of TP, an unrecognized source, or uncertainty in the data. As mass balances are based on sporadic samples of concentrations combined with daily flow rates, errors may be large for loads and exports. Nonetheless, net export was found for 20 out of 23 years which suggests that internal load is real.

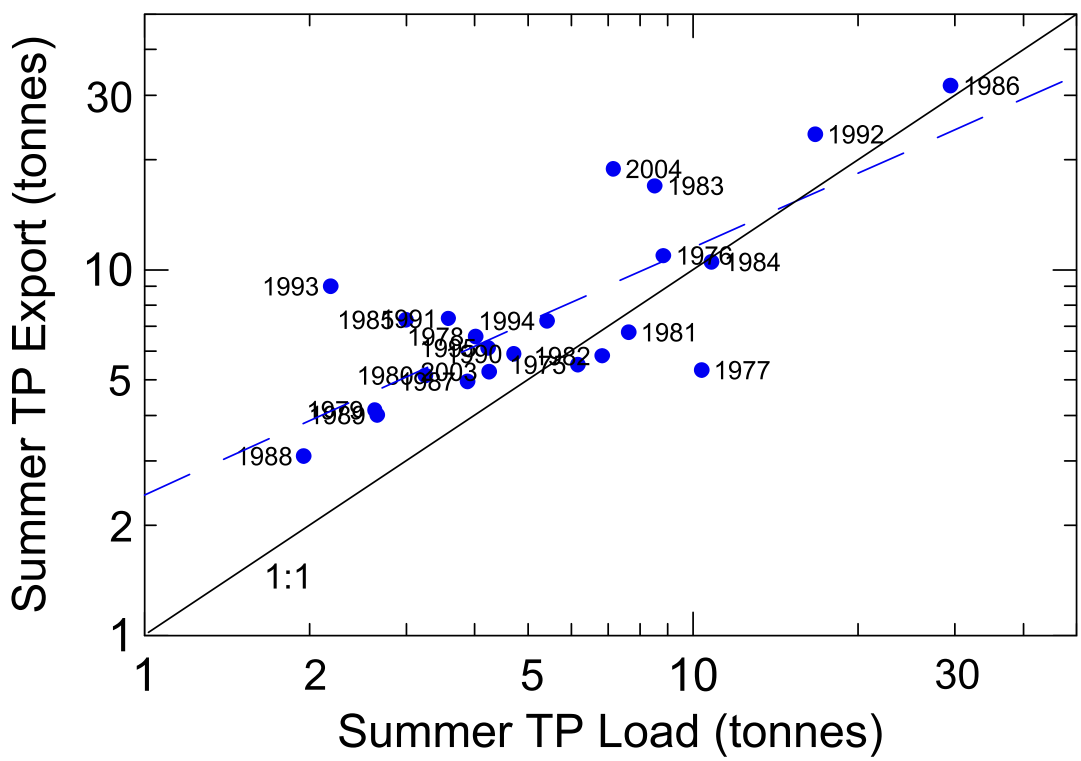


Figure 4-2. Summer TP export versus summer TP load of main inflow.

Note that in most summers export was much higher than load, indicating an internal phosphorus load. Regression line (broken) and 1:1 line are shown.

Internal load can actually be expected to be larger than the mere difference between export and input, since the settling (retention) of incoming load has to be taken into account. A retention model (based on annual waterload, q_s, Nürnberg and LaZerte 2001) can be applied to correct for retention. After this correction, the mass balance approach predicted a median internal load of 5.3 $g/m^2/yr$. Values based on this approach are included in Figure 4-1.

Another approach to estimating internal load is based on the TP concentration of the bottom sediment, as it is correlated to release rates (Nürnberg 1988). Consequently, sediment cores were taken in 2004 at the three routine sampling stations in Fanshawe Lake (Figure 2-2) and analysed for general characteristics and TP (Appendix H). Low water content and organic matter are indicative of a scoured bottom without soft and enriched layers. (Low sediment accumulation in Fanshawe Lake was also observed by Hayman et al. 1984, who studied sediment cores and sediment mass balances.) The phosphorus concentrations were similar at the different sites and less than 1.4 mg/g dry weight. According to regressions of Nürnberg (1988) based on cold anoxic hypolimnia, such concentration should produce a release rate of about 8 mg/m^2 of anoxic sediment surface per day. But because of the high bottom temperatures of above 20 ºC in Fanshawe Lake (Section 3.2), it is likely that the release rates are actually far higher than those predicted by the sediment phosphorus model. A temperature increase of ten degrees can increase TP release by 3 to 7 times (Jensen and Andersen 1992), so that the actual release rate may be between 24 - 56 $mg/m^2/d$ in the summer.

Release rates can be transformed into a summer internal load by measuring (in deep lakes) or predicting (in shallow lakes) the extent and duration of anoxia at the sediment surfaces. In a stratified lake this anoxia can be quantified from oxygen profiles as the anoxic factor. In a polymictic lake, such as Fanshawe Lake, the observed anoxic factor is usually too low, as the open water may not show any anoxia while sediment surfaces may still be anoxic and release phosphorus. Therefore a value of 60 d/summer was used, which is typical for eutrophic lakes. (This value is slightly higher than 49 d/summer of 1989, the highest recorded anoxic factor in Table 3-4). This value was multiplied by the release rate to obtain a measure of internal load (Nürnberg 2005). A release rate of about 24 - 56 $mg/m^2/day$ multiplied by an anoxic factor of 60 d/summer translates into an internal load of about 1.4 - 3.4 $g/m^2/summer$. Assuming that there is no internal load in the winter, this value represents an annual value, and is lower than the values derived from mass balance calculations (5.3 $g/m^2/year$).

In summary we conclude that on average internal loading (long-term median of 5.3 $g/m^2/year$) is as high as a third of the external load (long-term median of 16.1 $g/m^2/yr$).

4.3. Prediction of Lake TP

Mass balance models predict whole lake annual or summer average concentrations (Nürnberg 1998). And even though there are models available that take into account seasonality by including internal load in different ways, they are based on a mixed volume. A spatially variable, run-of-the river reservoir like Fanshawe Lake would ideally be treated like two or more separate basins, so that inflows are the export of the upper basin. Then models could be applied to each basin in sequence and tested independently. Such an approach requires morphometric and TP data for all basins however and such data are not available. Especially, the available in-lake TP concentrations, needed to test such models, are mostly taken at the dam site (F1), and data for two upstream stations are only available for the most recent year 2004 (Table 3-2, Table 3-3). As these data indicate a significant trend of decreasing TP along the reservoir, and also because of the observed TP increase with depth, whole reservoir average concentrations are certainly higher than those of the euphotic zone near the dam.

Nevertheless, a whole lake average concentration can be modelled even though it is difficult to verify the result. An over-the-years median of predicted annual average concentrations is 0.075 mg/L, using external load and flows w/o internal load, while a higher value of 0.095 mg/L, representing hyper-eutrophic conditions, is predicted when considering internal load. The lower value appears to adequately describe the data in 2004, when the average of all three stations (F1, F2, and F3) was 0.074 mg/L (although a volumetric average should be computed, when more detailed basin morphometry becomes available). As 2004 appears to be a relatively high water quality year with high flushing and comparably low TP at the dam, using the prediction based on no internal load (0.075 mg/L) is more appropriate. In a low flow year with greater amounts of internal load possibly mixed into the surface layer, the larger predicted value (0.095 mg/L) would be more appropriate.

This leads to the following conclusion. In high flow years, much of Fanshawe Lake's internal load leaves through the bottom outlet without contaminating the euphotic zone. Supporting evidence is: (1) bottom and outflow concentrations are much higher than the surface layer concentration (see above and Section 3.2), and (2) phosphorus export via the bottom outlet is higher than input, (3) at least in one year, 2004, phosphorus models overestimate phosphorus concentration in Fanshawe Lake when internal load is considered as being mixed into the surface layers. However, in other years when flow rates are small, export is small too, and internal load likely has a larger effect on lake TP concentration and hence algal blooms.

5. Evaluation of the Water Quality of Wildwood and Pittock Reservoirs

Wildwood Reservoir is located about 33 km upstream of Fanshawe Lake on Trout Creek, a tributary of the North Thames River, while Pittock is a dam of the South Thames River, which joins the north branch below Fanshawe Lake. Therefore the water quality of Wildwood affects that of Fanshawe Lake as it represents part of Fanshawe's inflow, while Pittock's water quality is not important for Fanshawe Lake.

Both reservoirs are said to have algal blooms and low general water quality. Unfortunately, there are not many water quality data available for these reservoirs, besides transparency measures in form of Secchi disc, and the data collected during the present study in summer 2004 (Appendix C). Some of the water quality evaluation can be based on in- and outflow quality, for which several years of nutrient and other chemical data are available. However their usefulness is limited, as (1) both reservoirs have bottom outlets, so that outflow water quality probably does not adequately represent the surface water quality during stratified periods and as (2) only small portions of the reservoir inflows are gauged. For Wildwood only 28% is gauged by the Trout Creek gage at Fairview, and for Pittock only 59% by the South Thames River gage at Innerkip.

Basic morphometric and hydrological characteristics are presented in Table 5-1. These values represent long-term annual average values. In a reservoir like Pittock, which is operated to have high summer storage capacity (286.7 m elevation) and low winter levels (282.7 m elevation), the seasonal characteristics differ substantially from the annual average.

Table 5-1. Morphometry and hydrology of Wildwood and Pittock in comparison with Fanshawe

	Wildwood	Pittock	Fanshawe
Altitude at average pool[1] (m above sea level)	322.4	284.7	262.4
Watershed area, A_d (km^2)	129	245.5	1,447.4
Surface area[1], A_o (ha)	200	148[2]	272.6
Area-Ratio, A_d/A_o	64.6	166.1	532
Maximum depth (m)	11.5	8.5	12.1
Mean depth[1], z (m)	4.24	1.9[2]	4.82
Morphometric index, $z/A_o^{0.5}$	3.0	1.6	2.93
Volume[1] (10^6 m^3)	8.48	2.83[2]	13.15
Outflow volume[1] (10^6 m^3 per yr)	65.6	101.1	560
Water residence time[1], τ (volume/outflow)	0.139 yr 50.8 d	0.031 yr 11.2 d	0.026 years or 9.5 days
Annual flushing rate[1], $\rho = 1/\tau$ (per yr)	7.2	32.5	38.4
Annual water load[1], $q_s = z/\tau$ (m/yr)	32.7	68.1	205
[1]Longterm average	*1967-2004*	*1979-2004*	*1954-2004*

[2]Summer average for Pittock: A_0: 200 ha; Mean depth: 2.3 m; Volume: 5.8 10^6 m^3

Pittock Reservoir is the smallest of the studied reservoirs, but its water residence time is almost as low as Fanshawe Lake's. Wildwood is almost as large as Fanshawe Lake, but its water residence time is 5 times higher on average. Mean depth and morphometric index are low in Pittock indicating highly mixed conditions, even in the summer, when it is operated at a higher water level. It can be anticipated from the morphometry and hydrology alone that Pittock is the most eutrophic of these reservoirs, with the most severe water quality problems.

Secchi disk transparencies in Wildwood are indeed higher (almost 150% of the long-term mean) than in Pittock, although even the most clear location still represent slightly eutrophic conditions according to the classification (Table 5-2, Table 5-3, Table 3-1). Conditions in Pittock have to be classified as hyper-eutrophic. In both reservoirs, transparency improves from the inflow to locations close to the dam, as was also found in Fanshawe Lake (Table 3-3). Different from Fanshawe Lake there is no improvement in recent years, an indication that the zebra mussel has not invaded these reservoirs yet.

Table 5-2. Summer Secchi disk transparency in Wildwood

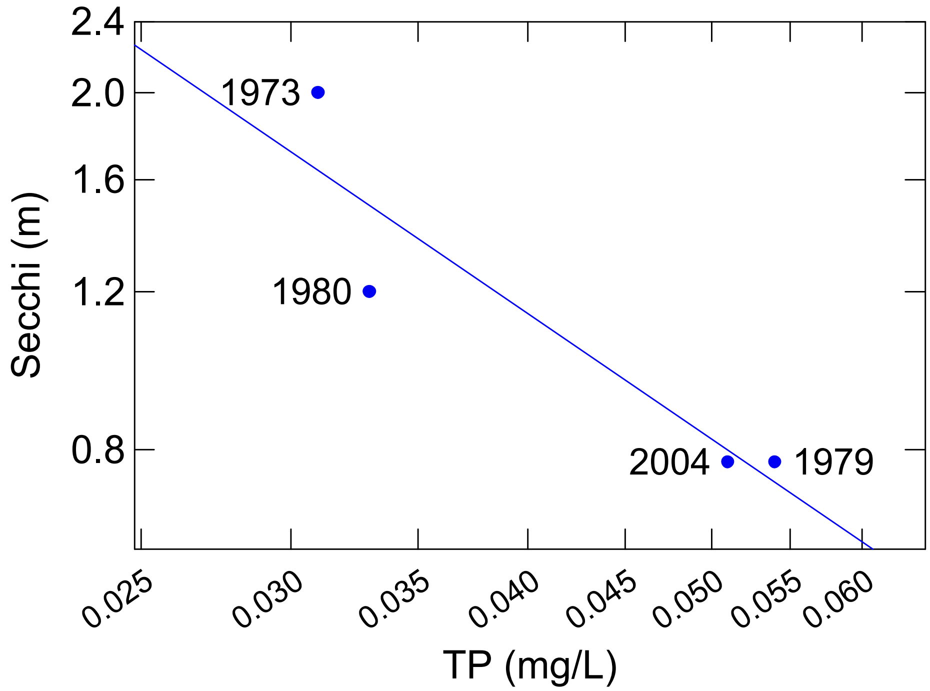
Year	Dam	Mid	Upper
1973	1.2		
1979	0.8		
1998	2.1		
1999	2.0	1.3	1.0
2001	1.3	1.1	0.5
2002	1.3	1.1	1.0
2003	1.9	1.5	1.0
2004	0.8	0.7	0.5
Average	1.3	1.1	0.8

Table 5-3. Summer Secchi disk transparency in Pittock

Year	Dam	Mid	Upper
1973	0.8		
1999	0.9	0.9	0.7
2001	0.9	0.9	0.8
2002	1.1	0.8	0.7
2003	1.2	0.6	0.6
2004	0.9	0.7	0.5
Average	0.9	0.8	0.6

There are only four summer TP averages available for Wildwood surface water with an overall average of 0.043 mg/L. This indicates eutrophic conditions, just like Secchi transparency. An expected negative correlation with Secchi transparency is supported by the available data (Figure 5-1). The two TP summer averages that are available (for 1976 and 2004) in Pittock are almost twice as high at 0.083 and 0.072 mg/L and hence indicate more eutrophic conditions.

Figure 5-1. Secchi disk transparency and TP summer surface averages in Wildwood.



Note: The line represents the regression line with n=4, R^2=0.85, p<0.10.

There is no evidence as to whether the period of low nitrate (LND) of the outflow can be used in these reservoirs as an indicator for algal blooms because of lack of data. No algal biomass data in form of chlorophyll are available, and there are only three matching Secchi or TP values per reservoir.

Nonetheless, LNDs were computed from available nitrate concentration in the outflows as they may prove to be indicative of algal blooms in the future. The 24-year average in Wildwood is 79 days, while the 22-year average in Pittock is 83 days. These values are slightly higher than in Fanshawe Lake, which 37-year average was 62. If LND means algal blooms in these reservoirs as well, it can be concluded that algal blooms are at least as frequent as in Fanshawe Lake. As in Fanshawe Lake, LNDs are significantly negatively correlated with summer inflow (Pittock: n= 22, R^2=0.49, p<0.001, Wildwood: n=24, R^2=0.34, phosphorus<0.01), indicating that water quality in these reservoirs may be highly controlled by hydrology as well.

The LNDs are presented for comparison in Figure 5-2 and Appendix D, 2. It appears that sometimes blooms (e.g. in 1998 and many earlier years) or no blooms (e.g. in 2000) occur in all three reservoirs simultaneously. Future monitoring of nitrate concentrations, Secchi disk transparency and chlorophyll concentration in conjunction with visual observations with respect to blooms would be useful to evaluate whether LND is indeed a bloom indicator in these reservoirs.

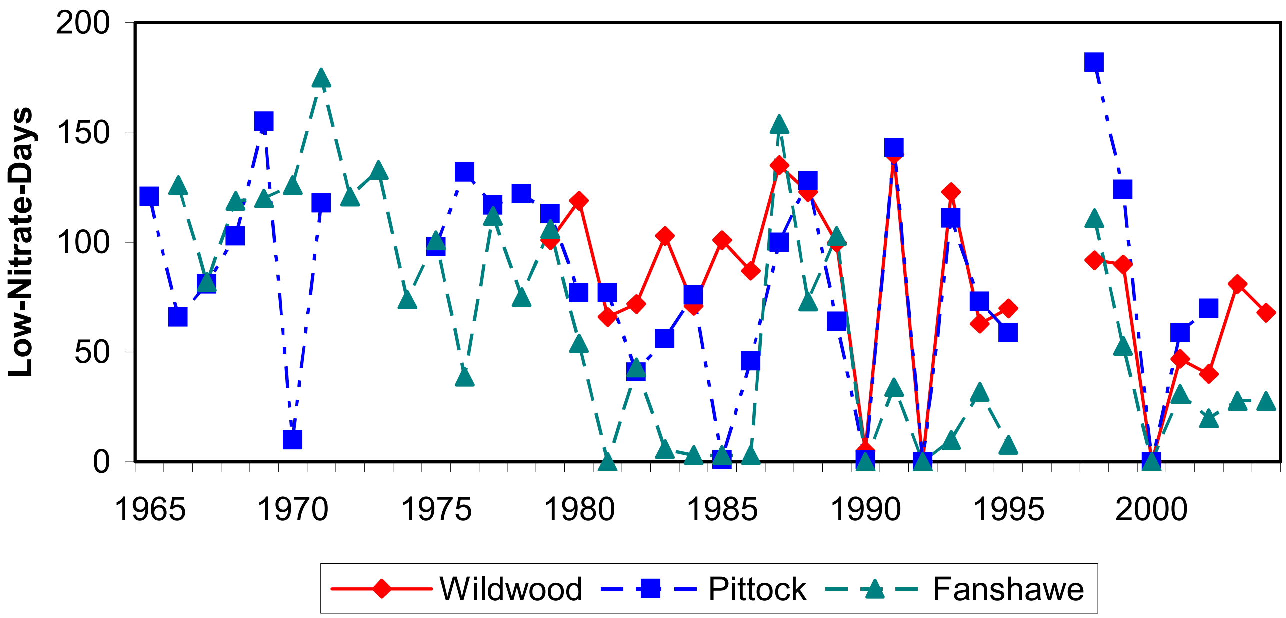


Figure 5-2. Algal bloom indicator, LND, in Outflow of the reservoirs

According to temperature and oxygen profiles, both reservoirs stratify occasionally at their dam locations and exhibit oxygen depletion down to below 1 mg/L dissolved oxygen in the summer. Such occasional anoxic periods can be expected to trigger phosphorus release from bottom sediments.

Only crude phosphorus budgets can be constructed from the gauged in- and outflows, because of the small proportion that is gauged, as already mentioned. Twenty-four years from 1979 to 2004 of inflow and outflow data with matching phosphorus concentrations for mass balances are available for Wildwood and 15 years between 1988 and 2004 for Pittock. The budgets were constructed by pro-rating the gauged proportion with the un-gauged one. Since land use appears to be similar for gauged and un-gauged portions of both reservoirs, the same average TP concentration was applied to the portion of the watersheds without monitoring data. (The long-term average inflow TP concentration is 0.096 mg/L for Wildwood and 0.100 mg/L for Pittock.) The mass balances reveal that Wildwood retains phosphorus so that less phosphorus leaves it than enters, while Pittock releases significant amounts of TP over and above the incoming load, just as does Fanshawe (Figure 5-3, Table 5-4).

Higher phosphorus retention in Wildwood may be caused by two of its characteristics. (1) Annual flushing rate is only a fifth of the other reservoirs' flushing rate, and annual water load is the lowest. These characteristics increase phosphorus retention as is obvious from the retention model (Section 4.2). (2) The other reason is the apparent low internal load, as indicated by the comparably low summer outflow TP concentration from the bottom outlet, which is on average 96% of the inflow concentration. In Pittock and Fanshawe the bottom concentrations are about 130% of the inflow concentrations (Table 5-4). Even though there is no net export of phosphorus in Wildwood, indications of some internal load exist as the bottom water concentration (via the bottom outlet, Table 5-4) is far larger than the surface summer concentration (Figure 5-1 and text above). The same was found for Pittock (and Fanshawe Lake). This indicates the likelihood of internal phosphorus load in all reservoirs. Profile sampling of phosphorus at the deepest location at the dam would verify increased hypolimnetic phosphorus concentrations.

From the analyses presented above it can be concluded that Wildwood does not adversely affect the Thames River water quality because there is no net export of phosphorus. This means that Wildwood is still retaining phosphorus in its sediments for now. Contrarily, Pittock and Fanshawe experience a net export and hence adversely affect downstream water quality. Not only is the concentration of the limiting nutrient, phosphorus, increased after the flow through these reservoirs, but it can be inferred that also algal biomass and other organic material increase after flowing through these dams. While these events decrease water quality downstream, they prevent increased eutrophication within the reservoirs and are the base for some of the suggested treatments in the following section (Section 6.2).

Table 5-4. Comparison of in- and outgoing TP concentrations and mass

	in	out	in-out/in
Wildwood			
Annual Concentration (mg/L)	0.096	0.081	
Summer Concentration (mg/L)	0.079	0.077	
Mass Balance (g/m^2/yr)	3.1	2.7	7%
Number of years	24	24	24
Pittock			
Annual Concentration (mg/L)	0.100	0.120	
Summer Concentration (mg/L)	0.058	0.092	
Mass Balance (g/m^2/yr)	5.9	8.2	-20%
Number of years	15	24	15
Fanshawe			
Annual Concentration (mg/L)	0.079	0.103	
Summer Concentration (mg/L)	0.062	0.092	
Mass Balance (g/m^2/yr)	18.8	22.4	-28%
Number of years	23	38	23

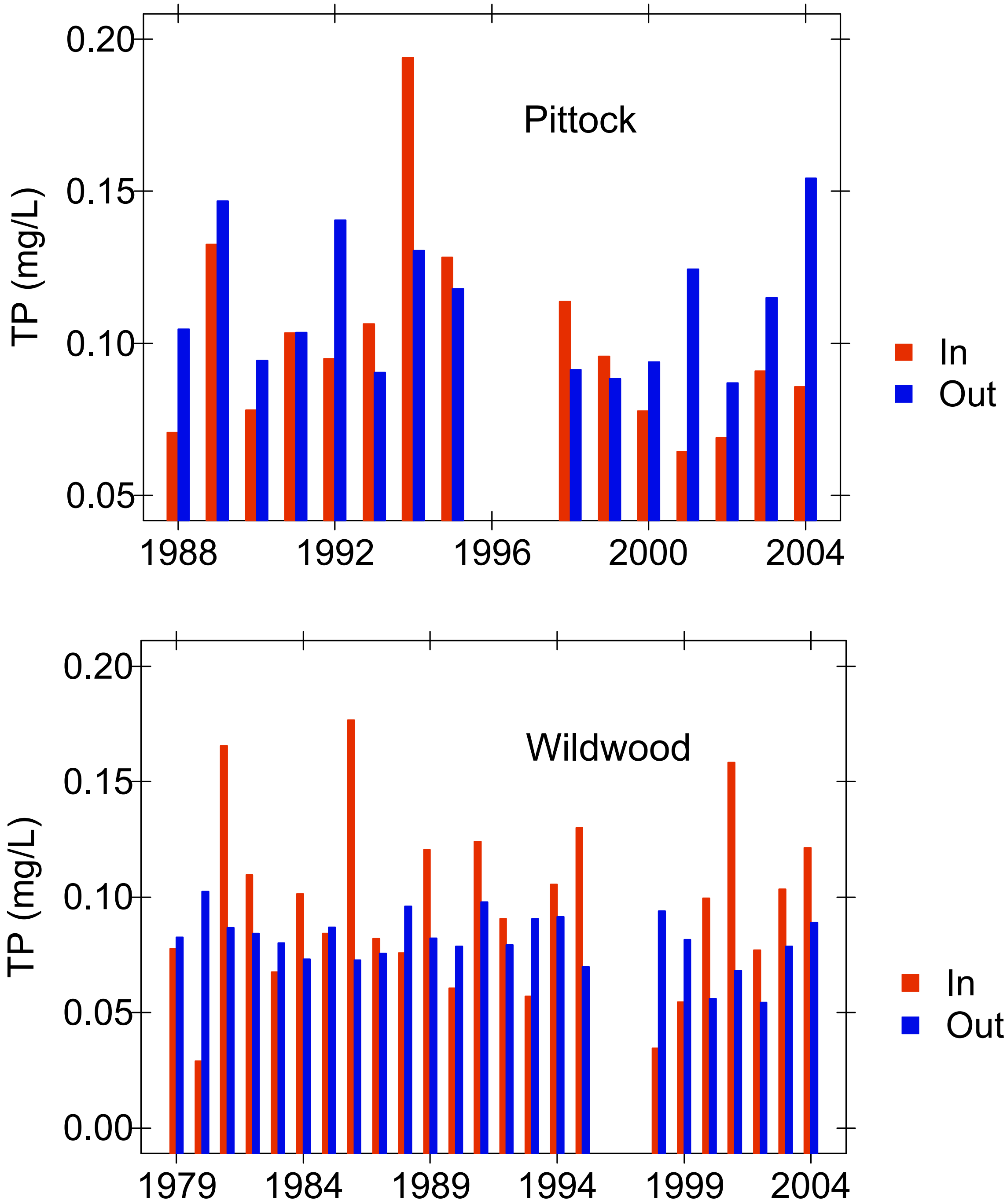


Figure 5-3. Annual and summer inflow versus outflow TP concentration

6. Treatment Options

The main purpose of the studies leading to this report is the determination of feasible treatment options that would improve the water quality of the three reservoirs and possibly the Thames River sections that the UTRCA is responsible for. However, water quality improvement cannot be the only goal, since flood control (the dam's main purpose) and other constraints like endangered species, recreational potential, costs and public acceptance all have to be considered. A detailed description of such "treatment option decision criteria" was assembled by the Project Task Force and is included as Appendix I.

In this section first, a general introduction of treatment choices used in other lakes and reservoirs is presented. Next, their application potential and probable success for Fanshawe Lake is analysed and further, the most feasible treatment options for all three reservoirs are presented. Last, advice for future activities, including monitoring, modeling, and public involvement is given.

6.1. Potential Technological Solutions

6.1.1. In general

As noted in the project proposal (Freshwater Research, June 10, 2004) the selection of restoration treatments is most critical. Not only do major monetary and human resources need to be committed for a number of years, but also a treatment has to be selected that does not do any harm, but actually improves water quality and other conditions according to the decision criteria of Appendix I. Restoration attempts are not always successful:

- Prolonged aeration/oxygenation and destratification led to a toxic algal bloom in winter 1998/99 in Lake Wilcox, Ontario (Nürnberg et al. 2003), as well as in three Swiss lakes (Bürgi and Stadelmann 2002).

- Oxygenation failed to prevent P release despite severely anoxic conditions in sediment and bottom water of several Swiss lakes, consuming a tremendous amount of financial resources (Gächter and Wehrli 1998).

- Addition of iron chloride created an algal bloom in the first year of application; it was attributed to alleviated iron limitation (Hall et al. 1994)

- In Lake Kortowski, Poland, early summer destratification was induced on purpose to shorten the period of anoxia by preferentially withdrawing lake water from the hypolimnion (Technique: hypolimnetic withdrawal, Nürnberg 1987). Consequently, nutrient mixing from the hypolimnion caused an algal bloom that was more severe than in pre-withdrawal years (Olszewski1973.).

Fortunately, many in-lake restoration attempts have been successful, as described in Cooke et al. (Cooke et al. 1993). Projects are more likely to perform as expected and attain the restoration goals when limnology, P cycles, and nutrient-algal relationships have been studied in detail.

6.1.2. Fanshawe Lake

The above mentioned treatment options and others were investigated with respect to their applicability to Fanshawe Lake. In Table 6-1 their expected performance is graded according to the Decision Criteria of Appendix I.

According to the present analysis treatment options for Fanshawe Lake are limited. The main reason for this limitation is the hydrology; it is a very fast flushing reservoir, almost like a slow river (summer water residence time is less than 20 d in Fanshawe Lake). This means that any chemical additions would have to be huge and would be flushed out before they have the opportunity to settle and improve water quality by co-precipitation of particles and nutrients. Such large additions would not only be extremely costly but may also adversely affect downstream habitat for endangered species, especially the macrobenthos, including mussels. Also, aeration and whole lake mixing or destratification would not have any beneficial effect. Instead a worsening of the water quality in the reservoir would be expected because increased mixing would bring the nutrients in the deeper, nutrient-rich water to the surface. There, they may trigger algal blooms rather then preventing them. Hypolimnetic or layer aeration is not warranted either, because the period of stratification is already short and sporadic, and may only be shortened by such a treatment.

In reservoirs where external loading is the main source of eutrophication, treatment of the inflow can help improve water quality. However, mainly because of the high flushing rate and the large volume of flow (1980 to 2004 average May to September inflow: 9.2 m^3/s and 121.6 10^6 m^3), such treatments would be too expensive and infeasible in Fanshawe Lake, because addition of chemicals for the precipitation of nutrients and organic particles would have to be too large. Even a pre-reservoir or settling basin, a recommendation that often improves conditions, may not be feasible because of these hydrological constraints.

Table 6-1. Treatment Evaluation

Treatment	Comments	Decision Criteria[1,2] 1	2	3	4	5	6
Treatment of Reservoir:							
Precipitating Chemicals:							
- Alum	Very expensive, Probably ineffective because of high flushing	+/-	?	-	+/-	---	-
- Calcium	Like alum, but less water quality improvement	+/-	??	-	+/-	---	?
- Iron	Like alum, but potential fertilization	+/-	-	-	+/-	---	?
Aeration:							
- Hypolimnetic	Not enough stratification	+/-	-	+/-	-	--	
- Whole lake	Worsen water quality	+/-	--	--	-	--	
Mixing	Worsen water quality	+/-	--	--	-	--	
Hypolimnetic withdrawal	Recommended strategy, but downstream water quality to be monitored	+/-	+	+	+/-	+	+
Treatment of Inflow:							
- Alum	Very expensive, Settling basin required Sludge build-up	-	+/-	+/-	+/-	---	-
- Settling basin	Very expensive, May not be useful because of high flushing	-	+/-	+/-	+/-	---	+

[1]Decision Criteria

1. Flood control operations
2. Water quality
3. Aquatic habitat
4. Recreational opportunity
5. Capital and operating costs
6. Public input / acceptance

[2]Key for expected effects:

+ positive
- negative, -- worse, --- worst
+/- none
? not known

6.2. Recommended Treatment Option

The analysis of Fanshawe Lake reveals the impact of hydrology and climate on water quality. Not only is flushing fast and flow volume large, but their inter-annual variation is correlated inversely with algal blooms (Section 3.5). While climate is obviously a given, flows are managed to a certain degree by the UTRCA. It appears to be a quite logical to operate the reservoir in such a way that the benefits of certain flow conditions are maximized. This means in particular, that the summer flows should be augmented so that the discharge of phosphorus-rich water from the bottom outlet is maximized. Such conditions could be enhanced by operating the reservoir so that potential early summer inflow is kept in the reservoir to make water available for late summer outflow. Such treatment is based on enhanced bottom water withdrawal and is known as the lake restoration technique of ***hypolimnetic withdrawal***. It has been found to successfully reduce phosphorus concentration in the surface and bottom water, decrease algal biomass and eventually decrease hypolimnetic oxygen depletion (e.g. Nürnberg 1987). It is especially powerful in stratified lakes, because of the potentially large export of extremely phosphorus–rich water, but has also been found to improve water quality in lakes that stratify only occasionally, like Fanshawe Lake. As long as the bottom water has higher phosphorus concentration than the surface water in the summer and fall, as is the case in Fanshawe Lake, phosphorus export during this period can be increased by hypolimnetic withdrawal.

Therefore, this treatment requires fine-tuning of reservoir operations by the UTRCA to maximise phosphorus export in late summer and fall. UTRCA staff arre presently looking at past operational journals to determine how much more flow could be released in future summers and falls under such an operation scheme. This treatment will probably not decrease eutrophication in a fast and immediately obvious way. Instead, it would be a careful and cost-effective way that is bound to benefit in the long run and may at least prevent decrease of the current water quality.

Although this treatment is based a bit on "exporting the problem", the downstream effect should not be much worse than now, because some phosphorus-rich water has been exported from Fanshawe and Pittock reservoirs in the past and at present. In the long run, the expected decreased eutrophication of thus-managed reservoirs should benefit downstream waters. To address any potential adverse effects, it is suggested to look at the Thames River watershed as a whole as elaborated in Section 6.3 below and predict any treatment effects, where possible.

Inverse relationships between the algal bloom indicator and summer flows were also observed for Wildwood and Pittock reservoirs. This suggests that a similar treatment of enhancing summer flow may decrease eutrophication in these reservoirs as well.

Although both reservoirs have bottom outlets, the benefit of hypolimnetic water export is probably less pronounced in Pittock, because it mixes more frequently. But the two available annual TP surface averages within Pittock Reservoir are about half as high as TP in the bottom outlet, suggesting that the bottom outlet and any operational changes that maximise export would decrease TP concentration in the reservoir. Because Pittock Reservoir has severe water quality problems and Secchi transparencies suggest hyper-eutrophic conditions, some additional treatment should be found. Furthermore, nutrient loading to downstream water is increased by the flow through Pittock, as determined by mass balance analysis. It can be expected that consequently organic loading, and oxygen depletion, is increased as well. The importance of these effects depends on the water quality expectation and anticipated recreational usage of that part of the South Thames River.

Hypolimnetic withdrawal would also benefit Wildwood, as it is stratified and has high phosphorus concentration in the bottom, although water quality improvement of Wildwood Reservoir may not be as crucial, as it is apparently only slightly eutrophic and still serves as a sink for nutrients. Nonetheless, improvement of Wildwood's water quality by operation of outflow may be achievable so that Wildwood reservoir has the potential of turning into an "exemplary reservoir" for the Southern Ontario agricultural region.

6.3. Future Activities

In order to find ways to improve water quality in the Upper Thames River and the reservoirs, it is important to "limnologically understand" the entire system. For example, any upstream reservoirs will affect the downstream ones (e.g. Wildwood affects Fanshawe) so that especially the more eutrophic ones (e.g. possibly Lake Mitchell and the dam at Stratford) will adversely affect the downstream river sections and reservoirs. It would be useful to apply the present exercise throughout the Upper Thames River watershed. This means, all available data should be collected and inspected; hydrology-based mass balances should be constructed for the nutrient phosphorus and future monitoring enhanced, after the most important data gaps have been recognized.

In particular, monitoring should concentrate on indicators of eutrophication. Surface and bottom phosphorus concentration, algal biomass indicators (chlorophyll concentration and Secchi transparency) and oxygen depletion are the most important determinants of eutrophication. Since algal blooms are particular problematic in the discussed reservoirs, emphasis should be put on the determination of reservoir chlorophyll concentration and Secchi disk transparency throughout the summer, and nitrate concentration to compute LNDs of reservoirs and their outflows in summer and fall. The keeping of a "Journal" on algal blooms would help corroborate the LND, algal bloom, and hydrological relationships described in this study; in addition, their applicability to less-studied reservoirs and river sections of the Upper Thames River could be determined.

Acquiring Secchi transparencies and entries for the bloom journal could involve interested and reliable stake holders of the specific reservoirs and river sections. An out-reach plan could be installed similar to the Lake Partner Program by the MOE. (In this plan registered volunteers determine Secchi disk depths throughout the summer and collect nutrient samples in prepared bottles.) Programs like this are established throughout North America and also in Europe and are very successful in producing background and baseline data on eutrophication trends. They are not only cost-effective, as they are based on free labour, but also educate the public and generate environmentally friendly positions and collaboration.

Most of the Upper Thames River watershed is agricultural and programs to reduce these impacts have been implemented for many years (Appendix I). It is conceivable, that increased monitoring and modelling efforts would help identify more severe pollution sources. As a consequence, the watershed efforts could be improved and combined with more hands-on restoration efforts of distinct river sections and reservoirs. In this way, the most promising treatments, those that are financially responsible and promise improvements with some certainty, can be prioritized. Also, the key locations (e.g. upstream important waters) can be selected and key purposes (e.g. important recreational water or endangered species habitat) determined and benefits for the reservoir weighed against disadvantages for the downstream water (as relevant in the hypolimnetic withdrawal treatment). We envision the creation of something like an Upper Thames River Master Plan (UTRMP) to combat eutrophication of the Thames River watershed under UTRCA's jurisdiction.

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Appendix A. Hypsographic information

Fanshawe Lake

Elevation (m a.s.l.)	Gauge Height (mm)	Storage (ha-m)	Area (ha)
251.128	-11000	0.00	0.00
252.128	-10000	5.00	10.00
253.128	-9000	20.00	22.50
254.128	-8000	50.00	40.00
255.128	-7000	100.00	65.00
256.128	-6000	180.00	85.00
257.128	-5000	270.00	107.50
258.128	-4000	395.00	142.50
259.128	-3000	555.00	182.50
260.128	-2000	760.00	217.50
261.128	-1000	990.00	237.44
262.128	0	1,234.88	261.00
262.228	100	1,264.00	265.74
262.328	200	1,293.50	269.79
262.428	300	1,323.20	273.82
262.528	400	1,353.30	277.87
262.628	500	1,383.60	281.89
262.728	600	1,414.30	285.94
262.828	700	1,445.30	289.98
262.928	800	1,476.50	294.00
263.028	900	1,508.10	298.04
263.128	1000	1,540.04	304.35
264.128	2000	1,877.36	338.73
265.128	3000	2,250.34	373.02
266.128	4000	2,662.78	411.52
267.128	5000	3,118.80	453.13
268.128	6000	3,622.86	508.80
269.128	7000	4,179.74	558.39
270.128	8000	4,794.53	607.92
271.128	9000	5,472.65	682.86
272.028	9900	6,141.86	775.00

Provided by UTRCA

To investigate fluctuation in area and volume of Fanshawe Lake, water level readings were consulted. Because readings are available from 1985 to present only, previous levels were estimated from a regression of available levels on inflow volume. The average annual lake level is quite constant and fluctuated only between 0.08 m in 1991 to 0.66 m in 1990 above the design level of 262.128 m.

Appendix B. Hypoxic and anoxic factors

The **Anoxic Factor** (AF, Nürnberg 1995) quantitatively summarizes the extent and duration of anoxia (lack of oxygen) in stratified lakes. It is based on a series of measured oxygen profiles and morphometric data and can be computed for any lake or reservoir. To render this index comparable across lakes of different sizes, AF is corrected for lake surface area by simple division. Expressed this way, AF is a ratio that represents the number of days in a year or season that a sediment area equal to the lake surface area is anoxic. Hence, its units are d/yr or d/season; i.e., summer or winter. Anoxic factors can be predicted from average phosphorus concentration and lake morphometry when DO profiles are not available.

To compute the Anoxic Factor, first, the oxycline must be determined from oxygen profiles. The criterion for anoxia is 2 mg/L. Next, the period of anoxia (t_i) must be multiplied by the corresponding area (a_i) and divided by the lake surface area (A_o) corresponding to the average elevation for that period. These terms of n, numbers of periods at different oxyclines (i.e. highest depths with 2 mg/L DO or below) are then added up. In this way, AF is comparable between lakes, like other areal measures, e.g., areal nutrient loads and fish yield.

The anoxic factor can be computed from following equation:

$$AF = \sum_{i=1}^{n} \frac{t_i \times a_i}{A_o}$$

where t_i, the period of anoxia (days),

a_i, the corresponding area (m^2),

A_o, lake surface area (m^2) corresponding to the average elevation for that period,

n, numbers of periods with different oxycline depths

When stratified lakes are classified with respect to trophic state, below 20 d/yr indicate oligotrophic conditions, 20 to 40 d/yr are usually found in mesotrophic lakes, 40 – 60 d/yr represent eutrophic conditions and above 60 d/yr is typical for hyper-eutrophic conditions.

The **Hypoxic Factor** summarizes the extent and duration of hypoxia, where DO concentration is below 5.5 mg/L, which are the guidelines for warm water biota by Environment Canada. It can be calculated from DO profiles like the anoxic factor, by substituting the oxycline depth with a depth for which the DO content is below 5.5 mg/L DO.

Appendix C. Additional water quality data for 2004

Fanshawe Lake						
Date	Site	Depth (m)	Layer	Secchi (m)	TP (mg/L)	Chlorophyll (µg/L)
8-Jul-04	F1		euphotic	1.9		
8-Jul-04	F2		euphotic	0.85		
8-Jul-04	F3		euphotic	0.5		
19-Jul-04	F1	1	euphotic	1.8	0.042	7.3
19-Jul-04	F1	8	8m		0.039	
19-Jul-04	F1	mob	bottom		0.055	
19-Jul-04	F2	1	euphotic	1.3	0.032	8.4
19-Jul-04	F2	mob	bottom		0.045	
19-Jul-04	F3	mob	euphotic	0.9	0.164	8.9
28-Jul-04	F1		euphotic	2.8		
28-Jul-04	F2		euphotic	1.6		
28-Jul-04	F3		euphotic	1.3		
6-Aug-04	F1	1	euphotic	2	0.035	4
6-Aug-04	F1	8	8m		0.044	
6-Aug-04	F1	mob	bottom		0.027	
6-Aug-04	F2	1	euphotic	1.1	0.044	5.9
6-Aug-04	F2	mob	bottom		0.091	
6-Aug-04	F3	mob	euphotic	0.9	0.072	11
16-Aug-04	F1	1	euphotic	3.3	0.022	4
16-Aug-04	F1	8	8m		0.044	
16-Aug-04	F1	mob	bottom		0.074	
16-Aug-04	F2	1	euphotic	1.7	0.037	13
16-Aug-04	F2	mob	bottom		0.153	
16-Aug-04	F3	mob	euphotic	1.2	0.129	8.9
3-Sep-04	F1	1	euphotic	2.1	0.041	40
3-Sep-04	F1	8	8m		0.03	
3-Sep-04	F1	mob	bottom		0.078	
3-Sep-04	F2	1	euphotic	1.1	0.13	84
3-Sep-04	F2	mob	bottom		0.058	
3-Sep-04	F3	mob	euphotic	1	0.095	100
17-Sep-04	F1	1	euphotic	1.8	0.042	3.5
17-Sep-04	F1	8	8m		0.035	
17-Sep-04	F1	mob	bottom		0.079	
17-Sep-04	F2	1	euphotic	0.8	0.104	76
17-Sep-04	F2	mob	bottom		0.093	
17-Sep-04	F3	mob	euphotic	0.6	0.118	96

Data for 28-Jul-04 were taken by University of Toronto students

mob, 1 m over bottom

For sites consult Map Figure 2-2

Year	Site	Wildwood Reservoir Secchi (m)	Wildwood Reservoir TP (mg/L)	Pittock Reservoir Secchi (m)	Pittock Reservoir TP (mg/L)
9-Jul-04	Dam	1.10		1.2	
9-Jul-04	Mid	0.60		0.8	
9-Jul-04	Upper	0.50			
22-Jul-04	Dam	0.60	0.032	1	0.054
22-Jul-04	Mid	0.60	0.058	0.8	0.064
22-Jul-04	Upper			0.6	
5-Aug-04	Dam	0.85		0.9	
5-Aug-04	Mid	0.50		0.7	
5-Aug-04	Upper	0.45		0.6	
19-Aug-04	Dam	0.60	0.069	0.7	0.089
19-Aug-04	Mid	0.80	0.039	0.5	0.131
19-Aug-04	Upper	0.60		0.4	
2-Sep-04	Dam	1.00		0.8	
2-Sep-04	Mid	0.60		0.7	
2-Sep-04	Upper	0.45		0.5	
16-Sep-04	Dam	0.50		0.7	
16-Sep-04	Mid	1.00		0.4	
16-Sep-04	Upper	0.40		0.2	

Site: sampling location at the dam (Dam), in the middle of the reservoir (Mid) and closer to the inflow (Upper)

Appendix D. Variable "Low-Nitrate-Days" (LND)

LNDs were computed from the period when nitrate concentration in the outflow was around 1 mg/L or less.

	Based on Fanshaw outflow data			Based on Lake F1		
Year	Date-beg	Date-end	total # of Days	Date-beg	Date-end	# of Days
1972	14-Jun-72	13-Oct-72	121			
1973	25-Jun-73	5-Nov-73	133			
1974	1-Sep-74	14-Nov-74	74			
1975	20-Jul-75	29-Oct-75	101			
1976	5-Sep-76	14-Oct-76	39			
1977	15-Jun-77	5-Oct-77	112			
1978	17-Jul-78	30-Sep-78	75			
1979	16-Jul-79	30-Oct-79	106			
1980	1-Sep-80	25-Oct-80	54			
1981			0			
1982	18-Aug-82	30-Sep-82	43			
1983	17-Sep-83	23-Sep-83	6			
1984	One date only		3			
1985	One date only		3			
1986	One date only		3			
1987	19-May-87	20-Oct-87	154			
1988	19-Jul-88	30-Sep-88	73	4-Jul-88	11-Sep-88	69
1989	10-Aug-89	21-Nov-89	103	31-Jul-89	19-Sep-89	50
1990			0			0
1991	17-Sep-91	21-Oct-91	34	25-Sep-91	4-Oct-91	9
1992			0			
1993	19-Sep-93	29-Sep-93	10			
1994	18-Sep-94	20-Oct-94	32			
1995	8-Oct-95	16-Oct-95	8			
1996	no data					
1997	no data					
1998	1-Aug-98	20-Nov-98	111			
1999	2-Sep-99	25-Oct-99	53			
2000			0			
2001	1-Sep-01	2-Oct-01	31			
2002	10-Oct-02	30-Oct-02	20			
2003	8-Sep-03	6-Oct-03	28			
2004	20-Sep-04	18-Oct-04	28			

Note:
Lake data are smaller than outflow data because they are availabe for a shorter season only.

Appendix D, continued

Comparison of LNDs as determined from nitrate concentration in the outflow of the three reservoirs.

	Wildwood	Pittock	Fanshawe
1965		121	
1966		66	126
1967		81	82
1968		103	119
1969		155	120
1970		10	126
1971		118	175
1972			121
1973			133
1974			74
1975		98	101
1976		132	39
1977		117	112
1978		122	75
1979	101	113	106
1980	119	77	54
1981	66	77	0
1982	72	41	43
1983	103	56	6
1984	71	76	3
1985	101	1	3
1986	87	46	3
1987	135	100	154
1988	123	128	73
1989	100	64	103
1990	5	1	0
1991	140	143	34
1992	0	0	0
1993	123	111	10
1994	63	73	32
1995	70	59	8
1996			
1997			
1998	92	182	111
1999	90	124	53
2000	0	0	0
2001	47	59	31
2002	40	70	20
2003	81		28
2004	68		28
Average	**79**	**83**	**62**

Appendix E. Comparison with worldwide regression models

Predictions

	TP observed	Chl observed	Chl f(TP)	Secchi observed	Secchi f(TP)	Secchi f(Chl)
1988	60	21.0	18.9	1.2	1.6	1.3
1989	104	73.1	30.4	1.1	1.2	0.6
1990	37	10.0	12.3	1.6	1.9	2.1
1991	60	19.4	19.0	0.9	1.5	1.4
2004	36	11.8	12.3	2.2	1.9	1.9
Average	**59.4**	**27.0**	**18.6**	**1.4**	**1.6**	**1.5**
Median	**60.0**	**19.4**	**18.9**	**1.2**	**1.6**	**1.4**

Secchi predictions include a color value of 17.3 platinum values as second variable

Predictions (shaded areas) are based on regression equations for Eastern North American lakes as listed in Table 3 of Nürnberg 1996. The predictor variable is indicated as “function of” or “f(_)”.

Appendix F. Physical Variables for Fanshawe Lake

Year	Precipitation	Elevation	Volume	Area	Flows (10^6 m^3)		tau_yrs	tau_days	qs
	(mm/yr)	(m asl)	(ha m)	(ha)	in	out	(yrs)	(days/yr)	(m/yr)
1954	1046.5	262.4658	1323.2	273.8	671.3	652.2	0.020	7.4	238.2
1955	781.5	262.3996	1293.5	269.8	365.2	404.1	0.032	11.7	149.8
1956	987.8	262.4577	1323.2	273.8	633.7	638.0	0.021	7.6	233.0
1957	961.5	262.4503	1323.2	273.8	599.6	611.4	0.022	7.9	223.3
1958	678.9	262.3701	1293.5	269.8	228.4	230.8	0.056	20.5	85.6
1959	1002.6	262.459	1323.2	273.8	639.7	677.4	0.020	7.1	247.4
1960	809.2	262.4371	1323.2	273.8	538.0	556.1	0.024	8.7	203.1
1961	882.4	262.3909	1293.5	269.8	325.3	318.1	0.041	14.8	117.9
1962	793.7	262.3805	1293.5	269.8	276.8	286.3	0.045	16.5	106.1
1963	543.6	262.39	1293.5	269.8	320.2	328.5	0.039	14.4	121.7
1964	888.4	262.394	1293.5	269.8	339.6	323.7	0.040	14.6	120.0
1965	964.1	262.4597	1323.2	273.8	643.1	594.7	0.022	8.1	217.2
1966	973.6	262.4395	1323.2	273.8	549.7	527.0	0.025	9.2	192.5
1967	1009.3	262.5152	1323.2	273.8	899.5	869.5	0.015	5.6	317.5
1968	1118.7	262.4483	1323.2	273.8	590.9	549.9	0.024	8.8	200.8
1969	868.3	262.4484	1323.2	273.8	590.4	603.7	0.022	8.0	220.5
1970	891.1	262.426	1293.5	269.8	487.3	438.2	0.030	10.8	162.4
1971	699	262.4138	1293.5	269.8	430.2	408.2	0.032	11.6	151.3
1972	1050.7	262.4383	1323.2	273.8	544.5	525.7	0.025	9.2	192.0
1973	941.6	262.4551	1323.2	273.8	621.8	610.8	0.022	7.9	223.1
1974	864.1	262.4467	1323.2	273.8	580.0	576.3	0.023	8.4	210.5
1975	949.5	262.443	1323.2	273.8	563.4	578.9	0.023	8.3	211.4
1976	1110.9	262.4648	1323.2	273.8	669.8	710.8	0.019	6.8	259.6
1977	1002.5	262.4556	1323.2	273.8	633.9	643.7	0.021	7.5	235.1
1978	816	262.4445	1323.2	273.8	571.2	595.6	0.022	8.1	217.5
1979	918.3	262.4753	1323.2	273.8	710.4	737.7	0.018	6.5	269.4
1980	943.2	262.4115	1293.5	269.8	420.3	422.2	0.031	11.2	156.5
1981	979.2	262.4413	1323.2	273.8	559.8	576.9	0.023	8.4	210.7
1982	1084.8	262.4908	1323.2	273.8	788.8	787.3	0.017	6.1	287.5
1983	1037.7	262.4501	1323.2	273.8	597.5	571.1	0.023	8.5	208.6
1984	1149.7	262.4626	1323.2	273.8	660.2	652.6	0.020	7.4	238.3
1985	1177.2	262.5988	1353.3	277.9	917.0	875.7	0.015	5.6	315.1
1986	1079.6	262.5784	1353.3	277.9	806.3	845.1	0.016	5.8	304.1
1987	841.9	262.3663	1293.5	269.8	455.9	446.3	0.029	10.6	165.4
1988	922.3	262.5063	1323.2	273.8	478.9	427.6	0.031	11.3	156.2
1989	813	262.4805	1323.2	273.8	343.5	321.2	0.041	15.0	117.3
1990	1300.4	262.7827	1414.3	285.9	820.3	729.6	0.019	7.1	255.2
1991	901	262.2147	1234.878	261.0	530.2	555.3	0.022	8.1	212.8
1992	1259.5	262.5955	1353.3	277.9	945.7	935.1	0.014	5.3	336.5
1993	927.5	262.4088	1293.5	269.8	551.6	572.5	0.023	8.2	212.2
1994	873.8	262.2969	1264	265.7	492.5	487.7	0.026	9.5	183.5
1995	958.7	262.3949	1293.5	269.8	656.9	631.4	0.020	7.5	234.0
1996	1243.3	262.5045	1323.2	273.8	850.6	865.2	0.015	5.6	316.0
1997	994.8	262.482	1323.2	273.8	660.9	669.8	0.020	7.2	244.6
1998	751.8	262.3614	1293.5	269.8	366.3	333.3	0.039	14.2	123.5
1999	844	262.4178	1293.5	269.8	304.0	237.9	0.054	19.8	88.2
2000	1137.1	262.7148	1383.6	281.9	689.2	594.2	0.023	8.5	210.8
2001	956.2	262.5166	1323.2	273.8	973.7	535.2	0.025	9.0	195.5
2002	870.7	262.369	1293.5	269.8	540.5	425.7	0.030	11.1	157.8
2003	989.7	262.3974	1293.5	269.8	685.8	537.1	0.024	8.8	199.1
2004	958.4	262.2687	1264	265.7	1,104.9	522.7	0.024	8.8	196.7
Average	951.94706	262.4487	1314.57	272.6	592.6	559.9	0.026	9.5	205.0
Median	956.2	262.4467	1323.2	273.8	590.4	572.5	0.023	8.4	210.7
Min	543.6	262.2147	1234.878	261.0	228.4	230.8	0.014	5.3	85.6
Max	1300.4	262.7827	1414.3	285.9	1,104.9	935.1	0.056	20.5	336.5
n	51	51	51	51.0	51.0	51.0	51.000	51.0	51.0

Appendix F, continued

Partitioning of total flows "in" of previous table

Partitioned Inflows (10^6 m^3)

Year	PloverMills	Wye Creek	Precip	Immediate Runoff
1954	637.8	16.2	2.9	14.4
1955	346.5	8.8	2.1	7.8
1956	602.1	15.3	2.7	13.5
1957	569.7	14.5	2.6	12.8
1958	216.2	5.5	1.8	4.9
1959	607.8	15.5	2.7	13.7
1960	511.3	13.0	2.2	11.5
1961	308.1	7.8	2.4	6.9
1962	262.1	6.7	2.1	5.9
1963	304.1	7.7	1.5	6.8
1964	321.8	8.2	2.4	7.2
1965	611.2	15.6	2.6	13.8
1966	522.0	13.3	2.7	11.7
1967	855.7	21.8	2.8	19.3
1968	560.9	14.3	3.1	12.6
1969	561.1	14.3	2.4	12.6
1970	462.7	11.8	2.4	10.4
1971	408.7	10.4	1.9	9.2
1972	516.8	13.2	2.9	11.6
1973	590.8	15.0	2.6	13.3
1974	553.6	11.5	2.4	12.5
1975	537.4	11.3	2.6	12.1
1976	633.7	18.9	3.0	14.3
1977	592.9	24.9	2.7	13.3
1978	544.0	12.7	2.2	12.2
1979	679.6	13.0	2.5	15.3
1980	398.8	10.1	2.5	9.0
1981	529.9	15.4	2.7	11.9
1982	748.2	20.8	3.0	16.8
1983	568.7	13.2	2.8	12.8
1984	623.7	19.3	3.1	14.0
1985	868.6	25.6	3.3	19.5
1986	769.1	17.0	3.0	17.3
1987	433.0	10.8	2.3	9.7
1988	456.0	10.1	2.5	10.3
1989	325.2	8.8	2.2	7.3
1990	776.6	22.5	3.7	17.5
1991	503.7	12.8	2.4	11.3
1992	899.1	22.9	3.5	20.2
1993	524.0	13.3	2.5	11.8
1994	467.8	11.9	2.3	10.5
1995	624.4	15.9	2.6	14.0
1996	808.4	20.6	3.4	18.2
1997	628.0	16.0	2.7	14.1
1998	347.6	8.8	2.0	7.8
1999	287.9	7.3	2.3	6.5
2000	654.6	16.7	3.2	14.7
2001	926.7	23.6	2.6	20.9
2002	513.5	13.1	2.3	11.6
2003	651.9	16.6	2.7	14.7
2004	1,051.9	26.8	2.5	23.7
Average	562.9	14.5	2.6	12.7
Median	560.9	13.3	2.6	12.6
Min	216.2	5.5	1.5	4.9
Max	1,051.9	26.8	3.7	23.7
n	51	51	51	51

Appendix G. Mass Balance Assumptions and Computations

Flows and concentrations for TP of the various external load sources to Fanshawe Lake were determined as described below. Whenever possible, loads for the period 1954 to 2004 were computed. For comparison, loads and volumetric averages were also determined for SRP.

Main Fanshawe Lake inflow and outflow: Measured flows and concentrations were provided by UTRCA. Concentrations were then volume-weighted to arrive at annual, seasonal and monthly loads.

Wye Creek: 1974 – 1990 flows were provided by the UTRCA. Based on a regression of these data with the main inflows, flows were predicted for all other years. The average of available concentration data for two years (0.083 mg/L TP for 2003, 2004) was multiplied with the flows to arrive at annual loads.

Precipitation: Precipitation height was provided. Based on values measured by the Ontario Ministry of the Environment a value of 0.030 mg/L TP was used to compute loads.

Immediate runoff: 2.25% of main inflow. (Since UTRCA indicated that the main inflow represents 95% of total flows, the immediate runoff volume was computed so that the sum of the flows besides the main inflow add up to 5%.) The Wye Creek concentration was assumed to be representative for the immediate runoff: 0.083 mg/L.

Phosphorus Loads (metric tonnes)

Year	Plover Mills	Wye Creek	Precip	Runoff	Total In	Total Out	in-out Minimum internal	(in-out)/in Retention
1954		1.3	0.1	1.2				
1955		0.7	0.1	0.6				
1956		1.3	0.1	1.1				
1957		1.2	0.1	1.1				
1958		0.5	0.1	0.4				
1959		1.3	0.1	1.1				
1960		1.1	0.1	1.0				
1961		0.7	0.1	0.6				
1962		0.6	0.1	0.5				
1963		0.6	0.0	0.6				
1964		0.7	0.1	0.6				
1965		1.3	0.1	1.1		45.7		
1966		1.1	0.1	1.0		57.7		
1967		1.8	0.1	1.6		109.8		
1968		1.2	0.1	1.0		61.1		
1969		1.2	0.1	1.0		104.1		
1970		1.0	0.1	0.9		52.3		
1971		0.9	0.1	0.8		47.1		
1972		1.1	0.1	1.0		62.7		
1973		1.3	0.1	1.1		102.5		
1974		1.0	0.1	1.0		69.1		
1975	41.3	0.9	0.1	1.0	43.3	62.9	19.5	-45%
1976	69.4	1.6	0.1	1.2	72.3	71.8	-0.4	1%
1977	61.3	2.1	0.1	1.1	64.6	74.8	10.2	-16%
1978	39.5	1.1	0.1	1.0	41.6	54.5	12.9	-31%
1979	60.3	1.1	0.1	1.3	62.7	73.3	10.6	-17%
1980	42.8	0.8	0.1	0.7	44.4	29.5	-15.0	34%
1981	25.3	1.3	0.1	1.0	27.7	47.1	19.4	-70%
1982	74.9	1.7	0.1	1.4	78.1	102.7	24.7	-32%
1983	28.6	1.1	0.1	1.1	30.8	41.0	10.2	-33%
1984	71.5	1.6	0.1	1.2	74.4	77.2	2.8	-4%
1985	65.6	2.1	0.1	1.6	69.5	86.9	17.4	-25%
1986	87.1	1.4	0.1	1.4	90.0	100.2	10.2	-11%
1987	26.5	0.9	0.1	0.8	28.3	34.5	6.2	-22%
1988	26.8	0.8	0.1	0.9	28.6	32.7	4.1	-14%
1989	25.8	0.7	0.1	0.6	27.2	43.8	16.7	-61%
1990	37.0	1.9	0.1	1.5	40.4	49.2	8.7	-22%
1991	40.0	1.1	0.1	0.9	42.1	47.8	5.7	-13%
1992	73.2	1.9	0.1	1.7	76.9	129.2	52.3	-68%
1993	33.2	1.1	0.1	1.0	35.4	46.5	11.1	-31%
1994	34.7	1.0	0.1	0.9	36.6	42.8	6.1	-17%
1995	21.7	1.3	0.1	1.2	24.3	60.0	35.7	-147%
1996		1.7	0.1	1.5				
1997		1.3	0.1	1.2				
1998		0.7	0.1	0.7		27.4		
1999		0.6	0.1	0.5		13.9		
2000		1.4	0.1	1.2		48.7		
2001		2.0	0.1	1.7		43.4		
2002		1.1	0.1	1.0		21.4		
2003	54.4	1.4	0.1	1.2	54.4	48.8	-5.7	10%
2004	87.2	2.2	0.1	2.0	87.2	102.4	15.1	-17%
Average	49.1	1.2	0.1	1.1	51.3	61.2	12.1	-28%
Median	41.3	1.1	0.1	1.0	43.3	53.4	10.2	-22%
Min	21.7	0.5	0.0	0.4	24.3	13.9	-15.0	-147%
Max	87.2	2.2	0.1	2.0	90.0	129.2	52.3	34%
n	23	51	51	51	23	38	23	23

Appendix H. Sediment sampling

Fanshawe Lake sediment samples were collected from three locations in mid-channel on July 28, 2004. One sediment core for each site was taken from an anchored boat, using a "Kajak-Brinkhurst" Corer. The surface of each core was sectioned into 0-5 cm and 5-10 cm lengths, which were placed into plastic zip bags and stored on ice in the dark. Upon return to the laboratory the samples were dried and stored dry until analysis in November 2004.

Samples were collected near the three routine lake stations (Fig. 2-2):

Site	Depth (m)	pH	Secchi depth (m)
Site A (dam), F1	9.5	8.2	2.8
Site B, F2	3.7	8.5	1.6
Site C (shallow end), F3	1.9	8.6	1.3

Sediment samples were analyzed for % water (Standard Error, SE circa 0.02), % organic matter (SE circa 0.02), and mg/g dry weight of total phosphorus, total silicate, total aluminum, total calcium, total iron, and total manganese.

Sample	Water (% of freshwt)	Organic Matter (% of dryweight)	**P** (mg/g dryweight)		**Al**	
A(0-5)	**67.38**	**7.84**	**1.38**	0.00	**24.00**	0.38
A(5-10)	**62.87**	**9.26**	**1.36**	0.04	**27.18**	0.79
B(0-5)	**70.44**	**10.00**	**1.35**	0.00	**23.71**	0.61
B(5-10)	**64.71**	**6.73**	**1.27**	0.02	**14.29**	0.85
C(0-5)	**67.67**	**8.57**	**1.28**	0.01	**16.95**	0.49
C(5-10)	**62.67**	**9.53**	**1.25**	0.03	**19.08**	0.27

Sample	**Ca**		**Fe** (mg/g dryweight)		**Mn**		**Si**	
A(0-5)	**119.61**	0.42	**31.1**	0.4	**0.99**	0.01	**36.71**	0.77
A(5-10)	**110.10**	3.10	**32.6**	1.0	**0.96**	0.03	**38.26**	0.63
B(0-5)	**103.85**	0.39	**28.9**	0.3	**0.90**	0.01	**35.95**	0.80
B(5-10)	**105.91**	1.08	**23.8**	0.9	**0.77**	0.01	**28.68**	2.58
C(0-5)	**109.00**	1.14	**24.0**	0.3	**0.79**	0.01	**28.40**	1.54
C(5-10)	**119.62**	1.58	**23.9**	0.3	**0.75**	0.01	**31.37**	0.62

Appendix I. Reservoir Treatment Option Decision Criteria

Assembled by Chris Harrington with input by the members of the Project Management Team

Reservoir Treatment Option Decision Criteria: ***FEB. 11, 2005***

When proposing treatment options to improve water quality in Fanshawe Lake a range of criteria need to be considered. The decision criteria reflect the multiple functions and uses associated with Fanshawe reservoir and how it is operated. The criteria listed below must be considered when looking at treatment options to ensure current functions are maintained and to ensure improvements in the reservoir are considered in combination with downstream effects.

Criteria	Description
1. Compatibility with flood control operations	The Upper Thames River Conservation Authority (UTRCA) in conjunction with its municipal partners owns, operates and maintains a system of structures that reduce damages caused by flooding. The primary purpose of Fanshawe Dam and Reservoir is to assist in flood control efforts to reduce the risk of loss of lives and property damages due to flooding. Fanshawe Dam can reduce flood peaks downstream by up to 40%. Treatment options must consider the flood control function of Fanshawe Dam and Reservoir as paramount respecting the intended purpose of the structure.
2. Water quality improvement	Protection and enhancement of water quality is a core business function entrenched in UTRCA's mission statement. The UTRCA has worked to implement upstream diffuse source pollution control over the last 25 years. Efforts to improve water quality in the watershed will continue through ongoing collaborative efforts to implement best management practices and conduct research and monitoring. Poor water quality in the three UTRCA managed reservoirs (Fanshawe, Wildwood and Pittock) has been a concern for the past two decades. It is recognized that treatment may be necessary to improve water quality in the reservoirs. Reservoir water quality varies from summer to summer depending on flow conditions. Indicators of poor water quality conditions in the reservoirs include elevated phosphorus and chlorophyll concentrations (correlated with algal growth) and decreased oxygen concentration. While improved water quality in the reservoirs is desired it needs to be considered in conjunction with downstream conditions. Treatment options must consider overall water quality conditions and avoid improving reservoir conditions at the expense of downstream conditions.

3. Aquatic habitat improvement	The Thames River and its tributaries are rich in aquatic life, with approximately 90 species of fish, 32 species of freshwater mussels and 30 species of reptiles and amphibians. The UTRCA co-chairs a recovery team responsible for the development of the Thames River Aquatic Ecosystem Recovery Strategy aimed at improving species at risk populations and limiting threats to these species and their associated habitats. The draft recovery strategy identifies turbidity, nutrient loadings, toxic compounds, altered water flow, barriers to movement, non-native species, disturbance and thermal pollution as the main threats. At risk species including Eastern Spiny Softshell Turtle, Eastern Hognose Snake, Queen Snake, Greenside Darter, Silver Shiner and Black Redhorse are found in both the upstream and downstream subwatersheds that are divided by Fanshawe dam (Plover Mills subwatershed and The Forks subwatershed). Specifically there are significant populations of Eastern Spiny Softshell Turtles and Queen Snakes found in close proximity to the outflow of Fanshawe dam. Species at risk are sensitive to environmental change and, like a canary in a coal mine, give warning signs of overall environmental health. Threats to aquatic species and habitat need to be minimized when considering treatment options for Fanshawe reservoir.
4. Recreational opportunity	Fanshawe reservoir is the focus of many recreational activities, some of which include rowing, camping, swimming, hiking, recreational boating, fishing etc. Recreational opportunities associated with the reservoir offer an enhancement to user's quality of life and allow the demonstration of conservation and management of natural areas. Recreational users of Fanshawe reservoir are well aware of water quality issues in the reservoir and are stakeholders for improved conditions. However, treatment options could potentially impact recreational activities enjoyed by reservoir users. When evaluating treatment options effort to minimize impact to the recreational opportunities associated with the reservoir need to be considered.
5. Capital and operating costs	All cost associated with treatment options need to be considered, including capital, operating and any costs associated with altering current operations (IE. Hydro-electric power generation). Some treatment options may be prohibitive based on capital cost and/or long term operating cost. Costs associated with treatment options will need to be fiscally responsible considering available resources and the expected outcomes.

6. Public input / acceptance	The UTRCA believes strongly in partnership and community collaboration as part of our projects and programs. Engaging the community helps develop value for a healthy environment and sound resource management. Involving the community in the project early fosters support and allows them to be partners in the goals to be realized. It is expected that public input for lake treatment efforts will come from the recreational user community, but will not be limited to that group. Lake treatment options need to be considered and evaluated openly with opportunity for public input. Consultation to present stakeholders with issues, options and the scientific rationale for these options needs to be included in the process.