Water Quality Assessment in the Thames River Watershed - Nutrient and Sediment Sources

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Cover map UTRCA 2000

Executive Summary

The Thames River has experienced excess nutrient levels for decades resulting in nutrient enrichment in the river system. Therefore, the Lake Erie Binational Nutrient Management Strategy, a product of the Lake Erie Lakewide Management Plan, identified the Thames River as one of the priority rivers delivering excess phosphorus to Lake Erie and the watershed a key Ontario watershed impacting Lake Erie's West Basin.

This study is focused on analysing best available water quality (WQ) and flow data to understand nutrient and sediment source areas and timing of delivery throughout the Thames River system. Determining the range of specific sources of nutrients and types of sediment is beyond the scope of this project and should be addressed in future studies.

This study is the first to summarize long-term routine monitoring data for the entire Thames River system together. The large temporal (up to 8 times per year for 24 years within 1986-2012) and spatial (83 stations) sampling of WQ combined with the extensive coverage by daily flows from 26 gauges makes it possible to describe and assess the variation of nutrients and sediments throughout the Thames River watershed.

The following nutrients were examined: total phosphorus (TP), dissolved reactive phosphorus (DRP), the sum of nitrate and nitrite (NO_3+NO_2 or NO32), and total nitrogen (TN) as the sum of NO_3+NO_2 and Kjeldahl-N. Suspended sediments were examined as total suspended solids (TSS) or "particulate residue".

We used three different models to calculate flow-weighted average chemical concentration and loads depending on data availability. Both EGRET (based on USGS program of *Weighted Regressions on Time, Discharge and Season*) and GAM (*General Additive Model* based on an optimally weighted regression with smoothing) include a relationship between flow and concentration in the model, and their results were used in this analysis wherever possible. LINEAR does not include any flow relationship and the results are mainly useful for comparison with previous studies. Flow-weighted average concentrations (FWC) are chemical loads divided by the total flow over the period of interest.

There are no temporal trends in river flows, but extreme seasonal differences with the largest flows in late winter and spring. There are some temporal and spatial patterns of nutrient and sediment FWC concentrations. FWC-TP decreased significantly with time from 1986 to 2012 along the Thames River below the Forks (TR), the South Thames River branch (STR), and possibly the North Thames River branch (NTR), while no consistent patterns were detectable for the other study variables. Often FWCs increased in the spring, coinciding with flows. Summer FWCs could be elevated (TP because of internal load from sediments) or decreased (TN, NO34 due to biogenic uptake).

Spatial trends include significantly decreasing DRP, TN and NO32 from the headwater stations of the STR and the NTR to the Forks, but this pattern is not significant for FWC-TP. FWC-TP and FWC-NO32 decrease in the lower reaches of the Thames River, while FWC-DRP and FWC-TN remain relatively constant. FWC-TSS significantly decreases in the NTR, but increases in the TR towards the mouth; there is no trend in the STR.

Loads are highly dependent on flows so that they increase from the headwaters towards the Forks, where they more than double, and further towards the mouth. Loads also follow the seasonal pattern of flows so that the highest loads occur during wet periods in the winter and spring.

Detailed trends along the Thames River depend on land use, impounding, tributaries, WWTPs and unknown factors and are investigated for each station, starting at the headwaters of the NTR and STR through to the mouth at St. Clair.

Export into Lake St. Clair was computed from known and modelled contributions of various tributaries with the gauged and monitored Thames River station closest to the mouth. Estimated annual exports are (t, metric tonnes): TP, 342 t/yr; DRP, 187 t/yr; TN, 24.1 10^3 t/yr; NO32, 21.0 10^3 t/yr; TSS, 113 10^3 t/yr for an annual flow of 2,030 10^6 m^3.

Concentration and loads in the river water are affected by internal loading from bottom sediments, especially in slow-moving sections and impoundments. These events increase TP and DRP concentrations in the summer, but also possibly under ice and depend on legacy loading of the bottom sediments, temperature and flow conditions.

Impounded areas, including large reservoirs, retain and accumulate pollutants over time, a benefit that becomes reversed under certain conditions so that aged impoundments can become a source of nutrients as internal loading. Whether a reservoir becomes a nutrient contributor (e.g., Fanshawe and Pittock Lake) or remains a nutrient sink (e.g., Wildwood Lake) depends on size, flushing rate, drainage area, and previous nutrient inputs and can change over time. For example, internal load in Fanshawe and Pittock Lake contributes to the P pool in the river from May to September, but P retention is enhanced during the rest of the year. Different approaches estimated an internal P load for Fanshawe of 4-16% of the long-term annual load downstream.

Waste water treatment plant effluents are high in nutrients but low in TSS. DRP data were not consistently available but based on effluent characterization studies it can be assumed that about 30% to 50% is DRP (S. Abernethy MOECC, pers. comm. Mar. 4. 2015) and therefore highly biologically available. N-concentrations can be high, but no consistent data are available. Effluents are especially influential at low river flows during the dry summer period. Data availability varies for WWTPs across the Thames watershed (e.g., City of London has longest data record) but data are consistently available since 2000 and WWTPs’ influence is underestimated when compared to the whole study period of 1986-2012. There is evidence for much higher nutrient loads in the past, which probably accumulated in downstream sediments.

Climate change predictions involve the increase in frequency and magnitude of storms which means that nutrient and sediment load would increase.

Recommendations include

a. Monitoring along the river: More intense monitoring for extreme (low and high) flow conditions, especially where flow gauges are available. Extensive monitoring of bottom sediments for P-fractions and organic content in the Thames River (deep) channel to determine their potential of internal P loading by increased phosphorus release and hypoxia, especially in the vicinity of past and present point and non-point sources.

   Installation of ISCO automated WQ stations especially in combination with continuous flow measurement to capture water samples year round and during peak flows on the main Thames River stations.

b. Monitoring load into Lake St Clair: Create a water quality measuring station closer to the mouth below Jeanette Creek. Install continuous thermostats to determine exchange flows between the lake and river. (Perhaps already attempted by EC.)

   Determination of the effect of pumping stations on DO, nutrients, and TSS for at least one specific site in the lower reach of the TR.

c. More consistent surveillance of WWTP effluent including nitrate loads. Diminishment of bypass events and elimination of CSOs.

d. Respective spatial variation: Phosphorus loads are cumulative and contributed across the watershed with similar annual loads from NTR, STR, and about 1.5 times of those loads from TR. Implement actions to reduce nutrients in each of these 3 branches of the Thames. Where adequate monitoring exists to inform targeting, prioritize actions to subwatersheds with highest unit area TP loads.

e. Respective temporal variation: Implement actions which minimize nutrients in runoff when largest loadings occur in winter and spring high flows. Investigate causes of elevated flow-weighted concentrations throughout the year and implement actions for their reduction.

f. Non-point sources contribute a large portion of the phosphorus and sediment load annually to the Thames. Implement non-point source actions to reduce nutrient loads and concentrations across the watershed.

g. Internal loading from bottom sediments, especially in slow moving sections and impoundments contribute to phosphorus concentrations and loads. Best practices should also be targeted to larger impounded sections of the Thames to minimize internal loading over time.

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Acronyms

BMP	Best Management Practice
CoL	City of London, ON
CSO	Combined Sewer Overflow, events when sanitary and catchment runoff are connected
EC	Environment Canada
EGRET	Exploration and Graphics for RivEr Trends,US-GS model
GAM	General Additive Model
HYDAT	EC flow station. Only the last 4 digits are noted, because they all belong to the subwatershed starting with 02G
LIN	LINEAR, Non-flow-weighted Linear Interpolation loads
LTVCA	Lower Thames Valley Conservation Authority
MNR	Ontario Ministry of Natural Resources
MOE	Ontario Ministry of the Environment and Climate Change
NTR	North Thames River
OGS	Ontario Geological Survey
PWQMN	Provincial Water Quality Monitoring Network
STR	South Thames River
TR	Thames River
UTRCA	Upper Thames River Conservation Authority
WPCP	Water Pollution Control Plant
WWTP	Waste Water Treatment Plant

Glossary

Annual areal water load, q_s (m/yr): The annual outflow volume (Q, cubic m) per surface area (A_o, square m), where $q_s = Q/A_o$.

Annual water detention time or *annual water residence time*, tau (yr): lake volume (V) divided by annual outflow volume (Q), where tau= V/Q.

Anoxic factor, AF (days/summer or days/year): active period and area that releases phosphorus and contributes to internal load

Box Whisker Plots: Present a data summary in a non-parametric way (Section 2.4)

Branches of Thames River:

- TR: Thames River, 0 - 209.45 km
- NTR: North Thames River, 209.5 – 287 km. This does not include ~50 km unmonitored to the source
- STR: South Thames River, 209.5 - 283 km. This does not include ~19 km unmonitored to the source.

Cyanobacteria: Often called *bluegreens* or *bluegreen algae*, although they are a type of bacteria. They can produce toxins that can create health effects if ingested in quantity (livestock, pets).

External load, L_{ext}: The sum of annual TP inputs to a reservoir from all external sources, i.e. stream, non-point and point sources, precipitation and groundwater. Units are in t/yr or in mg per square meter of lake surface area per year ($mg/m^2/yr$). External load is a gross estimate. Much of its phosphorus is in a chemical form that is not immediately available to algae.

FWC-: Flow-weighted average concentration of study water quality variables; equals load divided by flow.

Internal load, L_{int}: Annual TP inputs from internal sources, i.e. the sediments. Units are in kg/yr or in mg per square meter of lake surface area per year ($mg/m^2/yr$). Gross estimates are usually used, but net estimates, based on mass budgets, can also be calculated. Most of the TP in L_{int} is in a chemical form (phosphate) that is highly available to phytoplankton and bacteria.

Limnological seasons used in this study: Spring: Apr, May; Summer: June, Jul, Aug, Sept; Fall: Oct, Nov; Winter: Dec, Jan, Feb, Mar

Main stem stations: Stations located directly on the Thames River or its branches and not on any tributaries.

Model used to compute FWC and loads:

EGRET, detailed USGS-based model, stations need at least 7 years and 100 sample points

GAM, General Additive Model optimally weighted regression with smoothing after log-log transformations

LINEAR, time-weighted loads computed by linear interpolation of concentration over time

Secchi disk transparency: The depth at which the round black and white Secchi disk disappears is an integrated measure of algal biomass. Because its use is wide-spread many relationships with nutrients and chlorophyll concentration from other lakes are available (as regression equations).

Sediment oxygen demand (SOD): organically enriched bottom sediment takes up oxygen from the overlaying water which creates anoxic conditions

Nitrogen: Studied N-compounds: Total nitrogen (TN): Sum of nitrate and nitrite (NO_3+NO_2 or NO32) and Kjeldahl-N (TKN)

Phosphorus:

Total phosphorus (TP): All phosphorus (P) that can be analyzed in a water or sediment sample. It includes phosphate (highly available for algae), particulate forms (includes algae and non-living suspended particles), and forms not easily available to algae.

Dissolved reactive phosphorus, DRP (also "SRP" – soluble reactive P): The phosphorus compound in water that most accurately resembles phosphate. It is deemed to be highly biologically available, but analytically demanding.

Total suspended solids (TSS), also "particulate residue"

Reservoir sections: *Lacustrine*, downstream, close to dam, deepest locations resembling a lake,

Riverine, upstream, deepening and widening section at river inflow

1 Introduction

The Thames River has experienced excess nutrient levels for decades resulting in nutrient enrichment in the river system. Therefore, the Lake Erie Binational Nutrient Management Strategy, a product of the Lake Erie Lakewide Management Plan, identified the Thames River as one of the priority rivers delivering excess phosphorus to Lake Erie and the watershed a key Ontario watershed impacting Lake Erie's West Basin. A survey of 30 Ontario rivers (Ministry of Environment, 2013) classifies the Thames River for the third highest TP concentration (median 0.088 mg/L) after the Grand (0.089 mg/L) and the Don River (0.150 mg/L).

This study is focused on analysing best available water quality and flow data to understand nutrient and sediment source areas and timing of delivery through the Thames River system. Quantifying the range of specific sources of nutrients and types of sediment is beyond the scope of this project. Future work to assess sources related to land use in high priority areas and seasons identified in this project, would benefit watershed implementation programs to reduce nutrient and sediment loads.

The overall goal of this project is to better understand nutrient and sediment sources, fate, and delivery throughout the Thames River system. Individual project goals are (as stated in the RFP):

(a) To develop an understanding of the relative contribution of nutrients from point source and non-point source areas across the Thames River watershed.
(b) To provide an evaluation of the fate and delivery of various forms of phosphorus (P) and sediment from headwaters to the mouth of the Thames; including effect of impoundments, river processes, landscape features, watershed physical characteristics, and seasonal effects.
(c) To determine future scenarios for river water quality/nutrient levels based on climate and extreme weather patterns.
(d) To develop recommendations for addressing nutrient loads including priority sources and priority areas of the watershed.

To accomplish these goals, five water quality variables were explored and summarized in various ways: used as raw monitoring results, time averaged, and flow averaged by several models that include seasonality and long-time trends to various extents. While raw monitoring values present a snapshot in time and space (e.g., Section 5.1.1.1), overall temporal and spatial patterns are most conclusive when analyzed using more sophisticated models (e.g., Section 4).

Certain general characteristics of the watershed and the river that have consistent impact on water quality are described separately in Section 3. Section 4 presents some temporal and spatial trends along each main branch of the river, while Section 5 goes into more detailed specifics for each water quality station. The likely influence of predicted climate change are explored in Section 6. Finally, recommendations are presented throughout this analysis and are summarized in Section 7.

This report does not have to be read in sequence. Information is collected in specific sections and referred to in other sections located before or after and can be accessed by links in the electronic file. Acronyms are listed above followed by an extensive Glossary that explains terms and definitions used here.

2 Methods

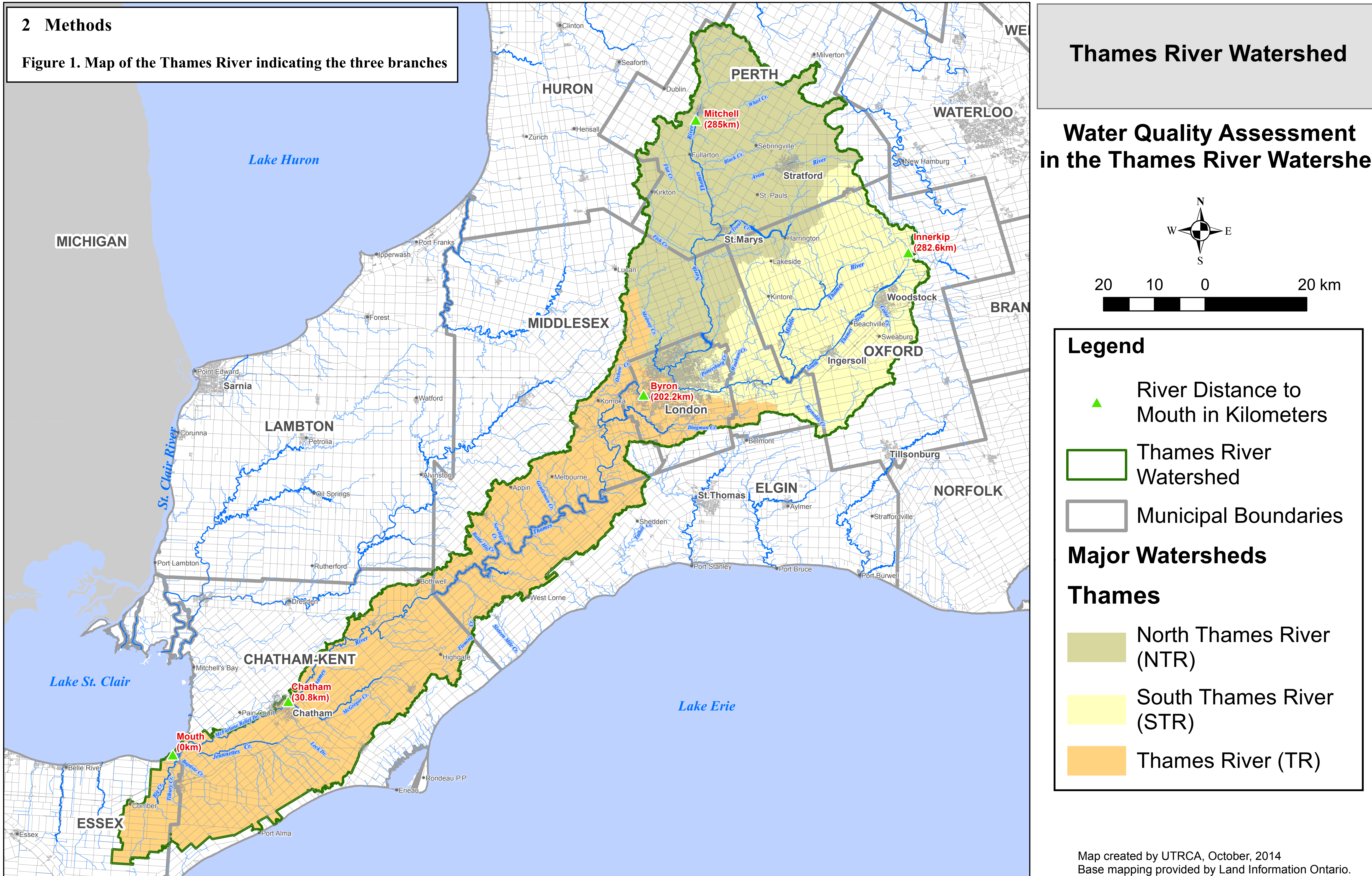


Figure 1. Map of the Thames River indicating the three branches

2.1 Division of the Thames River watershed in this study

The Thames River was divided into three parts (Figure 1):

- TR: Thames River, 0 - 209.5 km
- NTR: North Thames River, 209.5 – 287 km. This does not include ~50 km unmonitored to the source
- STR: South Thames River, 209.5 - 283 km. This does not include ~19 km unmonitored to the source. (In this study, occasionally a nominal 100 km were added for separation from NTR on graphs.)

The NTR and STR branches have similar length and drainage area (Table 1). Analysis was often done separately for these branches.

Table 1. Basic characteristics of the Thames River Branches at flow gauges closest to the confluence of NTR and STR at the forks (209.4 km)

Characteristics	NTR	STR	TR	TR at Mouth
Hydrology Station	D003	D001	E003	Na
Location (river km)	223.1	213.3	202.2	0
Drainage area (km^2)	1,426.7	1,345.5	3,089.0	5,692
Annual flow rate statistics (10^6 m^3/yr):				
Arithmetic Mean	601	539	1,766	Na
Geometric Mean	566	513	1,682	Na
Minimum	299	235	872	Na
Maximum	1,018	855	2,986	Na

Na, no flow gauges are available at the mouth and flow has to be modeled (Method Section Appendix B)

2.2 Data source

The period 1986 - 2012 was analysed to investigate time periods that are representative of relatively recent conditions and still have enough data available for many stations to conduct a detailed analysis. Previous studies found that total phosphorus concentration was elevated in the seventies (Nürnberg and LaZerte, 2005, 2006), so this earlier data was excluded.

Annual average as well as growing season (May-Sept) averages were computed. While annual loads and conditions provide a comparable baseline, the growing season is crucial for water quality issues, especially phytoplankton growth and cyanobacteria proliferation. Occasionally, winter or spring (Mar-Apr) averages were investigated because they typically provide the most flow and load per month.

Flow and water quality data from stations for the whole Thames River and its tributaries were used and assigned the distance in km upstream of the inflow into Lake St. Clair (Staff of UTRCA). Locations of water quality monitoring stations, flow gauges, and waste water treatment plants are listed in a table and presented on maps in Appendix A.

Long-term daily flow data were available from HYDAT gauges (Appendix A). Not all hydrology stations have complete records over the 1986-2012 period of interest. To fill missing data we used two approaches. For short time periods (periods with steady flows or steadily changing flows, no evidence of peaks or dips at other stations) we used simple linear interpolation. For longer periods of missing data with evidence of changing flows, peaks or dips (at other stations), we consulted with the UTRCA hydrologist and constructed a non-parametric correlation matrix (Kendall's tau) between all hydrology stations. Using this information we chose a surrogate station to fill in data as explained in Appendix B.

Hydrology stations were assigned to chemical stations and flows were adjusted by watershed area, if locations did not overlap (Appendix B, Table 3). An additional hydrological station was constructed to represent the mouth of the Thames River where it enters Lake St. Clair, using Thamesville flow (E003) and additional flow of the whole watershed area below Thamesville that is prorated from McGregor Creek (E007).

The following water quality variables indicating nutrients and sediments were used in this study (units: mg/L).

Nutrients:

- Total phosphorus (TP),
- Dissolved reactive phosphorus (DRP),
- Sum of nitrate and nitrite (NO_3+NO_2 or NO32),
- Total nitrogen (TN) is the sum of NO_3+NO_2 and Kjeldahl-N

Sediments: total suspended solids (TSS), called “particulate residue” in the original data sets.

Sources for the water chemistry data and their locations are listed in Appendix A.

Some outliers were removed before analysis as specified in Appendix B.

City of London TP concentrations had a tendency for higher TP values compared to those monitored in the PWQMN program (Appendix C). After detailed analyses, we decided to combine the data from the two monitoring programs for the following reasons:

(1) Follow-up comparison showed no difference (UTRCA results 2014)

(2) There are many stations that are only monitored by one of the two agencies and elimination would disregard a lot of information

(3) In the following stations that were sampled by both agencies, a large of amount of data would be disregarded if only one source would be used

There were three such simultaneously monitored stations that were merged, PWQMN ID (LoC ID):

- WQ-Stn 27 (Highbury-Clarke) on NTR, Flow-Stn D003
- WQ-Stn 47 (Komoka) on NTR, no flow is associated, so no loads were computed
- WQ-Stn 96 (Stoney Creek) on Stoney Creek, NTR, Flow-Stn D028

2.3 Hydrology and water quality concentration

To examine temporal as well as spatial trends in water quality, it is necessary to have a good estimate of water chemistry over the period of interest. However, most water chemistry sampling regimes can only manage at most eight monthly samples per year. And, as shown next, riverine water chemistry can vary widely with flow rates, time and location. Consequently, we use several models to translate the few samples available per year and station into flow-weighted average concentrations (FWCs) and loads that can be used for comparative purposes over the periods of interest.

Because flow relationships can differ between the studied water quality variables, all modeling is done separately for each variable and each station. If there are only a small number of data and years available, meaningful modelling cannot be achieved (Section 2.3.3) and monitoring data have to be interpreted with extreme caution. In addition, some smaller water chemistry stations have no flow data available (e.g., see Section 5.1.1.1 for special monitoring stations) so the computation of FWCs was not possible and only temporal averages were used.

The large temporal (up to 8 times per year for 24 years within 1986-2012) and spatial (83 stations) sampling of WQ combined with the extensive coverage by daily flows from 26 gauges makes it possible to describe and assess the variation of nutrients and sediments throughout the Thames River watershed.

2.3.1 Comparison of monitored concentrations and flows: example

We present the NTR station at 285 km above the inflow into St. Clair as an example. Simple plotting of daily flow versus monitored TP concentration reveals that extreme TP concentrations occur at both low and high flows (Figure 2, Figure 3). Further, there are seasonal patterns that are discernable in this overview graph and are investigated in Section 4.2. There is also a long-term trend that becomes apparent after filling in missing data by modeling to obtain flow-weighted concentrations and is explored in Section 4.1. A direct comparison of monitored variables against flow is presented in scatter graphs (Figure 3; figures for all stations are available in a separate file).

Figure 2. Comparison of Flow at station DO14 with TP concentration at WQ44, both at 285km of the NTR near Mitchell

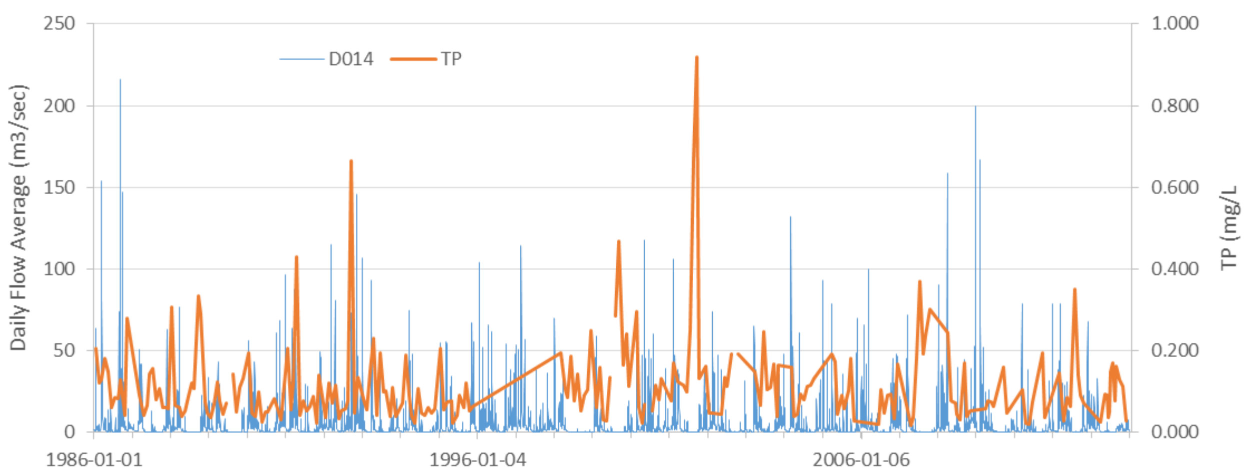
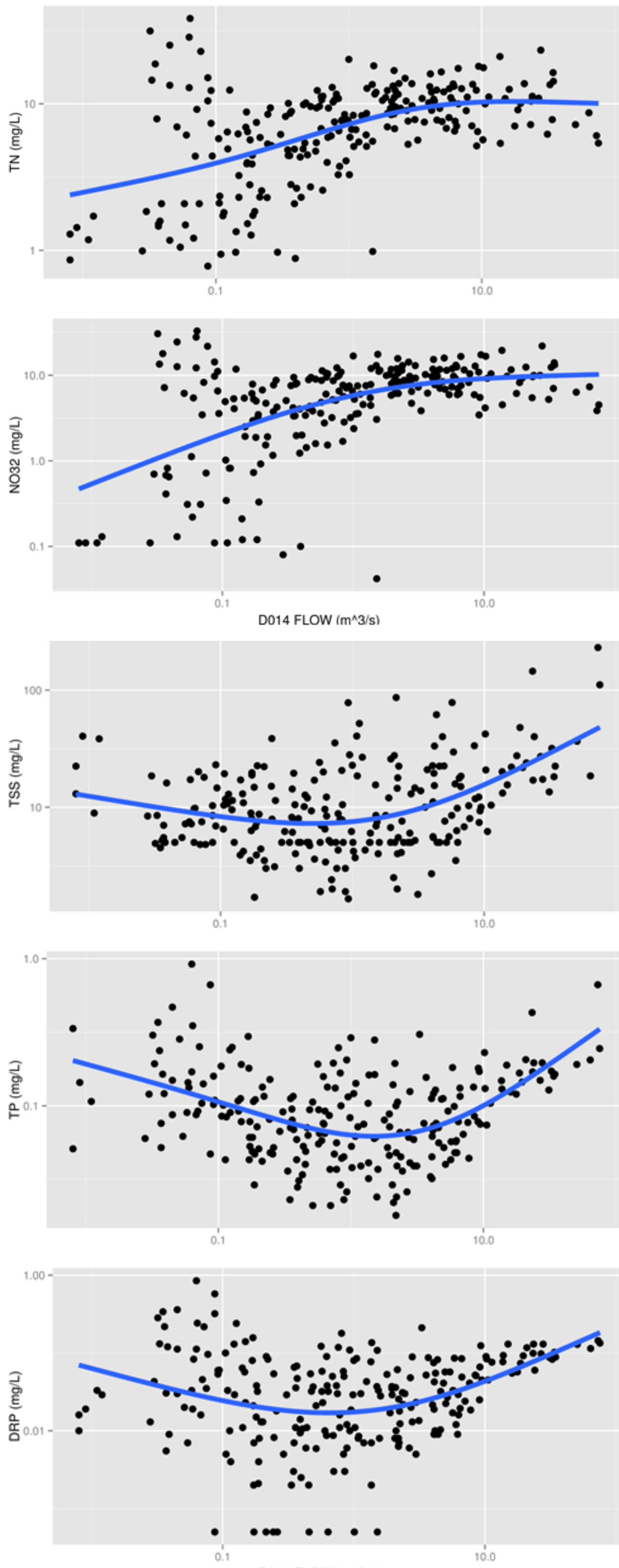


Figure 3. Water quality monitoring data (WQ44) compared to flow (D014) and smoothing curve



2.3.2 Combining flow with water quality concentration: Models

Detailed methods are described in Appendix B.

Flow-weighted average concentrations are chemical loads divided by the total flow over the period of interest. The method of load calculation was based primarily on data availability. As high quality hydrology data were usually available in daily increments, we chose daily increments for load calculations. Daily loads were calculated as the product of total daily flow and an estimated daily average chemical concentration at the same (or neighboring) location. The main error associated with this calculation is the estimate of daily average chemical concentration, as discussed below.

We used three different models to calculate daily chemical concentration and daily load depending on data availability. Both EGRET and GAM include a relationship between flow and concentration in the model. LINEAR does not.

1. EGRET: “EGRET” (Exploration and Graphics for RivEr Trends) is based on the USGS program of “Weighted Regressions on Time, Discharge and Season" (WRTDS) as implemented in R in the EGRET library (Hirsch et al., 2010; Sprague et al., 2011). WRTDS makes no assumptions about statistical distributions and provides no statistical confidence limits. It also makes no assumptions about the shape of relationship between logarithmically transformed water quality and flow variables, and explicitly includes long-term and seasonal changes in that relationship. Although the developers recommend a minimum of 20 years of data and at least 200 individual samples, we were able to run EGRET on stations with at least 7 years and a 100 sample points. About twenty stations had insufficient data for this model.

2. GAM: This General Additive Model is an optimally weighted regression with smoothing after log-log transformations using the default non-isotropic tensor product splines. We included the date and daily flow in the main smoothing function. Smoothed sinusoidal seasonal terms were added as well. All stations had sufficient data for this model.

3. LINEAR: As in the earlier Upper Thames Studies (Nürnberg and LaZerte, 2006, 2005), daily chemical concentrations obtained by linear interpolations between sampling dates were combined with daily flows to obtain daily loads. All stations had sufficient data for this model.

Results were computed for all three models wherever possible. In this way, comparisons could be made between stations, even if some did not have sufficient data for the data intensive model.

The load estimates differ between models. On average, EGRET estimated loads and FWCs are higher than results by GAM or the simpler model of non-flow-based LINEAR loads and concentrations, especially for TP, DRP and TSS. GAM is also mostly higher than LINEAR. In general, the differences between estimation methods are smaller when the water quality variables have little relationships with flow. But the models that use flow-based estimates of concentration (EGRET and GAM) are higher on average than the non-flow-based LINEAR model, probably because the non-flow-based linear interpolation method misses too many peak flows with higher concentrations. These differences between model results have to be taken into consideration when comparing loads and FWCs and interpreting results.

2.3.3 Challenges: Infrequent sampling and missing high flow events during monitoring

High flows have the most impact on annual loading rates, especially where water quality variables are highly correlated with flow rates. In low frequency monitoring efforts such high flow conditions are often missed so that models must extrapolate beyond the calibration data set. The number of days when flows were higher than those for which monitoring data are available was computed to obtain a relative estimate of how much extrapolation was required (Table 2). The highest number of outlying days occurs at the assembled TR mouth station, which is to be expected, since the flow of McGregor Creek was added to the flow of the WQ monitoring station upstream (Appendix B). The next frequent occurrence of 10 per year on average occurs in the TR at 127 km, station 308302, Currie Rd. The loads and FWCs at these stations may be less reliable than loads at most other stations, where such occurrences did not happen as frequently.

Table 2. Number of days when flows exceeded the flows when water quality was monitored at the stations indicated by km

Thames River Branch	WQ Station	Average per year	Max # days per single year	Sum of all years	Number of yrs (n)
TR	Mouth (0km)	14.1	37	197	14
TR	McGregor Cr (29.7km)	5.4	11	38	7
TR	Kent Bridge (49.7km)	4.5	16	76	17
TR	Currie Rd (127.2km)	10.3	24	72	7
TR	Dingman Cr (186.5km)	0.1	1	2	25
TR	Oxbow Cr (194.2km)	4.4	11	44	10
TR	Byron (202.2km)	0.3	3	9	27
NTR	Medway Cr (214.1km)	5.8	14	144	25
NTR	Stoney Cr (216.5km)	7.7	16	77	10
NTR	Clarke (223.1km)	0.7	5	18	27
NTR	Thorndale (232.8km)	0.5	2	9	19
NTR	Fish Cr (248.6km)	1.8	4	11	6
NTR	St. Marys (255.9km)	6.3	10	69	11
NTR	Trout Cr ds (256.5)	1.6	10	40	25
NTR	Trout Cr us (256.5)	1.3	4	31	24
NTR	Avon R (265.9km)	0.5	3	15	32
NTR	Mitchell (285km)	1.2	5	29	25
STR	Adelaide (213.3km)	2.7	8	24	9
STR	Waubuno Cr (222.5km)	0.9	2	7	8
STR	Middle Thames (240.3km)	0.3	2	8	25
STR	Reynolds Cr (241.6km)	6.4	14	32	5
STR	STR-Ingersoll (247km)	0.5	3	10	20
STR	Cedar Cr (267.4km)	1.2	4	12	10
STR	Woodstock (267.6km)	0.4	5	8	20
STR	Woodstock Historic (270km)	1.3	7	20	15
STR	Innerkip (282.6km)	0.3	2	6	20

The importance of sampling high flow events is illustrated by the generally high concentrations measured after the major rain event in 29 May 2013 (Table 3).

In 156 sampling events since 1986 at Kent Bridge only one had a TSS above 420 mg/L (694 mg/L, 27 May 1991), while aimed sampling at a major rain event on 29 May 2013 produced a high value of 660 mg/L. Similarly, in 200 sampling events since 1986 the second largest TSS in Trout Creek (WQ66) was 282 mg/L on 26 July 2005 while it was 290 mg/L at the major rain event on 29 May 2013. Also, the TP concentration was extremely high at the major rain event on 29 May 2013 at several stations (Table 3).

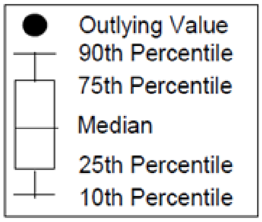
Table 3. Water characteristics for a major rain event on 29 May 2013

Location km	Site (WQ Stn)	TP mg/L	DRP mg/L	TSS mg/L
49.69	Thames Kent Bridge (305802)	0.88	0.045	660
89.81	Fleming (310902)	0.54	0.180	170
115.16	Newbiggen (307302)	0.57	0.270	97
127.19	Thames @ Dutton	0.60	0.081	330
184.00	Thames @ Delaware	0.48	0.048	160
232.81	Thorndale (50)	0.19	ND	75
256.51	Trout Ck (upstream Wildwood) (66)	0.69	0.130	290
265.94	Avon (25)	0.18	0.034	38
267.96	Glengowan	0.03	ND	ND
322.54	Waubuno (97)	0.73	0.110	260
340.27	Middle Thames (41)	0.70	0.160	220
347.04	Thames Ingersoll (42)	0.40	0.068	120
367.45	Cedar (17)	0.25	0.051	78
367.58	Thames Woodstock (16)	0.21	0.058	47

2.4 Statistical analysis

Computations and data and graphical analyses related to loads and FWC were done using R version 3.1.0 (2014, The R Foundation for Statistical Computing) with additional CRAN (Comprehensive R Archive Network) packages (Appendix B).

In addition, statistical analysis, using Systat version 13, was used to decide whether a pattern was likely "real" or due to chance alone. Such analysis can only indicate potential trends but cannot determine with certainty whether any trends are missing. In particular, when there are different sample sizes for the different models and variables the direct comparison of significance levels is not valid. 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. Multiple regression analysis was used to determine temporal (Year) and spatial (KM) trends along the river. To compare whether results were significantly different between certain characteristics, such as stations or computation methods, paired t-tests were used.



Variables were also compared using box and whisker plots. The upper and lower horizontal line of the box, the horizontal line within the box, and the error bars represent the 25th and 75th percentiles, the median, and the 10th and 90th percentiles, respectively of the presented data distribution. The circles or stars represent outliers.

2.5 GIS based information

All GIS based information was provided by UTRCA. This information includes watershed areas, distance from the mouth expressed as river-km, and land use related information. In some cases overall areas are slightly different depending on the data source.

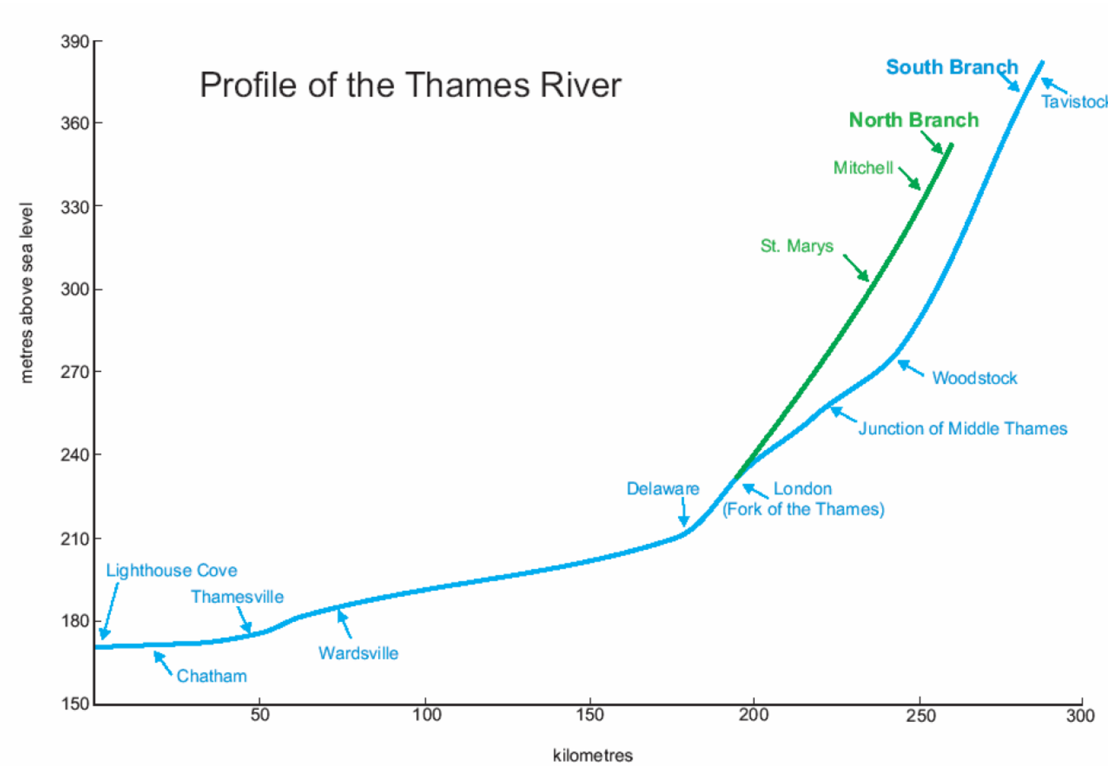
Source: UTRCA data (urban boundary delineation), and Ontario Ministry of Agriculture and Food's Constructed Drains and Agriculture Resource Inventory, 2010 data layers.

3 General influences on Thames River water quality

3.1 Basin morphometry and land use

River elevation changes about 210 m and extends from about 175 m asl at St. Clair to 385 m asl at the STR branch in Tavistock (Figure 4). While the NTR and STR experience relatively drastic but similar elevation changes, the elevations along the TR below the Forks and Delaware decrease towards the mouth where back flow and exchange with Lake St. Clair occur frequently.

Figure 4. Elevation changes along the Thames River (UTRCA 2008)



Throughout the entire watershed of 5,692 km^2, the prevailing land use is agriculture (80% of total watershed area, Table 4, Figure 5). The next frequent land use is urban (7.8%), treed by deciduous trees (5.1%) and wetlands (4.6%). While these proportions vary slightly between sub-watersheds, the combined agricultural and urban area is close to or above 80% in all explored cases (except the tributary to the STR, Dorchester Swamp Creek, which includes 30% wetland in its watershed area of 18 km^2). Consequently, we do not expect major spatial differences in river water quality due to different land usage within the water basin.

These vast anthropological influences (agriculture + urban = 87% TR, 89% NTR, 85% STR) in combination with naturally fertile soils characteristic of the St. Lawrence basin (Chambers et al., 2001), are the most important causes for the high nutrient and sediment status of the Thames River water.

As agriculture makes up 80% of the land use throughout the basin, any changes in farming practice within the region can have a large effect. On average 59% of the agricultural area is tile-drained of which two thirds are systematically installed drains and one third is random.

There was evidence that increasing crop and livestock densities were related to increased TP and nitrate in a study on 15 streams including 5 in the Thames River watershed (Ministry of Environment, 2012). If the “Agriculture” land use category were broken into smaller units reflecting different crop and livestock practices, it might be a more informative spatial predictor of river water quality.

The “Dams” land use category shows similar levels of impoundment in both the NTR and STR. The general benefit of impoundments (locations in Appendix D) in accumulating TP and TSS and thereby reducing downstream nutrient levels is apparent by comparing upstream with downstream water quality characteristics and loads, although there is some seasonal variation (Section 3.5). Even smaller ponds decrease TP and TSS in the river, but retain it in the ponds themselves, e.g., in Southside Pond on Cedar Creek, tributary to the STR at 267km (UTRCA 2010 and unpublished data). A more detailed analysis of several of the larger impoundments in the Thames River watershed is presented in Section 3.5, Appendix E and Nürnberg and LaZerte (2005, 2006).

Table 4. Land use along the Thames River, percentages for the drainage area above each listed station

Name	ID	km	Main	Trib	**Area**	AG	Urban	Treed	Swamp	AG+Urb	Dams	Tile
					(km^2)			(% of total area)				%of AG
Thames River												
Total watershed	Mouth	0.00	x		**5,692**	80%	7.8%	5.1%	4.6%	87%	0.24%	59%
Jeannettes Creek	311002	3.47		x	**330**	92%	5.0%	0.5%	1.1%	97%	0.00%	68%
Thames River	308202	14.83	x		**5,005**	78%	8.2%	5.8%	5.1%	86%	0.28%	58%
McGregor Creek	308102	29.74		x	**203**	89%	5.9%	1.4%	1.9%	95%	0.00%	71%
Thames River	305802	49.69	x		**4,569**	77%	8.2%	6.2%	5.4%	85%	0.30%	58%
White Ash Creek	305702	64.96		x	**76**	85%	3.3%	4.4%	5.3%	88%	0.00%	68%
Fleming Creek	310902	89.81		x	**113**	83%	3.2%	4.8%	6.1%	86%	0.03%	59%
Newbiggin Creek	307302	115.16		x	**46**	87%	5.9%	3.4%	2.7%	93%	0.00%	63%
Thames River	308302	127.19	x		**3,845**	77%	9.1%	6.1%	5.1%	86%	0.36%	59%
Komoka Creek	63	185.70		x	**18**	64%	10.7%	5.4%	14.3%	75%	0.00%	0%
Lambeth	Dingman Cr	186.45		x	**134**	64%	23.9%	6.3%	3.9%	88%	0.01%	44%
Byron	Byron	202.23	x		**3,106**	78%	8.9%	5.3%	4.8%	87%	0.42%	62%
North Thames River												
Dundas	Dundas	209.81	x		**1,714**	81%	7.1%	4.7%	4.2%	89%	0.40%	73%
Medway	Medway	214.13		x	**205**	81%	7.8%	6.6%	2.9%	89%	0.08%	79%
27_Clarke	Clarke	223.06	x		**1,433**	84%	4.9%	4.3%	4.4%	89%	0.47%	72%
North Thames River	50	232.81	x		**1,352**	84%	4.8%	4.1%	4.5%	89%	0.32%	73%
Gregory Creek	95	241.93		x	**60**	89%	3.1%	5.1%	1.8%	92%	0.00%	69%
Fish Creek	90	248.58		x	**153**	90%	2.4%	3.0%	3.1%	92%	0.00%	71%
Nineteen Creek	310002	248.58		x	**27**	92%	1.8%	3.7%	2.6%	93%	0.00%	82%
North Thames River	45	254.79	x		**1,073**	83%	5.3%	4.0%	5.0%	89%	0.40%	73%
Trout Creek	64	256.51		x	**141**	75%	2.9%	7.4%	6.6%	78%	2.55%	61%
Otter Creek	94	262.38		x	**59**	86%	2.7%	4.3%	5.0%	88%	0.06%	83%
Flat Creek	89	263.13		x	**90**	88%	2.2%	4.4%	4.7%	90%	0.02%	74%
Avon River	25	265.94		x	**114**	69%	17.3%	3.2%	8.8%	86%	0.32%	80%
Black Creek	92	274.18		x	**139**	82%	3.2%	2.9%	10.2%	85%	0.00%	63%
North Thames River	44	284.95	x		**318**	90%	3.8%	2.9%	2.4%	94%	0.04%	68%
Whirl Ck	Whirl	286.80		x	**128**	90%	2.7%	3.7%	2.9%	93%	0.00%	71%
South Thames River												
York	York	210.23	x		**1,364**	76%	9.6%	6.0%	5.7%	85%	0.43%	47%
Adelaide	Adelaide	213.33	x		**1,353**	76%	9.0%	6.0%	5.7%	85%	0.42%	47%
Waubuno Creek	97	222.54		x	**96**	84%	3.5%	7.1%	3.8%	87%	0.00%	71%
Dorchester Swamp Creek	52	228.51		x	**18**	49%	12.7%	5.3%	30.1%	62%	0.02%	20%
Middle Thames River	41	240.27		x	**311**	82%	3.6%	7.4%	5.2%	85%	0.11%	60%
Reynolds Creek	91	241.59		x	**148**	85%	3.3%	5.5%	4.4%	88%	0.01%	39%
South Thames River	42	247.04	x		**551**	77%	9.2%	5.2%	5.5%	86%	0.73%	40%
Cedar Creek	17	267.45		x	**96**	71%	15.0%	3.7%	7.6%	86%	0.20%	40%
South Thames River	16	267.58	x		**272**	79%	7.0%	4.8%	5.7%	86%	1.20%	44%
South Thames River	80	282.62	x		**151**	85%	4.4%	4.9%	4.1%	90%	0.01%	53%
South Thames River	310202	282.65	x		**29**	84%	2.1%	5.5%	7.6%	86%	0.00%	87%

Main stem stations are shaded.
AG, agriculture; Tile, proportion of agricultural area that is tiled.
Swamp, wetlands w/o larger lakes and impoundments
Dams, impounded area
No entry: missing data, Source: UTRCA, from *Ontario Land Cover Compilation*

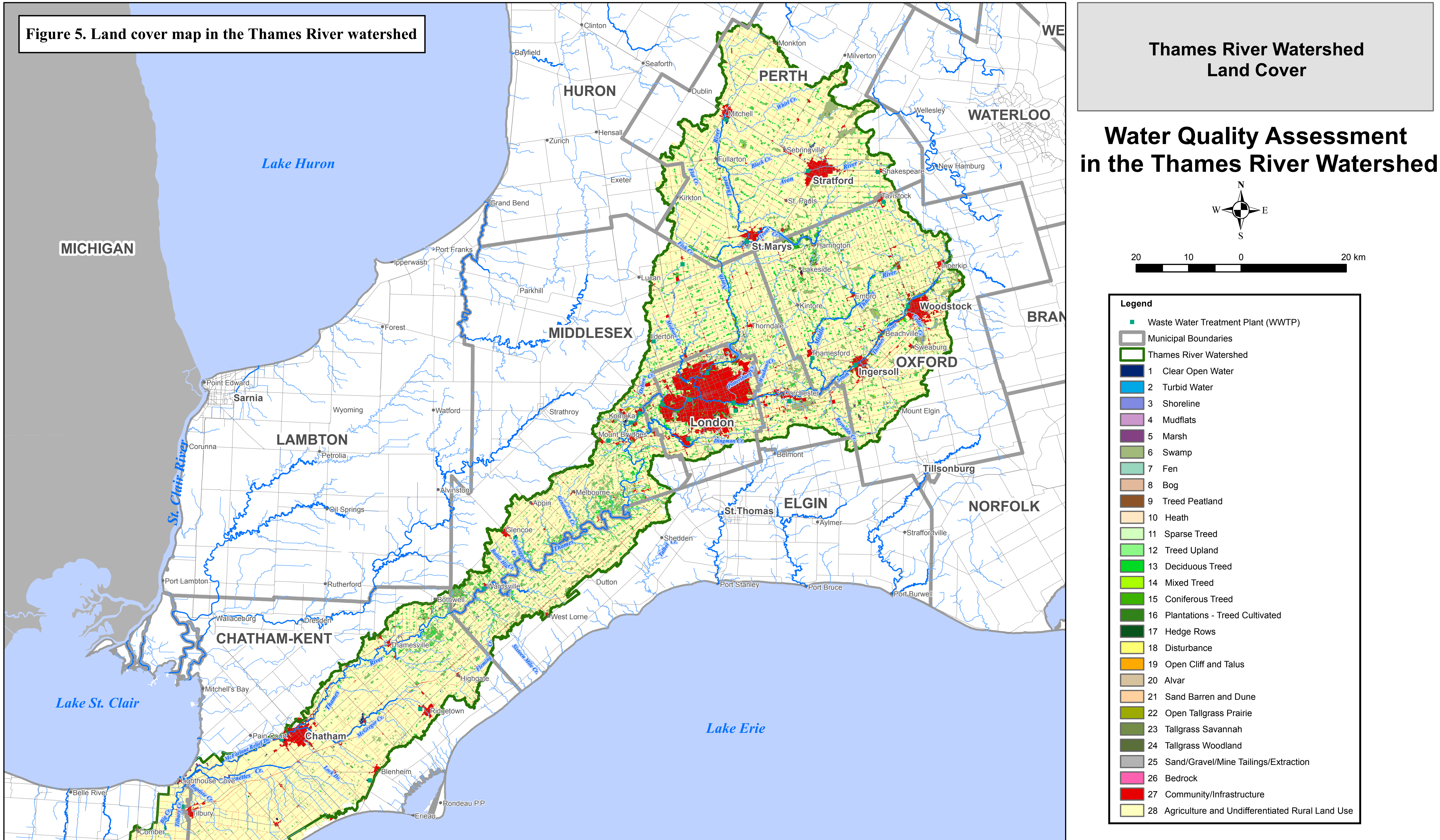


Figure 5. Land cover map in the Thames River watershed

Map created by UTRCA, October, 2014
Base mapping provided by Land Information Ontario.

From OFAT III - The Ontario Land Cover Compilation, consists of 30 land cover classes derived by combining the Provincial Land Cover Database (2000 Edition), Far North Land Cover Version 1.3, and the Southern Ontario Land Resource

3.2 Flows

Flows determine loads (i.e., the product of flows and concentration) to a large extent, and when analysing loads, temporal and spatial patterns in hydrology are important and must be recognised.

To this end, multiple regression analysis (with variable “Year” for temporal and variable “KM” controlling for spatial trend) was conducted to determine whether there are any temporal trends that would have to be considered. Analysis was executed along the three parts of the Thames River and its tributaries separately for annual and seasonal average flows.

There was no long term significant trend with Year over the period of interest (1986-2012) for stations on the main stem, as well as on included stations of tributaries. This implies that any temporal trends in loads and flow-weighted concentration of the studied water quality variables would not be due to temporal trends in flow. It also means that there is no evidence that annual and seasonal average flows are affected by climate change (yet), despite predictions of more frequent and severe storm events (Section 6).

To obtain insight into seasonal variation, daily flow averages were summed by month (Figure 6). The seasonal variation of flows in the Thames River watershed is large as observed in other South Ontario streams (Ministry of Environment, 2012) and the Grand River (Loomer and Cooke, 2011). There are two distinctly different periods with respect to flow: a high-flow spring period (Mar-Apr) followed by a decreasing flow period in the summer (May-Sep); water quality for these periods was inspected in more detail throughout this study. Remaining months, Nov-Feb, were not specifically investigated mainly because the water quality variables were rarely measured during this period so that uncertainty would be higher than for the other periods. In addition, this period can be considered less important with respect to recreational water usage and cyanobacteria proliferation.

The seasonal pattern was slightly different for the three branches probably reflecting their different water management and reservoir operations.

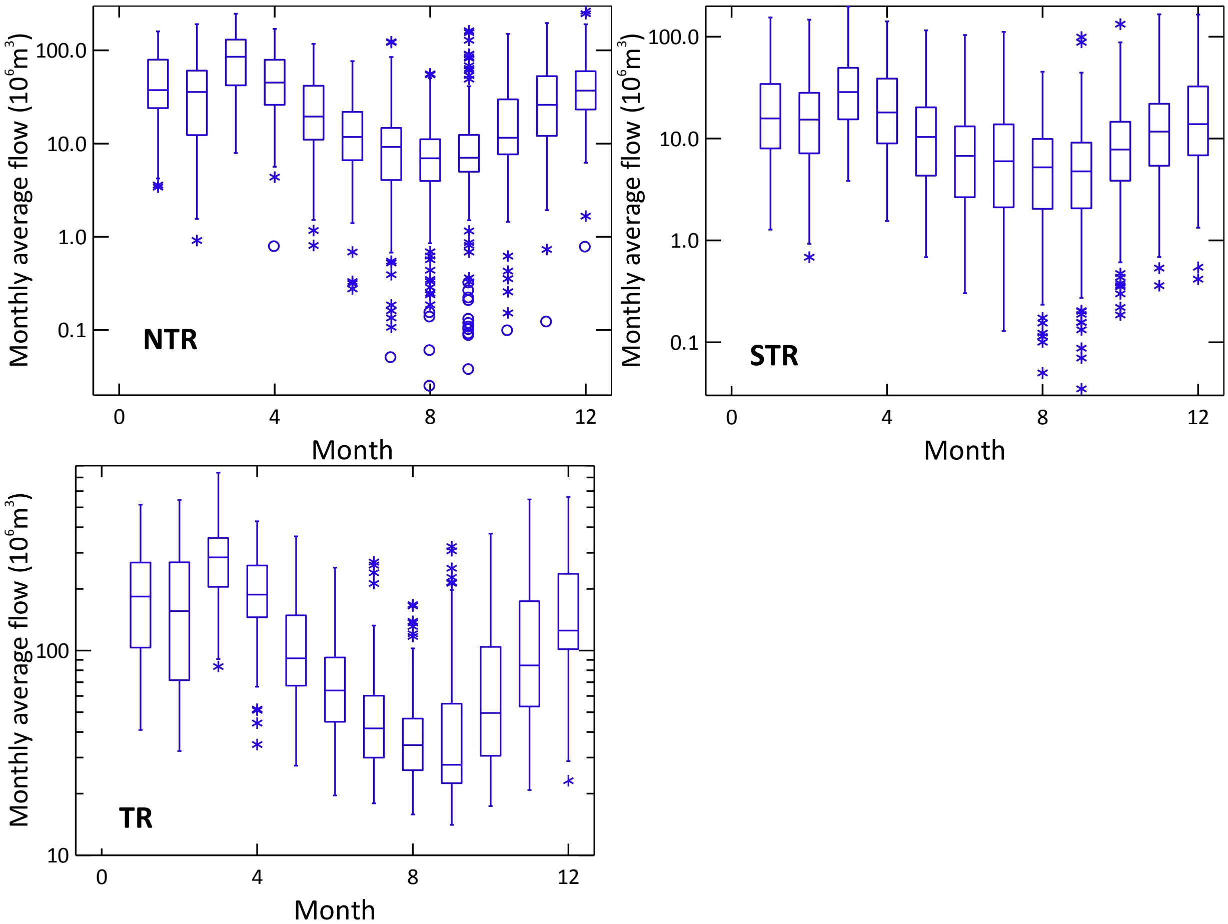
NTR: Mar-Aug downward trend, Aug-Sep almost constant at low flow period. There are several reservoirs including Wildwood Reservoir on Trout Creek and Fanshawe Reservoir on the main stem.

STR: Mar-Sep downward trend, Jun-Sep slight decrease, but almost constant during low flow period. This section includes the Pittock Reservoir.

TR: Mar-Sep downward trend, very pronounced. No explicit reservoir, but regulation by weirs throughout and large tributaries

Both upstream branches contribute about the same proportion of the flow to the TR (Table 1).

Figure 6. Monthly average flows in main stem flow stations of the three river branches (1986-2012)



3.3 Internal loading

Although this study is primarily concerned about loads along the river that originate in the watershed, phosphorus can also originate within a water body from previously settled bottom sediments, senescent water plants and disturbance of nutrient rich sediments.

Along the Thames River internal loading can contribute to phosphorus loading most obviously in three situations (1) in impoundments at the upper Thames, (2) in slow moving large river sections in the lower Thames and impoundments created by pumping and (3) in fine sediments contributed from upstream locations. Marshland and natural wetlands can contribute nutrients if they become hypoxic, although they retain phosphorus if less enriched (Mitsch and Day, 2006). Macrophytes have the potential to modify the retention of dissolved nutrients during the summer low flow and can increase deposition and stabilization of particulate forms in the sediments (Jones et al., 2012). The role of fine sediments in the eutrophication of anthropogenic affected rivers is often overlooked (Jones et al., 2012).

The most important type of internal loading to be anticipated in the Thames River is the phosphorus that is released from anoxic bottom sediments (Nürnberg, 2009). 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 released as phosphate, in a form that is close to 90% biologically available, while the biologically available fraction of the external load is usually less than 50% available, except for point sources. This high DRP can quickly be assimilated by phytoplankton biomass if other conditions (light, temperature, micro-nutrients) are favorable.

Evidence and size of internal load in the Thames River reservoirs was investigated in previous studies (Nürnberg and LaZerte, 2006, 2005). Because it is influenced by temperature, largest internal loading rates occur in the summer, but is possible under ice as well. Internal load is further described in this report in Section 3.5, *Reservoirs,* and its effect on the river is explained throughout the detailed water quality analysis in Section 5 by comparing upstream with downstream monitoring results. Supporting observations for the lower Thames waters are presented in Section 5.3.3. Evidence of internal load as P release from anoxic sediments was also described in reservoirs along the Grand River (Grand River Water Management Plan, 2013; Loomer and Cooke, 2011).

3.4 Waste Water Treatment Plants

Each of the 26 WWTPs analyzed TP and TSS, but only 11 plants analysed for NO32 and some for Kjeldahl-N so that TN was only available for 10 plants. DRP data were not consistently available but based on other effluent characterization studies it can be assumed that about 30 to 50% of TP is DRP. (Scott Aberneth, MOECC, pers. comm. Mar. 4. 2015: "For secondary treatment and an effluent limit of 1 mg/L the ratio or percentage is about 30 to 50% because there is minimal addition of treatment chemical which precipitates the soluble DRP fraction." Effluent reports were available for 2000-2012 with varying completeness for the different WWTPs. This means that in comparing long-term contributions along the river, often the more recent effluent conditions (2000-2012) are compared to longer term (1986-2012) river concentrations. Consequently the contributions from WWTPs may be underestimated, considering that effluents were probably more enriched before the period of data availability.

Location and average (mostly for 2000-2012) effluent concentrations and loads are listed in Table 5. Characteristics vary with size of the plants, service volume, technical standard and period of reporting. In general, nutrient concentrations are higher than in the Thames River, while TSS concentration is lower (Figure 7; for comparison, long-term average flow-weighted concentration ranged approximately 0.1-0.2 mg/L TP, 7-10 mg/L TN, 20-40 mg/L TSS along the Thames River, Section 4). Elevated TP and low TSS concentrations were also noted in WWTPs along the Grand River (Loomer and Cooke, 2011).

Table 5. Characteristics of effluents from WWTP throughout the Thames River watershed.

Facility	km[1]	Average Concentration (mg/L)				Yr[2]	Average Load (kg/yr)			
		TP	TN	NO32	TSS	n	TP_L	TN_L	NO32_L	TSS_L
					Thames River					
Tilbury	1.29	0.528	10.4	6.8	7.2	9	523	10,720	7,033	7,363
Merlin PV Lagoon	3.48*	0.412	4.3	1.7	10.8	0	------------	Not enough data	---------	
Chatham	24.95	0.453	16.7	14.8	6.2	12	3,751	146,173	130,160	49,887
Blenheim Lagoon	29.74*	0.279	6.9	4.0	2.2	7	241	6,426	3,730	1,904
Ridgetown	29.74*	0.248	10.7	13.9	5.3	9	206	6,637	21,747	4,782
Thamesville	65.16	0.489	12.2	9.9	7.2	13	45	1,109	899	663
Wardsville	93.45	0.223			3.3	5	8			125
Glencoe	115.16*	0.267		9.5	4.8	1	59		2,097	1,068
Mount Brydges	185.03	0.255			2.5	1	6			58
Southland Park, Dingman Cr	186.45*	0.296			6.9	13	24			555
Komoka	189.08	0.140			2.8	4	33			649
Kilworth Heights	192.12	0.117			1.7	4	26			387
Ilderton	194.21*	0.198	13.2	11.2	4.1	8	40	2,020	1,716	807
Oxford	200.89	0.438			3.9	13	1,241			10,527
Greenway	207.50	0.435			6.4	13	22,371			320,472
w/o 2000	207.50	0.409			6.2	12	19,120			290,046
					North Thames River					
Granton	214.13*	0.476			15.9	8	19			659
Adelaide	217.55	0.484			5.3	13	4,603			50,826
St.Marys	256.51	0.231	5.4	4.6	9.6	5	336	7,766	6,004	13,610
Stratford	265.94*	0.143	17.0	13.8	3.6	5	951	107,362	94,552	23,660
Mitchell	285.16	0.341			5.0	13	508			7,349
					South Thames River					
Vauxhall	214.8	0.420			6.6	13	2,892			46,885
Pottersburg	217.70	0.499			6.0	13	4,830			58,074
Dorchester	228.51	0.377			3.0	10	26			208
Thamesford	240.27*	0.213			3.6	11	95			1,668
Ingersoll (old and new)	251.30	0.407	20.9	17.7	8.6	15	577	27,927	23,775	12,172
Woodstock	269.30	0.389	19.2	17.6	6.4	9	3,201	150,858	138,217	52,747
Mount Elgin	241.59	--------------------		Subsurface and infiltration, no data available					--------------	
Tavistock	282.64*	0.086	4.7	1.8	6.9	4	53	2,712	970	4,115
Shakespeare	282.65*	0.189			3.2	2	9			145

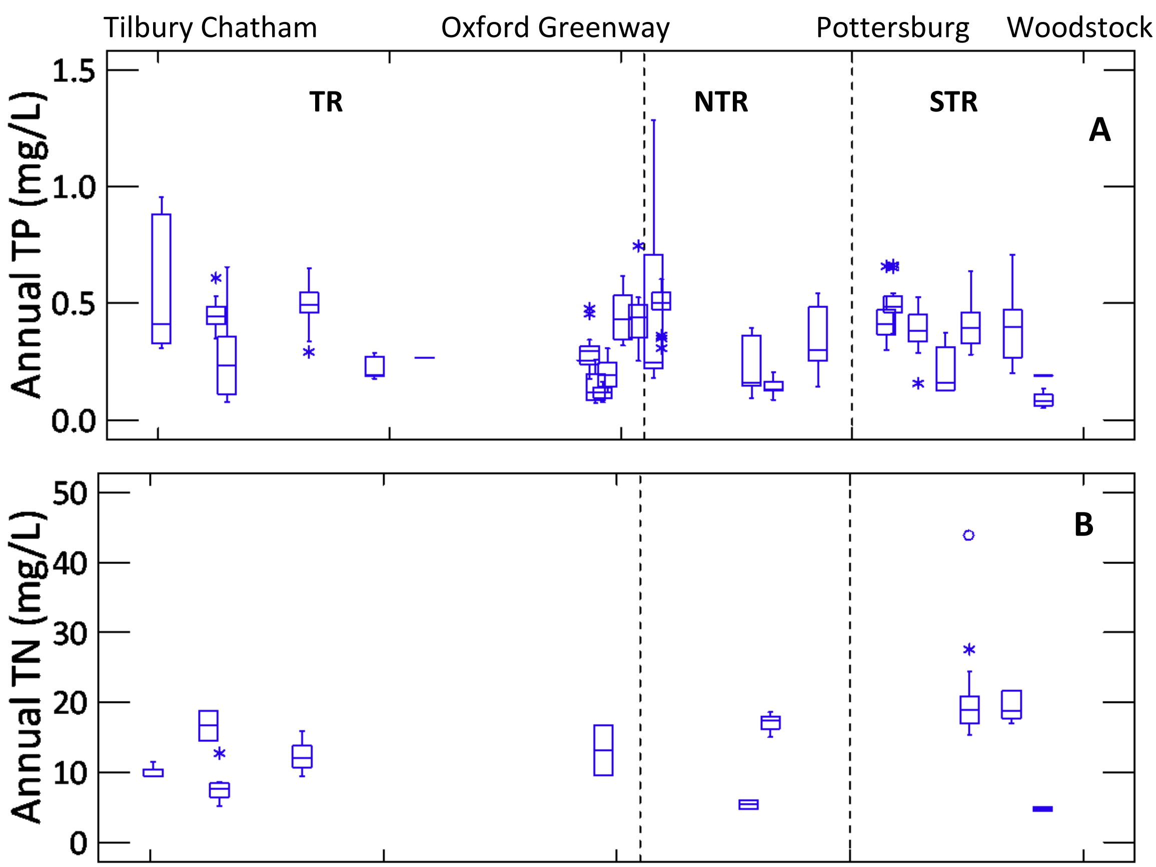
[1]on tributary*; [2] values are reported for the number of years (n) with at least 7 months of TP and TSS data since 2000, less data were available for nitrate and TN. 0, only individual months for different years available.

Differing quantity and unknown influences contribute to temporal variability in TP export between years (Figure 8, for all WWTPs) and months (Figure 9, for the largest WWTPs in each branch). Because of temporal gradients (e.g., decrease in TP effluent load, Figure 8 and Figure 9) summary statists are not comparable if not available for the same period (Table 5).

Because of the low river flow in summer months, water quality impacts of WWTPs are most pronounced at that time, which is also described in the Grand River (Grand River Water Management Plan, 2013) and British watersheds (Jarvie et al., 2006). The influence of individual WWTPs effluents on the surrounding river is discussed in Section 5.

Figure 7. Annual average TP (A), TN (B) and TSS (C) concentration in WWTP effluent along the Thames River main stem and tributaries (within 2000-2012)

Broken vertical lines indicate the start of the different branches, names depict location of some larger plants



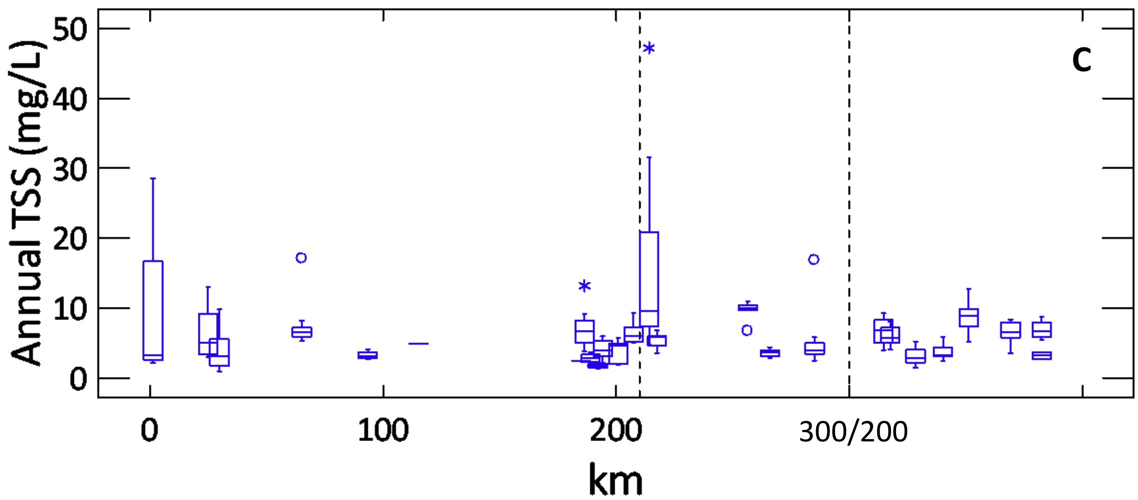
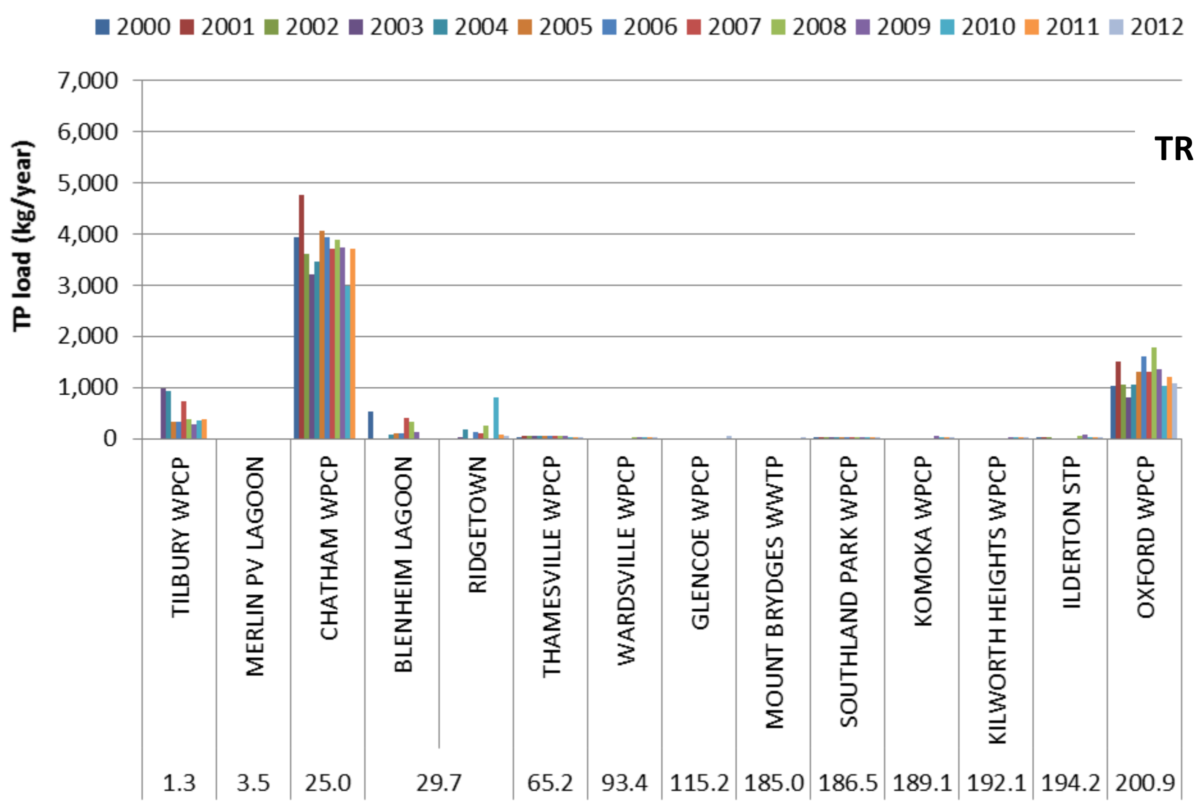
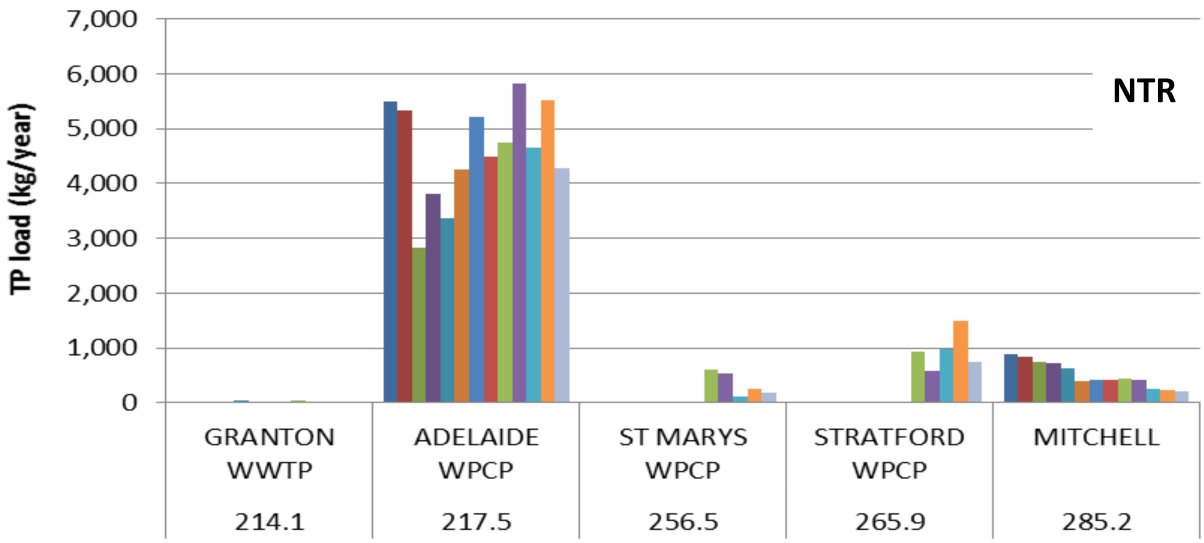


Figure 8. Waste water treatment plant annual TP loads along the Thames River main stem and tributaries

Loads are variable with time but tend to decrease in the larger WWTPs of the TR and STR.
WWTP name, location of plant on the main stem or where its tributary joins the main stem, in km upstream of the mouth, and year of record. The large CoL Greenway WWTP at 207 KM is shown separately in Figure 9.




TP load (kg/year)
7,000
6,000
5,000
4,000
3,000
2,000
1,000
0
NTR
GRANTON WWTP
214.1
ADELAIDE WPCP
217.5
ST MARYS WPCP
256.5
STRATFORD WPCP
265.9
MITCHELL
285.2

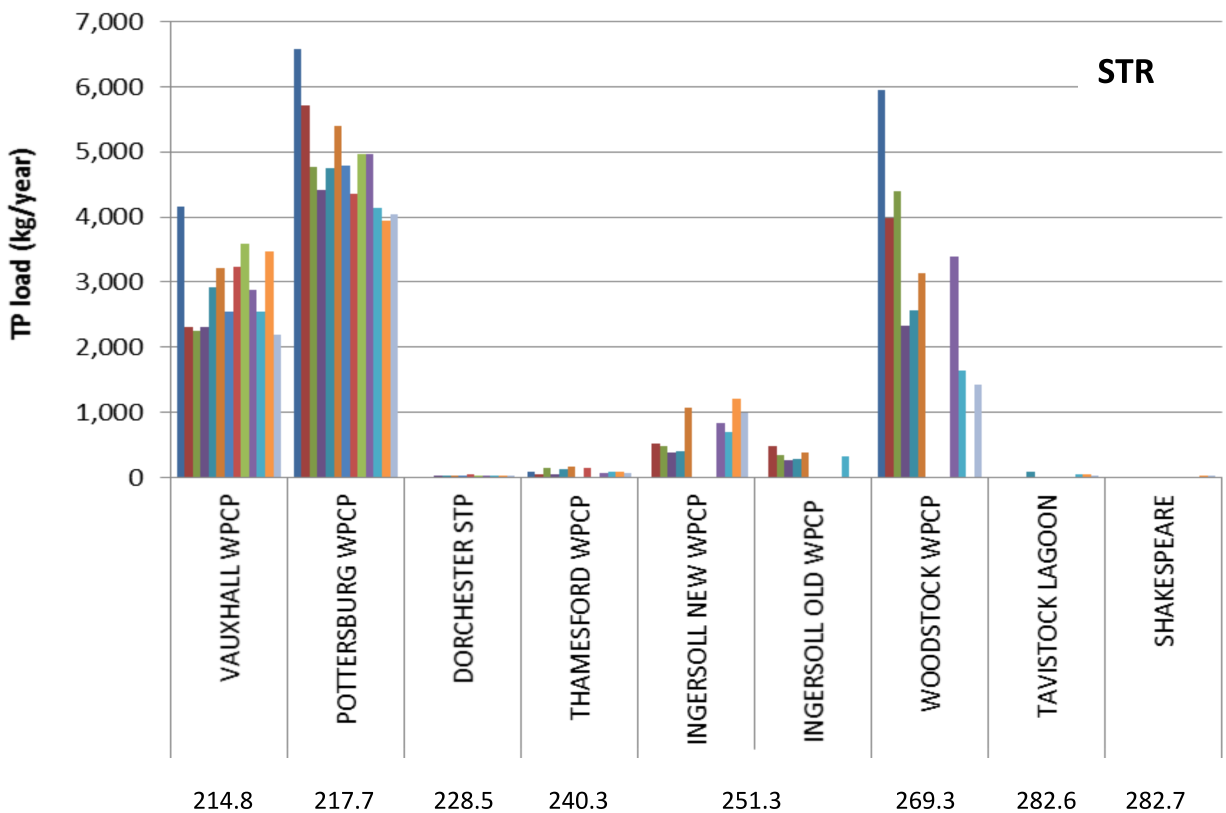
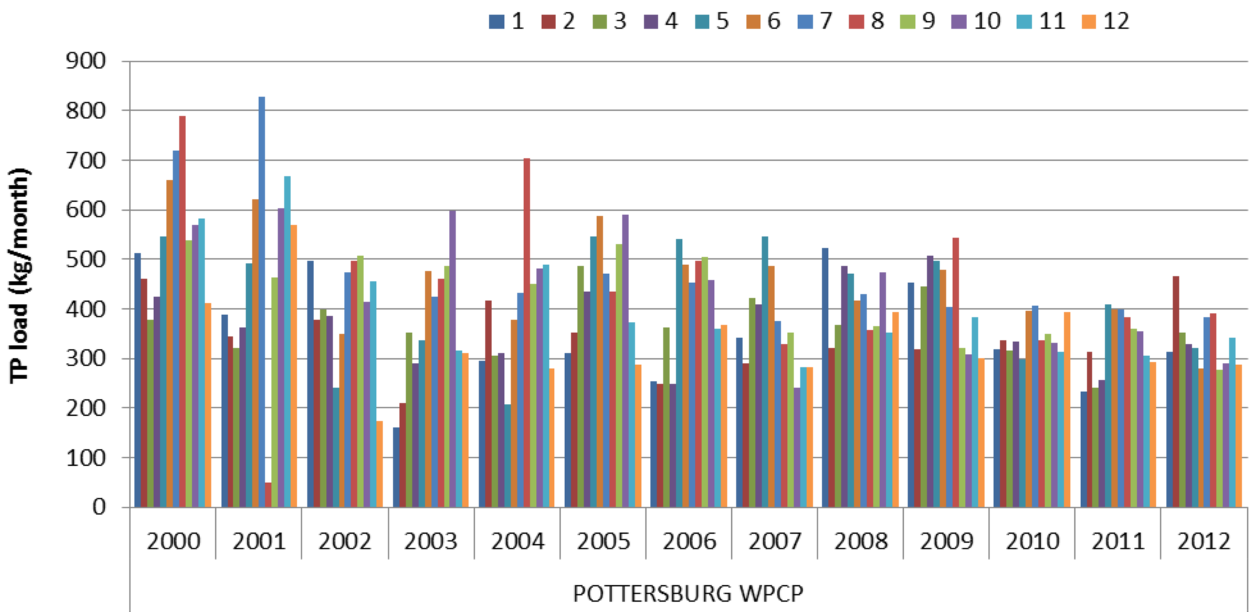
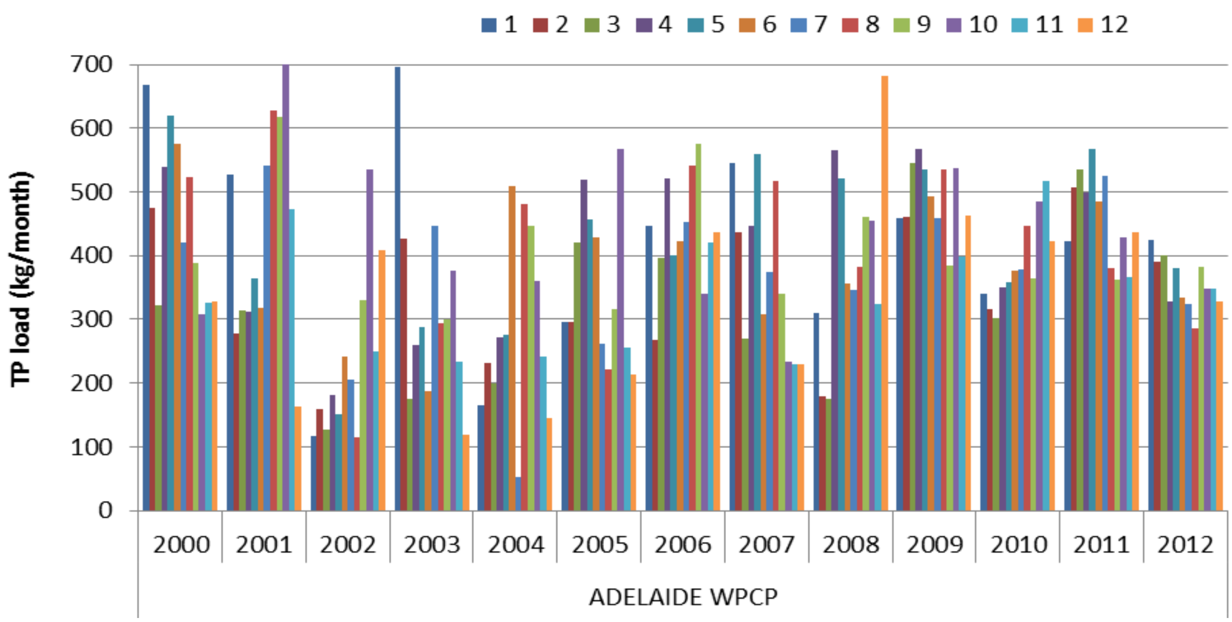
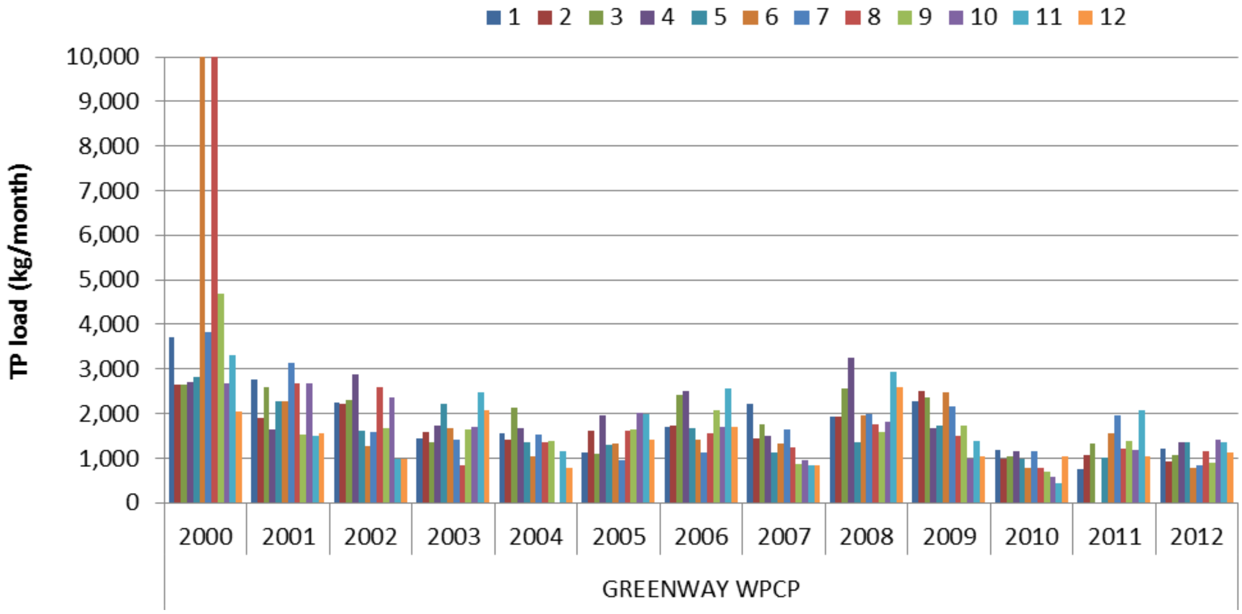

TP load (kg/year)
7,000
6,000
5,000
4,000
3,000
2,000
1,000
0
STR
VAUXHALL WPCP
POTTERSBURG WPCP
DORCHESTER STP
THAMESFORD WPCP
INGERSOLL NEW WPCP
INGERSOLL OLD WPCP
WOODSTOCK WPCP
TAVISTOCK LAGOON
SHAKESPEARE
214.8
217.7
228.5
240.3
251.3
269.3
282.6
282.7

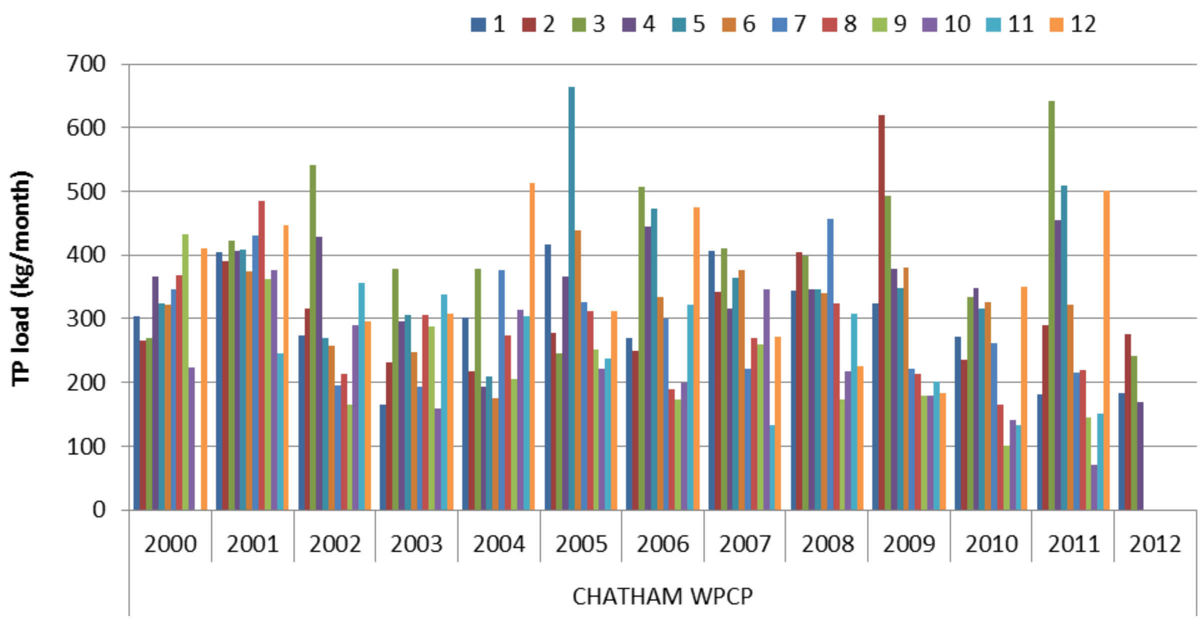
Figure 9. Monthly (1,2,3, ...) TP export from the larger WWTPs in each of the branches (Pottersburg WPCP on STR 217.7km, Adelaide on NTR 217.5km, Greenway WPCP, 207.5km and Chatham WPCP, 25km on TR).

There is no obvious monthly pattern discernable. In Greenway 2000, two extreme values exceed the scale (14,255 and 16,041 kg)









Combined sewer outfalls (CSO's) are not considered in the effluent load. For example the City of London collector pipes had a combined length of 25.5 km which is 2% of the whole system in 2011. While vast progress has been made, these remaining combined sewer pipes are mainly a result of weeping tiles and foundation drains that are connected to the sanitary private drains in homes that were constructed prior to 1985 (Podolsky, 2013).

Bypass is not included in the WWTP data presented here. They differ in size but primary bypass was extreme at the Vauxhall plant and secondary bypass was large at the Greenway plant (Table 6) probably negatively affecting the Thames River (see discussion in Section 5).

Table 6. Bypass volume for the WWTP, total (sum) for reported bypass in 2000-2012.

Facility	Volume (10^3 m^3)	
	Bypass	Secondary Bypass
Adelaide		104.1
Chatham	252.2	31.5
Greenway	41.6	8,540.4
Kilworth Heights	0.029	0.002
Oxford		59.8
Pottersburg	237.8	314.3
Tavistock Lagoon	1.6	
Thamesford	0.4	124.0
Thamesville	134.0	59.5
Vauxhall	7,220.9	409.6
Woodstock	23.5	348.0

Past (but not monitored) WWTP loads may affect water quality even now, especially in slower moving river sections and impoundments. The temporal often decreasing trend in the effluent concentration of many WWTP may mean much higher concentration in earlier years, where no data exist. There probably was a larger nutrient input in the past that has now accumulated as a legacy load in the bottom sediments downstream of the WWTPs, especially in slower moving sections (e.g., lower Thames River) and impoundments (e.g., Fanshawe Lake on the NTR).

These legacy loads could be reflected in relatively high sediment TP concentration and may explain the incidence of internal P loading and the elevated summer and fall TP concentrations and occasional cyanobacterial blooms. In addition, the gradual release of this legacy sediment TP will delay the impact of any WWTP reductions on river TP.

The extent of legacy sediment contamination could be determined by further analysis of bottom sediment downstream of large WWTPs.

In summary:

1. WWTP's usually have high TP and low TSS relative to ambient river concentrations (also found in the Grand River), however their contributions have been decreasing in some cases.

2. It is assumed that a high proportion of TP is DRP (no DRP data are available).

3. Only a small amount of N data are available. However, N-concentrations can be high.

4. Bypass volumes can be high. Combined sewers are still present, but were not studied in detail.

5. Legacy TP loads could have accumulated in the sediments leading to internal P load and delayed TP reduction in the river.

High volume WWTPs contributes substantially to the Thames River nutrient concentration. Their influence will be discussed in more detail in Section 5.

3.5 Reservoirs

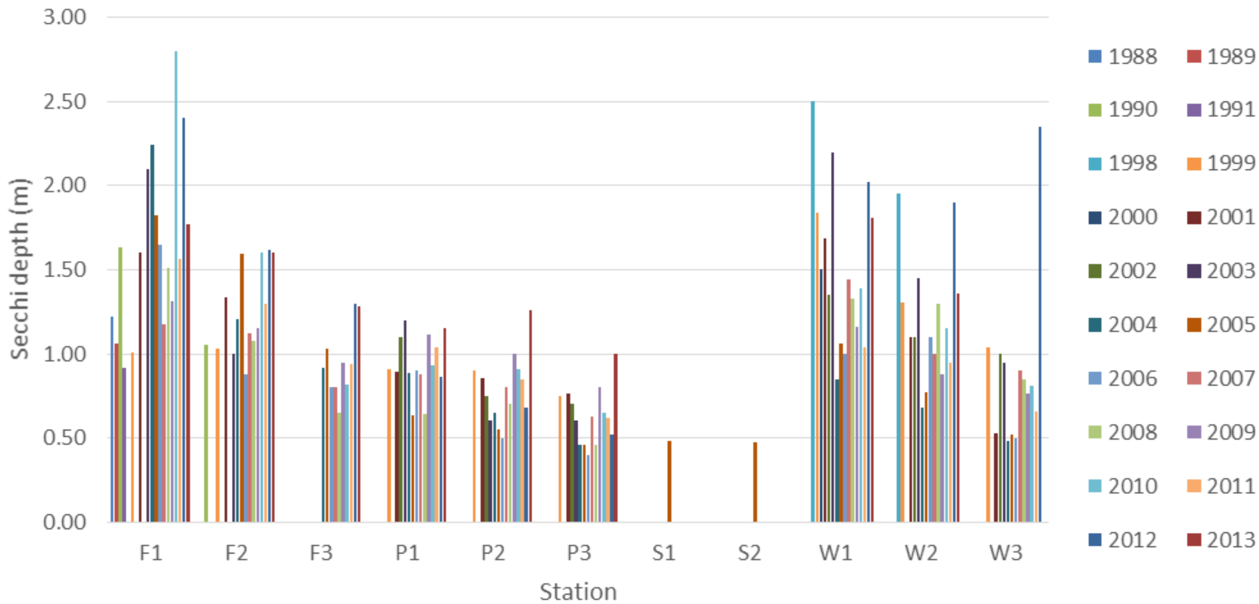
Reservoirs act like a settling pond as they trap and retain particles so that typically TP, TN and TSS concentrations decrease when passing through a reservoir. Dissolved inorganic nutrients are less affected because they do not settle out unless taken up or adsorbed by living or non-living particles. The extent of these characteristics depend on impoundment shape and water residence time, outlet depth, and direct inputs to the reservoir.

However, the retention capacity for phosphorus in lakes and reservoirs can be counteracted by internal P loading when stagnant conditions occur in the deeper water level that trigger P release from anoxic bottom sediments. This occurs especially in eutrophic reservoirs with low flushing rates with a previous history of organic and nutrient accumulation in the bottom sediments, as is the case in some Thames River reservoirs and was also observed in reservoirs along the Grand River (Loomer and Cooke, 2011).

Along the Thames River settling and retention occurs in Fanshawe Lake, Pittock Lake, and Wildwood Lake despite outlets at deeper depths and seasonal internal loading. (Because TP concentration increases with depth in the stratified summer reservoirs, deep outlets tend to have higher export than surface outlets.) Significant decreases in the studied water quality variables between upstream and downstream sampling sites are especially obvious in Wildwood Lake on Trout Creek (Section 5.1.1.4), which has the lowest annual flushing rate of the large Thames reservoirs (7.2 times per year, compared to 32 for Pittock and 38 for Fanshawe). Fanshawe (Section 5.1.3) and Pittock Lake (Section 5.2.1) decrease TSS and DRP concentration on an annual basis, even though they often experience internal P loading, especially during low-flow summer conditions (Nürnberg and LaZerte, 2005). The upper impoundments at Mitchell (no upstream data available), Stratford (Lake Victoria, no significant change along the reservoir) and St. Marys (no direct upstream data available) do not act like retention ponds probably because of both, their high flushing rates and internal P loading. These impoundments were classified as eutrophic to hyper-eutrophic (Nürnberg and LaZerte, 2006).

The reservoirs' characteristics and water quality examined in detail in Nürnberg and LaZerte (2005 and 2006, Appendix E) revealed that the reservoirs were mainly eutrophic to hyper-eutrophic and exhibited cyanobacterial blooms in some summers and falls. Additional transparency measurement by Secchi disk are available for the main reservoirs and continue to reveal annual and spatial differences (Figure 10). Lacustrine reservoir stations closer to the dam in deeper water (F1, P1, W1) have higher transparency compared to the upstream stations within the riverine sections (F3, P3, W3) indicating the settling and retention capacity of the reservoirs. Lake Victoria's sites in Stratford (S1 and S2) are not different from each other for the one available summer, showing a lack of such settling mechanisms.

Figure 10. Secchi disk transparency in Fanshawe (F), Pittock (P), Victoria (S) and Wildwood Lake (numbers indicate locations: 1, lacustrine; 2, transient; 3 riverine)



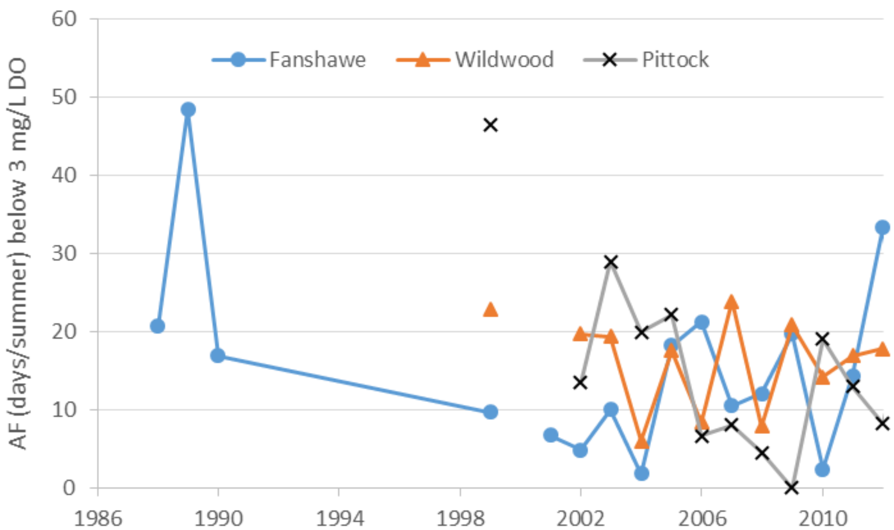
In 2004 and 2005 Zebra Mussels were discovered in Fanshawe Lake and may have contributed to higher transparency (Section 3.6 and Nürnberg and LaZerte, 2006).

Recent profile data confirm the previously determined hypoxia of the reservoirs. The three large reservoirs stratify in the summer and usually experience a period of hypoxia, at which dissolved oxygen concentration (DO) decreases to below 2 mg/L and below. In the fast-flushed and shallow reservoirs such stratification rarely occurs. Nonetheless occasional hypoxia was also determined in shallow Lake Victoria (e.g., 31 Aug 2012, 3.5 mg/L and 30 Aug 2013, 2.5 mg/L at 2 m depth). No lake measurements are available in Mitchell Lake, but DO in samples from just below the dam often tend to be below 5 mg/L, which means that hypoxia probably occurs above the dam in the reservoir.

There are many small impoundments throughout the Thames River watershed. For example, the small pond in the Fullarton Recreation Area (0.4 km up the Neil Drain at 272.3 km of the NTR) displays occasional hypoxic conditions despite its shallowness (e.g., 22 Jun 2012. 2.6 mg/L DO; 3 Aug 2012, 1.1 mg/L; 19 Jul 2013, 2.0 mg/L, monitored since 2005). Similarly are the Cove Ponds (208km) affected by internal P release from their bottom sediments (Nürnberg, 2007a). We expect that other impoundments frequently exhibit hypoxia, even if they are not deep enough to stratify for longer periods. Because of the long history of eutrophication, the high nutrient concentrations, and the relatively warm water it is likely that P release occurs throughout the Thames River from sediments under more stagnant water, especially in the summer during low flow episodes.

The spread of anoxia in space and time was quantified by the anoxic factor (Nürnberg, 2004, 2002, 1995) (defined in Appendix F) in the three larger reservoirs. Its large variability between years is due to differing water levels, flows, temperature and other eutrophication related factors (Figure 11, Table 7, Fanshawe 1.8-48.3 d/summer, Wildwood 5.9-22.9 d/summer). At higher values of AF, the impact on water quality is largest, as it affects the bottom fauna and fish, as well as increases P release from bottom sediments. Because the values are computed from observed DO concentration profiles, it also indicates the amount of stratification and stagnancy, which is related to temperature. The possibility of fish kills increases at high values because sudden destratification and mixing events can render the water column, including the surface water, hypoxic. Such events can be transient and hard to catch in routine monitoring efforts, but examples are available for all studied impoundments. (Fanshawe, 30 Jul 2013, water column <5.6 mg/L surface-10m; 15 Aug 2012 <5.3 also at mid-lake station F2. Wildwood, 6 Aug 2001 <3.3 surface-6m. Pittock, 21 Jun 20112 <5.3 surface-5m. Victoria, 31 Aug 2012 <5.5 surface-2 m. Mitchell, 21 Aug 2007 <5.5, Fullarton, see above)

Figure 11. Summer anoxic factor for DO profile values below 3 mg/L.



Note: unusually strong stratification in Fanshawe 2012.

The previously applied approach for estimating internal load in Fanshawe Lake from anoxic factors (modeled AF to account for polymixis of Fanshawe Lake, Appendix F) and sediment P release rates (at least 24 mg/m^2/d based on sediment TP content, Nürnberg and LaZerte, 2005) yields a long-term (between 1988-2013) average of 1,454 mg per square meter of reservoir surface area or 3,954 kg/summer (Table 7). This approach represented a minimum estimate in the previous study, compared to a mass balance analysis that estimated about three times as much (Nürnberg and LaZerte, 2005).

The importance of climatic conditions becomes evident when comparing individual years. The Fanshawe Lake internal load value for 2005 was almost three times that of external load and much higher than the long-term average which was a third of external load (Nürnberg and LaZerte, 2006). The high 2005 internal load indicates that most of internal load and net export of TP occur during low flow years when total external loads are small and reduced flushing rates increase water temperature and anoxia.

Similar calculations for less enriched Wildwood Lake would reveal smaller internal load because the sediment is probably less enriched (no data available), while a large internal load can be expected in Pittock Lake because of its enriched status and the difference of upstream and downstream sediment TP concentration (Section 5.2.1).

Table 7. Fanshawe bottom temperature, anoxic factor based on DO profiles, and internal P load computed with a constant P release rate of 24 mg/m^2/d

Year	Jul-Aug bottom Temperature ºC	AF days	Sumer Internal load** mg/m2	kg
1988	20.7	20.7	1,986	5,402
1989	21.3	48.3	4,641	12,623
1990	20.0	16.9	1,621	4,409
1999	21.2	9.7	932	2,534
2000		large		
2001	23.1	6.7	647	1,760
2002		4.8	461	1,253
2003	23.3	10.0	962	2,616
2004	22.8	1.8	173	470
2005	24.3	18.2	1,745	4,747
2006	23.2	21.3	2,042	5,553
2007	22.1	10.5	1,006	2,736
2008	23.2	12.0	1,156	3,146
2009	21.9	19.7	1,890	5,141
2010	23.4	2.4	229	624
2011	24.3	14.3	1,376	3,742
2012*	16.7	33.4	3,205	8,717
2013	21.3	6.7	644	1,751
1988-2013	**22.0**	**15.1**	1,454	3,954
2006-2013	22.0	15.0	1,444	3,926
For comparison predicted AF:		60		

*Low bottom temperature in 2012 indicates an unusual strong stratification that enabled a long anoxic period with a high AF

**Internal load computed as product of release rate with AFx4 (predicted/observed AF=4) to extend active bottom area in polymictic Fanshawe Lake

Previous studies of Fanshawe Lake, Wildwood and Pittock reservoirs and other smaller NTR impoundments revealed that during high-flow years the water quality is relatively good with little bloom activity, while during low-flow years it was poor and cyanobacteria proliferated (Nürnberg and LaZerte, 2006, 2005). This means that poor water quality and higher phosphorus concentrations in specific years is correlated with the lower phosphorus loadings of low-flow years and that FWC-TP concentration is a better determinant of water quality than loads.

The effect of smaller impounded areas and associated drainage pumping in the lower reaches of the Thames River are discussed in Section 5.3.4.

3.6 Introduced species

The colonization of Fanshawe Lake by the zebra mussel (*Dreissena polymorpha*) was first observed in 2004, but a drawdown operation in the fall and winter 2005 seemed to have eradicated the mussel almost completely. The zebra mussel population has increased over subsequent years in Fanshawe but not nearly to numbers seen in 2005 (S. Musclow, Fanshawe CA, pers. comm). It is not known how far the mussel has spread in the Thames River.

Carp and goldfish have been observed in slow moving areas and ponds, e.g., Coves (Nürnberg, 2007a). They disturb the sediments and can deliver P as internal P loading.

Other non-native species that significantly affect the water quality of the Thames River are not known (John Schwindt UTRCA fisheries biologist, pers. comm).

4 General water quality at the main stem stations

Flow-weighted concentrations along the main stem (non-tributary) stations were examined in detail because loads are affected by the water volume that increases along the river. Some trends in loads along the main stem are briefly discussed in Section 4.4 and investigated in more detail in Section 5.

Long-term temporal and spatial trends were determined for flow-weighted averages of the water quality variables by multiple regression analysis (after log-transformation). In this way, the time trend variable “Year” was separated from the location variable “KM”; the results are reported separately below. This analysis was done separately for the three branches (TR, NTR, and STR), for both the annual and the growing season (May-Sep averages), and for all three models (EGRET, GAM and LINEAR, Section 2.3.2).

4.1 Long-term temporal trends

Annual averages of EGRET modeled flow-weighted concentrations are drawn with time for all studied water quality variables (Figure 12). FWC-TP tended to decrease with time throughout the river in all models, but was highly significant only for the lower main branch, TR (Table 8, Figure 12 for EGRET**,** and Figure 13 for all models and separate branches). FWC-DRP did not change significantly, except when computed by LINEAR. Trends in flow-weighted concentrations of TN and NO32 were often ambiguous by increasing until about 2000 and then decreasing (Figure 12 for EGRET) so that long-term changes were often not significant. However, FWC-TN decreased significantly in the lower Thames River in two models, and FWC-NO32 decreased in all three models in TR, but did not change in NTR and increased in STR according to EGRET (Table 8). Linear temporal trends in FWC-TSS (Figure 12 for EGRET) were usually not significant (Table 8).

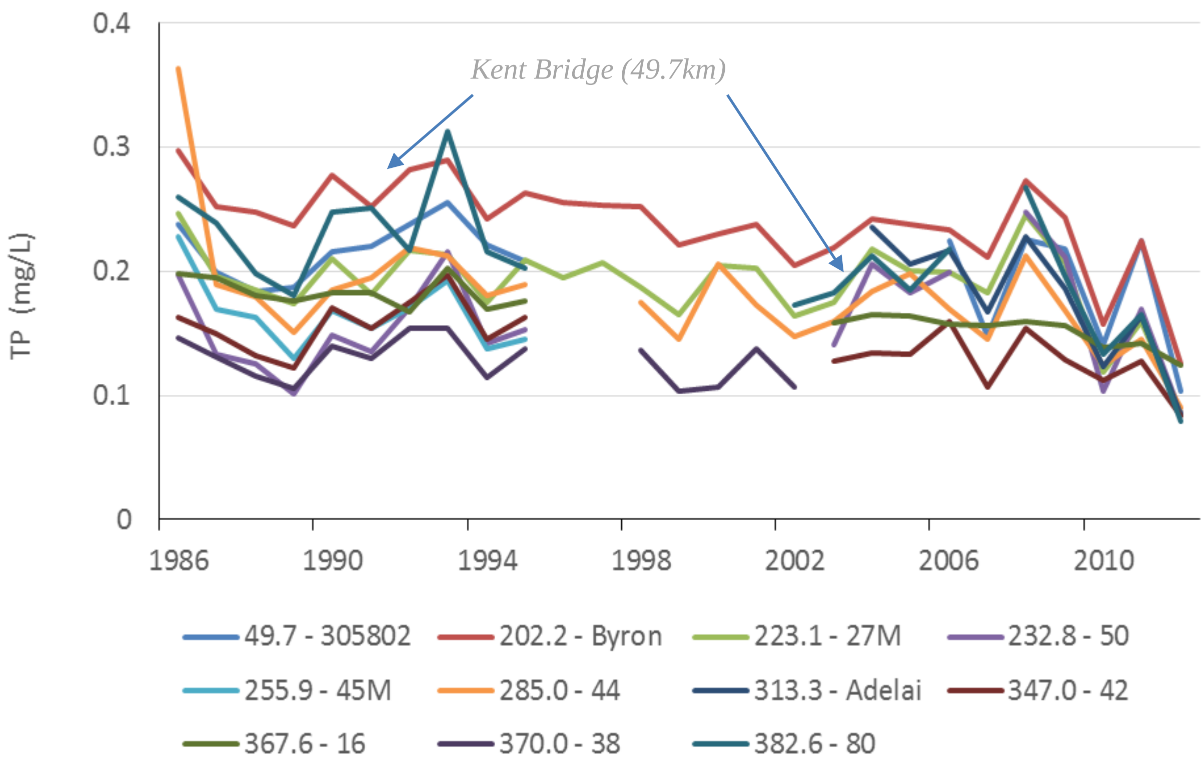
The inconsistent trends of the nitrogen variables for both annual and seasonal averages (Section 4.2, Table 8) may not be easily interpreted. Nitrogen is dominated by nitrate (all monitoring data: n=6 453, mean of proportion NO32 / TN = 0.775, median = 0.842), which was also noted in the Grand River (Loomer and Cooke, 2011). Consequently, much of any variability in TN can be explained by NO32.

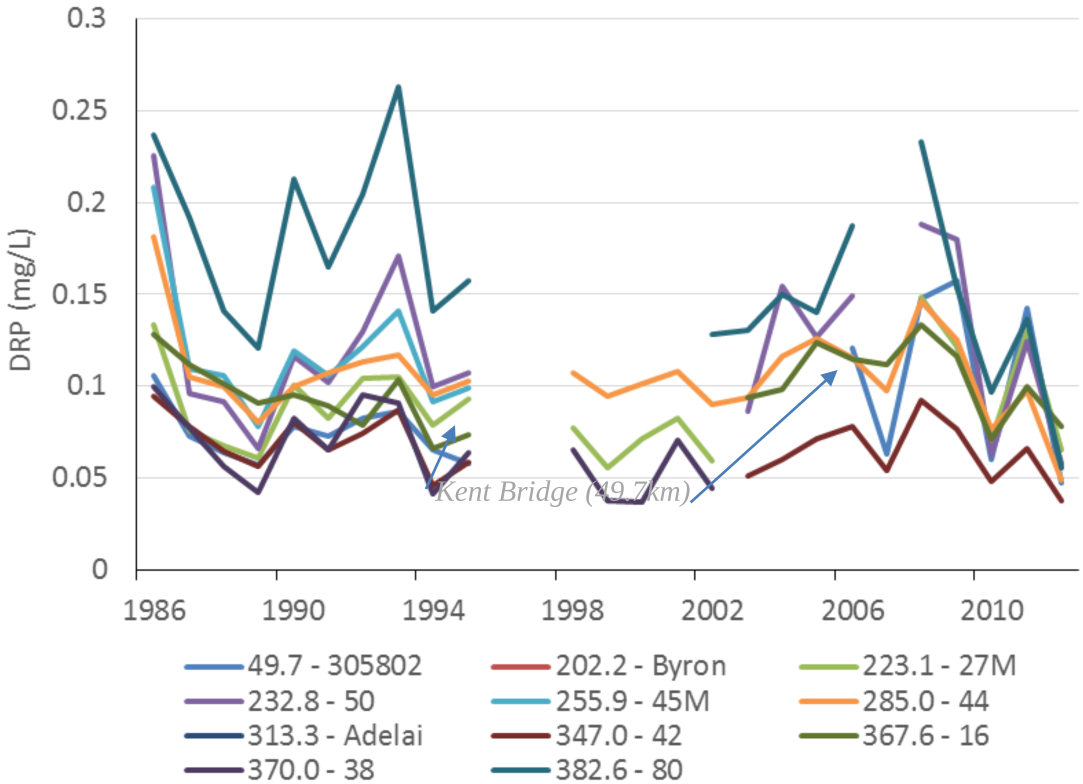
It appears that N sources have been decreasing since 2000, especially along the main branch TR, to create an overall downward trend in FWC-N. But annual FWC may be affected by the strong seasonal pattern of nitrate concentration (decline from spring to fall, Section 4.2) that can render computations from infrequent monitoring results inexact.

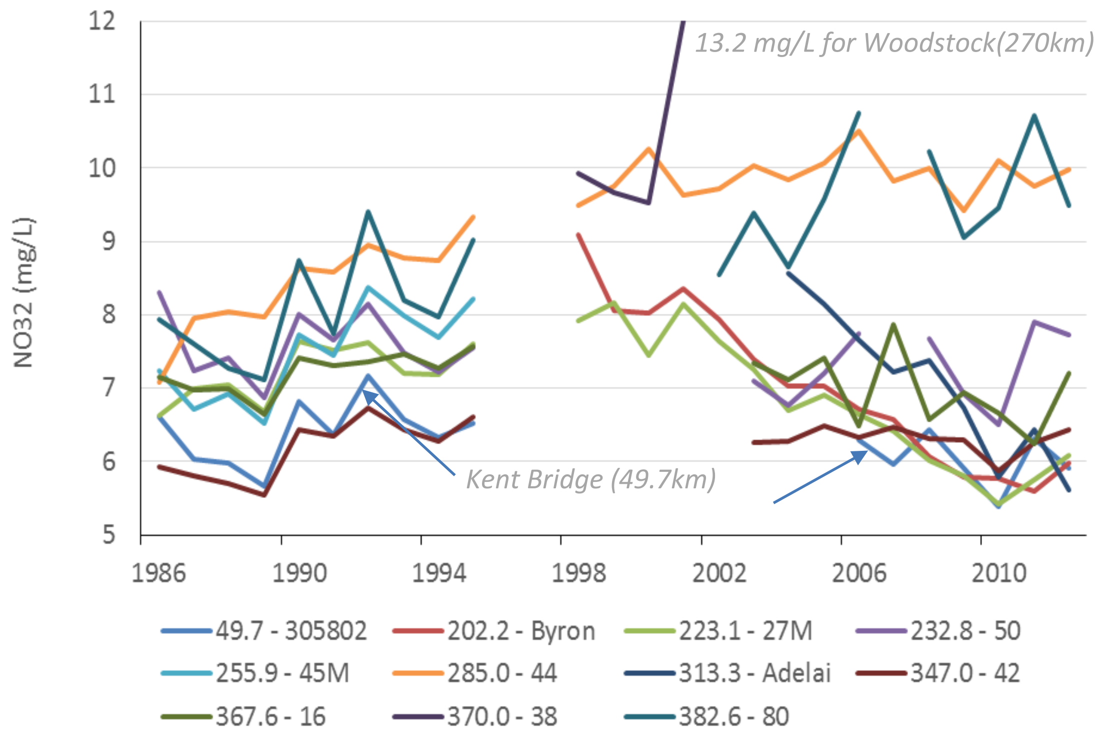
The interpretation of DRP is like-wise affected by changes due to biological uptake during the growing season and its conversion to particulates during periods with high TSS concentration. This means that low DRP concentrations do not always imply low pollution and good water quality, but can be the consequence of biogenic uptake and physical adsorption that preferentially coincide with low water quality.

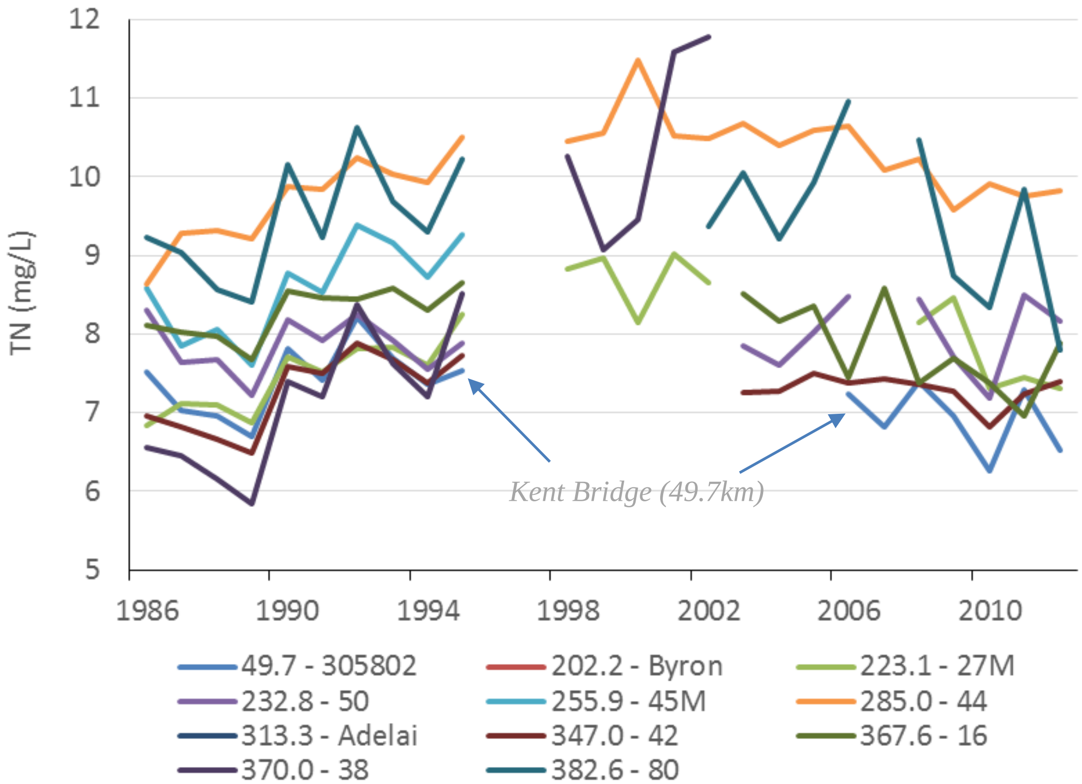
Figure 12. Flow-weighted average concentration of the study variables for all years and main-stem stations that can be computed with EGRET.

Key	Name -TR	Key	Name - NTR	Key	Name - STR
49.7	Kent Bridge (49.7km)	223.1	Clarke (223.1km)	313.3	Adelaide (213.3km)
202.2	Byron (202.2km)	232.8	Thorndale (232.8km)	347.0	Ingersoll (247km)
		255.9	St. Marys (255.9km)	367.6	Woodstock (267.6km)
		285.0	Mitchell (285km)	370.0	Woodstock Historic (270km)
				382.6	Innerkip (282.6km)

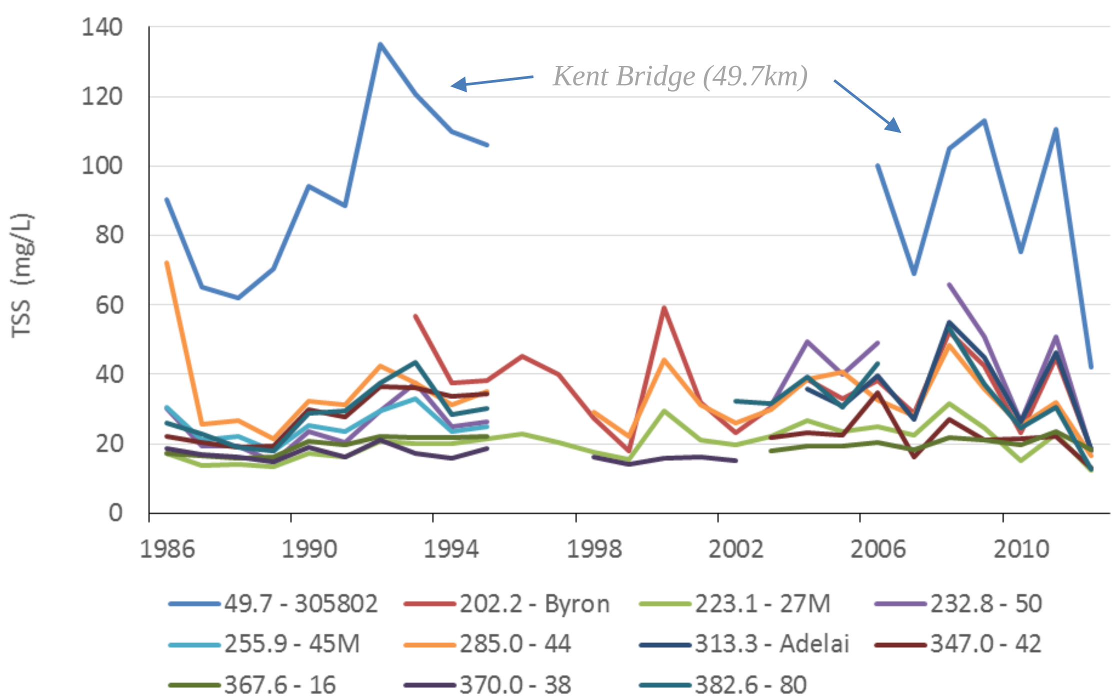



0.3
0.25
0.2
0.15
0.1
0.05
0
DRP (mg/L)
1986
1990
1994
1998
2002
2006
2010
Kent Bridge (49.7km)
49.7 - 305802
202.2 - Byron
223.1 - 27M
232.8 - 50
255.9 - 45M
285.0 - 44
313.3 - Adelai
347.0 - 42
367.6 - 16
370.0 - 38
382.6 - 80


12
11
10
9
8
7
6
5
NO32 (mg/L)
13.2 mg/L for Woodstock(270km)
Kent Bridge (49.7km)
1986
1990
1994
1998
2002
2006
2010
49.7 - 305802
202.2 - Byron
223.1 - 27M
232.8 - 50
255.9 - 45M
285.0 - 44
313.3 - Adelai
347.0 - 42
367.6 - 16
370.0 - 38
382.6 - 80



TN (mg/L)
12
11
10
9
8
7
6
5
1986
1990
1994
1998
2002
2006
2010
Kent Bridge (49.7km)
49.7 - 305802
202.2 - Byron
223.1 - 27M
232.8 - 50
255.9 - 45M
285.0 - 44
313.3 - Adelai
347.0 - 42
367.6 - 16
370.0 - 38
382.6 - 80



TSS (mg/L)
140
120
100
80
60
40
20
0
1986
1990
1994
1998
2002
2006
2010
Kent Bridge (49.7km)
49.7 - 305802
202.2 - Byron
223.1 - 27M
232.8 - 50
255.9 - 45M
285.0 - 44
313.3 - Adelai
347.0 - 42
367.6 - 16
370.0 - 38
382.6 - 80

Table 8. Time trend analysis – multiple regression with variables "year" and "station-km", separately for each branch, model and period.

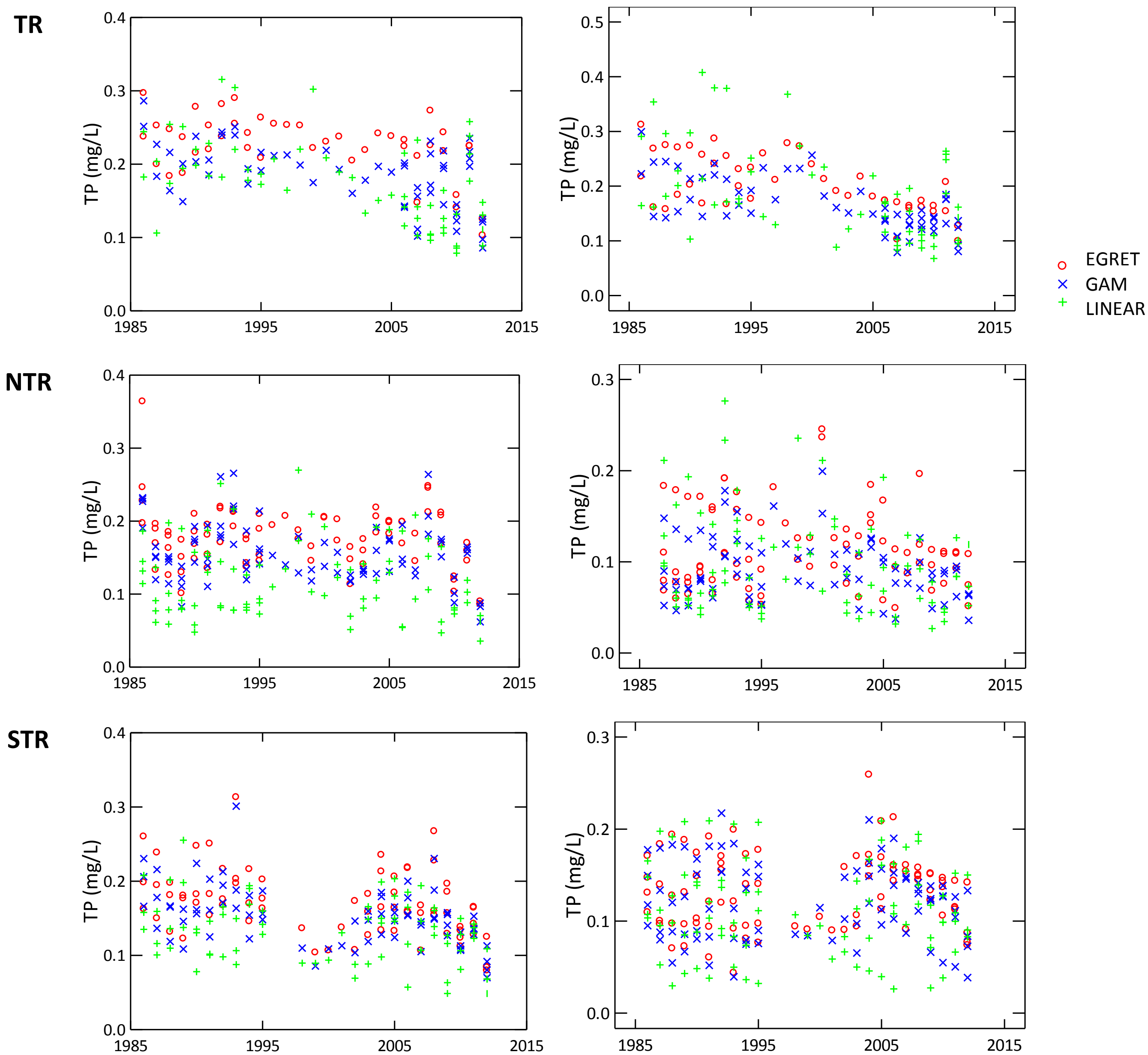
Significance of partial-p for variable "year" is presented to indicate a potential trend within 1986-2012 for annual averages. May-Sep averages in parenthesis if different from annual results.

Variable	Branch			Model
	TR[1]	NTR	STR	
TP	-***	n w/o 86	n w/o 86	EGRET
	-***	n w/o 86	-*(n w/o 86)	GAM
	-***	n (n w/o 86)	-*(n w/o 86)	LINEAR
DRP	n	n	n	EGRET
	n	+* (n)	n	GAM
	-* (n)	n	-** (n)	LINEAR
TN	n	+** (+*)	n (+*)	EGRET
	-*	n (+***)	n (+***)	GAM
	-**	n (+**)	n	LINEAR
NO32	-**	n (+*)	+***	EGRET
	-***	n (+***)	n (+**)	GAM
	-***	n (+**)	n	LINEAR
TSS	n	+* (n)	n	EGRET
	n	n (-*)	n	GAM
	n	n (-*)	n (-*)	LINEAR

Significance levels: * p<0.05; ** p<0.01; ***p<001; p>0.05, no significant trend
+, positive trend; -, negative trend; n, no significant trend
Model: EGRET, the most reliable model, but not available for stations with low sample size
GAM, available for all stations that can be combined with flow data
LINEAR, same availability as GAM, but less sophisticated
[1]Does not include Mouth for EGRET results

Figure 13. Annual (left) and May-Sep (right) average FWC-TP from the three models in main stem stations of the three river branches.

Values for 1986 are excluded for May-Sep in the NTR branch



The most consistent change is the decrease with time in FWC-TP in the TR and STR. The apparent lack of a decrease in NTR may be related to the mixing of data sources (UTRCA and LoC, Appendix C) which may conceal such a relationship.

The decrease in FWC-TP (Figure 13, right column) and the other nutrients (Table 8) in the TR over time is further supported by the significant decline shown for the May-Sep period averages. Summer relationships are less consistent for the study variables in the NTR and STR (Table 8).

Similarly, decreases in TP concentration were determined in 113 stream stations across southern Ontario, not including the Thames River (Raney and Eimers, 2013). Long-term trends (1975–2010) were evaluated and declines in TP were evident at the majority of sites (68%), including those both with (n = 49) and without (n =64) upstream municipal wastewater treatment plants. The authors of that study conclude that the increase of urban land cover at the expense of agricultural land area in southern Ontario past 35 years may be the main reason because the TP decline coincides with chloride increases, a sign of urbanization (increased impervious surfaces and therefore increased road salt applications).

Another study determined that the *risk of P pollution* from agriculture has declined in the region because of diminishing fertilizer application (van Bochove et al., 2011). The Canadian watersheds of the Great Lakes basin showed a 39% reduction in their P applications in excess of crop requirements between 1981 and 2006. P export in particular the areas north of Lake Erie that include the Thames River watershed decreased by a comparably large amount of 6-8 kg/ha in this period.

In conclusion, information about the impact of urban and agriculture is inconsistent and not the goal of this study. Any changes in the concentrations observed here are interesting and probably "real" whenever higher significant levels are reached by several models. The lack of any highly significant trends in the loads (not shown) is most certainly related to the fact that the lack of trend in water volume (no significant trend over time) outweighs any variability in concentration.

4.2 Seasonal trends

Flow-weighted average concentrations of TP, DRP and TSS varied by month as illustrated for TP (Figure 14) and TSS (Figure 15). TP and TSS tended to be highest in March, and elevated in the surrounding months, but lower in the summer. This pattern is strongest in the values calculated by the models EGRET and GAM that take the dependency of concentration on flow explicitly into account. They are less apparent in the LINEAR model that only marginally yields seasonal patterns. These monthly patterns are not appreciably different between station summaries of the three illustrated branches (Figure 14 for TP and Figure 15 for TSS).

The consistent observation of higher flows and concentrations during the spring months underlines the importance of sampling at high flows, at least for the P-compounds and TSS.

Figure 14. Monthly FWC-TP of the main stem stations on TR, NTR, and STR for the three models (EGRET, first row, GAM, second, LINEAR, third. Note the slightly different scales.)

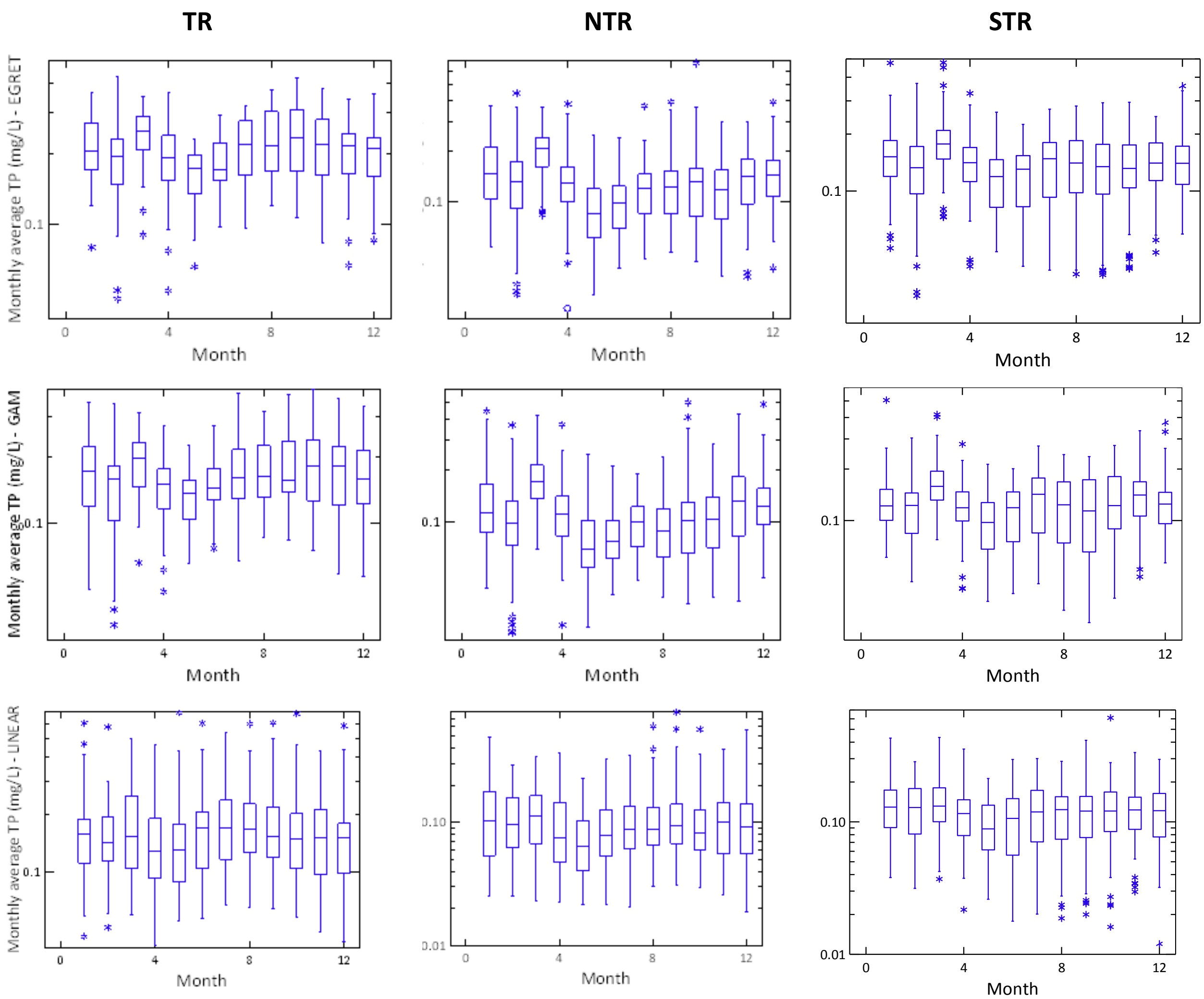
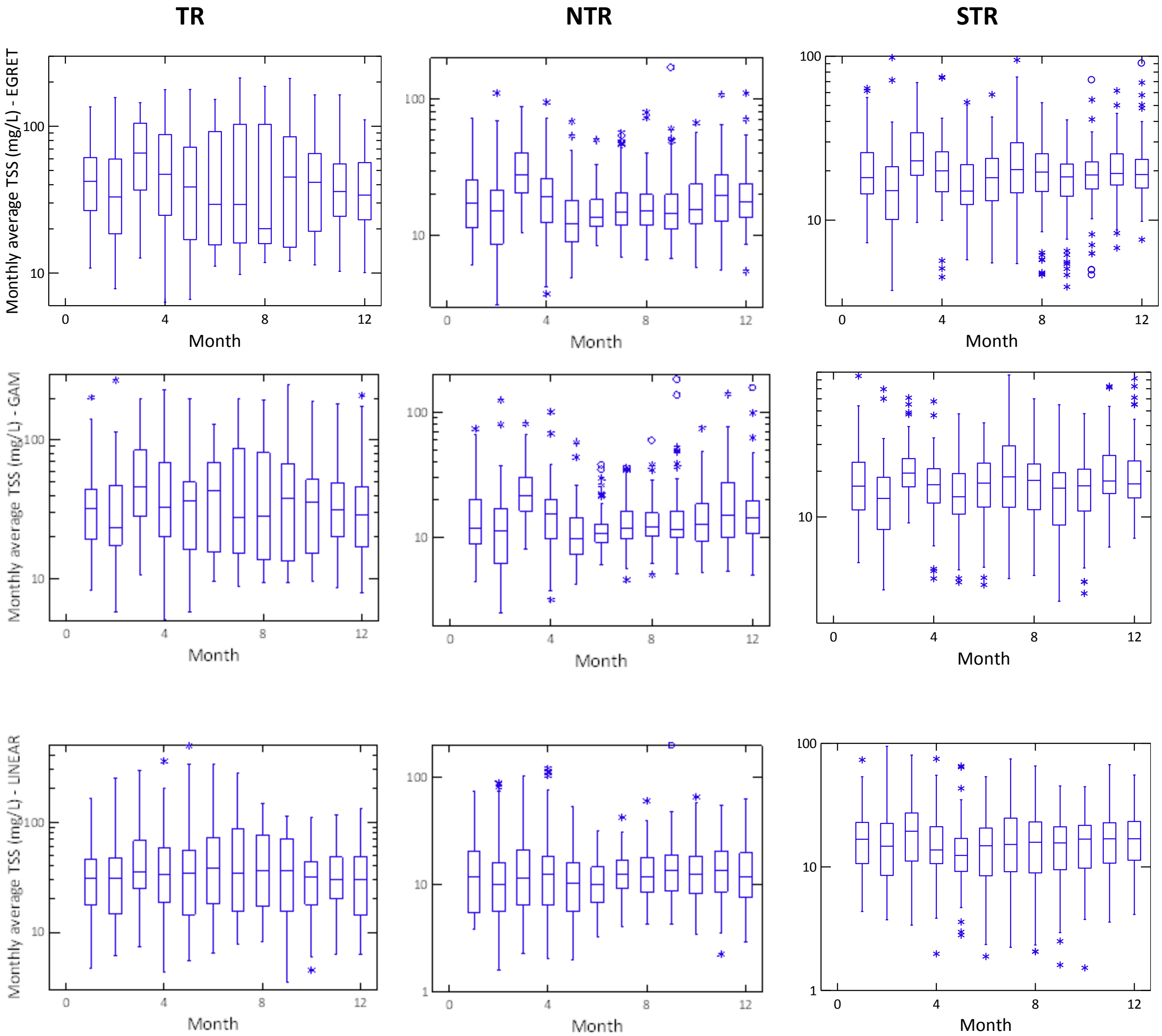


Figure 15. Monthly FWC-TSS of the main stem stations on TR, NTR, and STR for the three models (EGRET, first row, GAM, second, LINEAR, third. Note the slightly different scales.)



The **nitrogen** variables exhibit a very different but still strong seasonal variation similar to that determined in the NTR previously (Nürnberg and LaZerte, 2005, 2006). In many years nitrate concentration decreased throughout the growing season probably as a consequence of biological uptake, because periods of low nitrate concentration were correlated with high cyanobacteria abundance and blooms (Nürnberg, 2007b). Therefore, low FWC-NO32 during the growing season does not necessarily indicate good water quality as there may be an abundance of cyanobacteria. FWC-TN exhibits a similar pattern (Figure 17) because TN mostly consists of nitrate (Section 4.1).

Large spring nitrate concentrations followed by a seasonal decline has been reported in many eutrophic rivers including the Grand River, ON (Loomer and Cooke, 2011), the Mississippi–

Ohio–Missouri River Basin (Mitsch and Day, 2006) and other watersheds in the Great Lakes Basin (Chambers et al., 2001).

Figure 16. Monthly FWC-NO32 of the main stem stations on TR, NTR, and STR for the three models (EGRET, first row, GAM, second, LINEAR, third. Note the slightly different scales.)

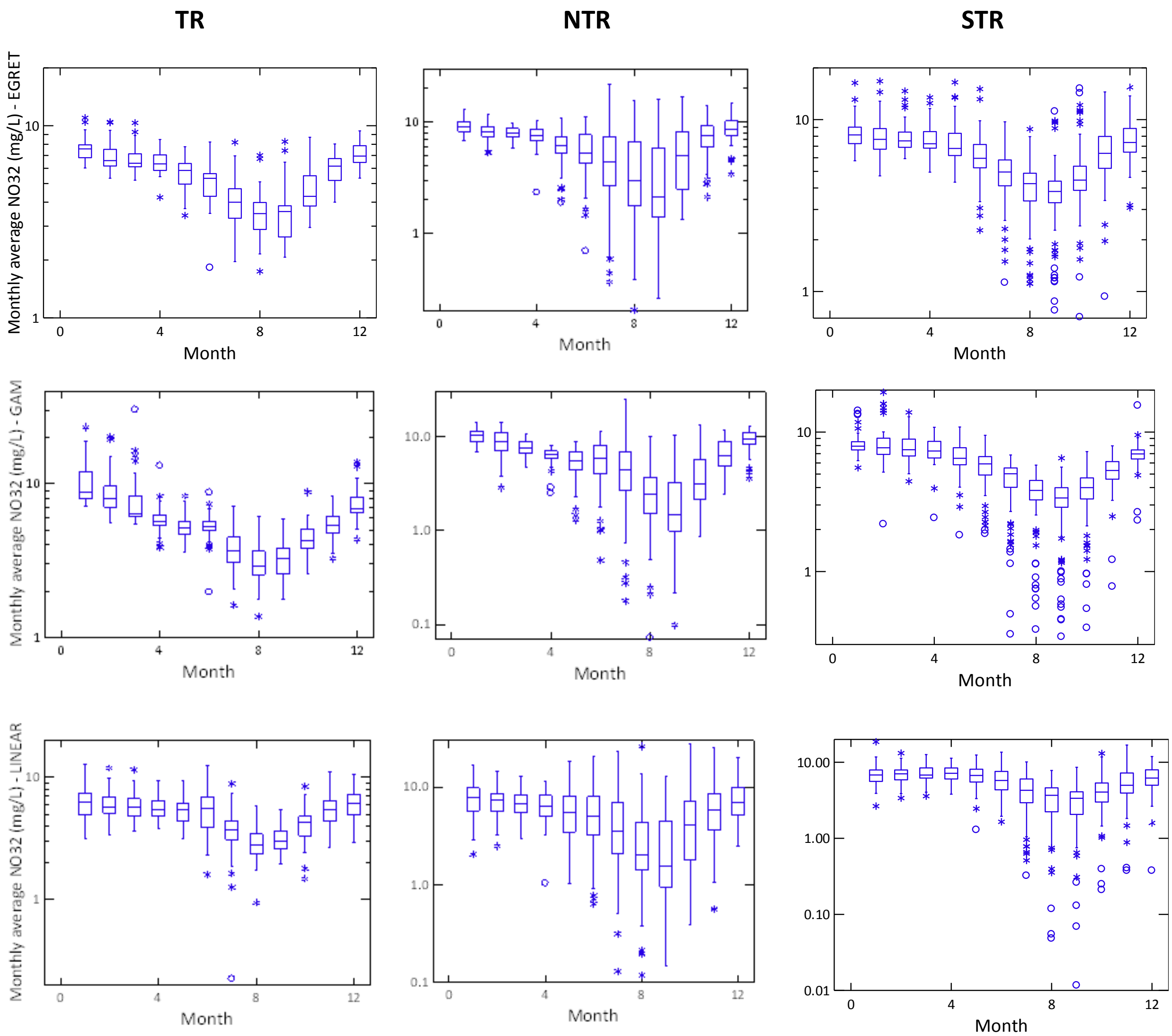
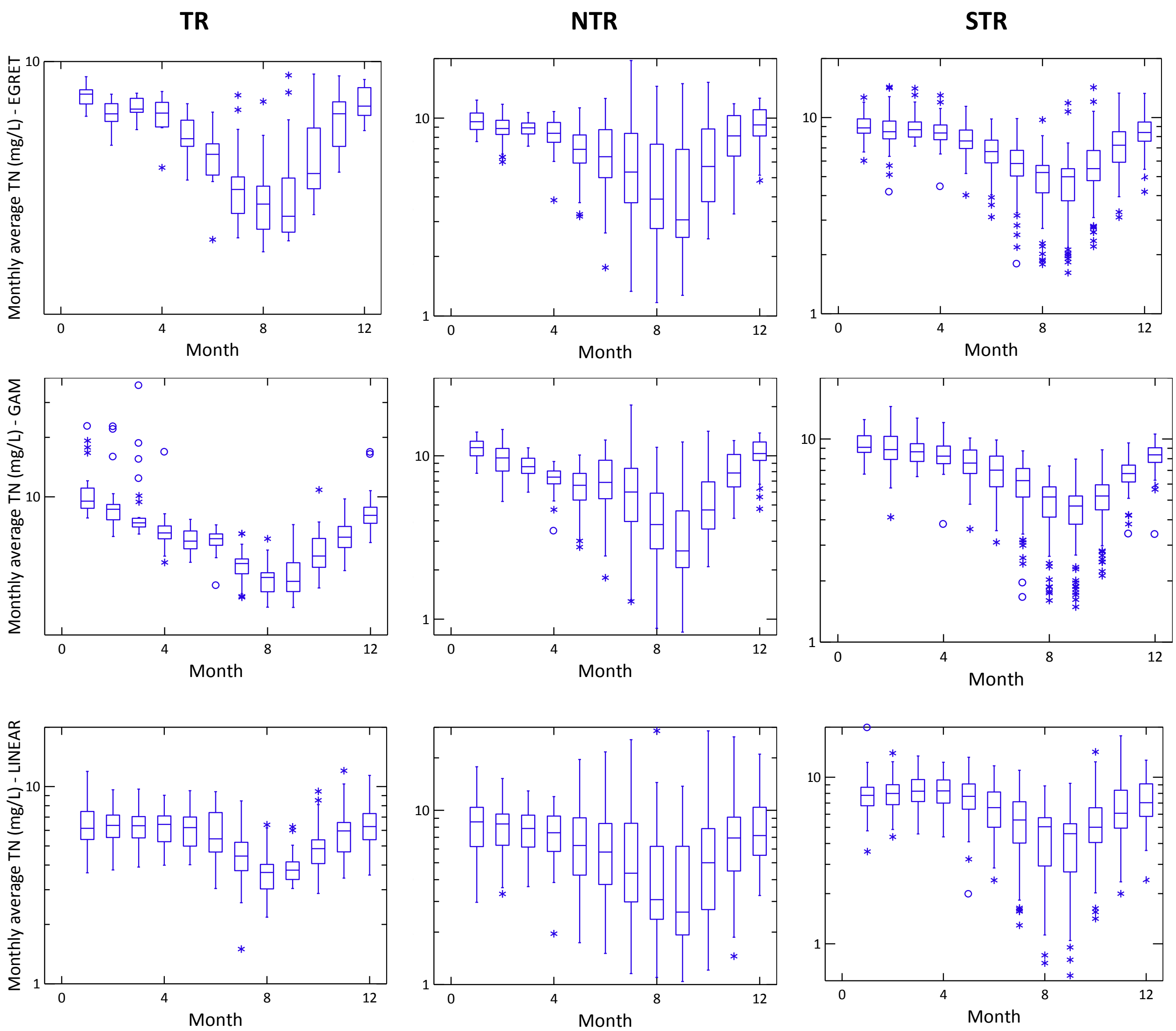


Figure 17. Monthly FWC-TN of the main stem stations on TR, NTR, and STR for the three models (EGRET, first row, GAM, second, LINEAR, third. Note the slightly different scales.)



4.3 Spatial trends

The spatial component of the multiple regression analysis described earlier (Section 4.1) indicates several significant trends (Table 9, Figure 18) even though the concentrations were flow-weighted, which compensates for the increasing flows down the river. FWC-TP decreases in TR towards the mouth, but does not consistently change in the NTR and STR. FWC-DRP decreases from upstream stations in the NTR and STR, but there is no change in the TR. A similar pattern occurs for FWC-TN and FWC-NO32, except that FWC-NO32 decreases in the TR towards the mouth. FWC-TSS is higher at the upstream stations in the NTR, but not in the STR, while it increases in the TR towards the last station before the modelled mouth station, where it decreased again (Figure 18). Explanations for these patterns include nutrient (DRP, TN, NO32) sources upstream of each of the branches, disturbances that elevate TSS in upstream NTR, and resuspension in the lower TR and will be investigated in more detail in Section 5.

Table 9. Spatial trend analysis – multiple regression with variables "year" and "station-km", separately for each branch and model.

Significance of partial-p for variable "km" is presented to indicate a potential trend within 1986-2012 for annual averages. In most cases variation due to "km" was significant (at least p<0.05). Note that the "station at the mouth" was modeled for GAM and LINEAR estimates, but is not available for EGRET (Table B).

Variable	Branch			Model
	TR	NTR	STR	
TP	-***	n	n	EGRET
	n	+*	n	GAM
	-***	+*	+*	LINEAR
DRP	n.a.	n	-*** (213km lev. 270 out)	EGRET
	n	-***	-*** (213km lev.	GAM
	n	-***	n	LINEAR
TN	n.a.	-***	-*** (213km lev. 270 out)	EGRET
	+*	-***	-*** (213km lev. 270km out)	GAM
	n	-***	-** (213km lev)	LINEAR
NO32	-***	-***	-*** (270km out)	EGRET
	n	-***	-**	GAM
	-***	-***	n	LINEAR
TSS	+***	-***	n	EGRET
	+***	-**	n	GAM
	+***	n	+***	LINEAR

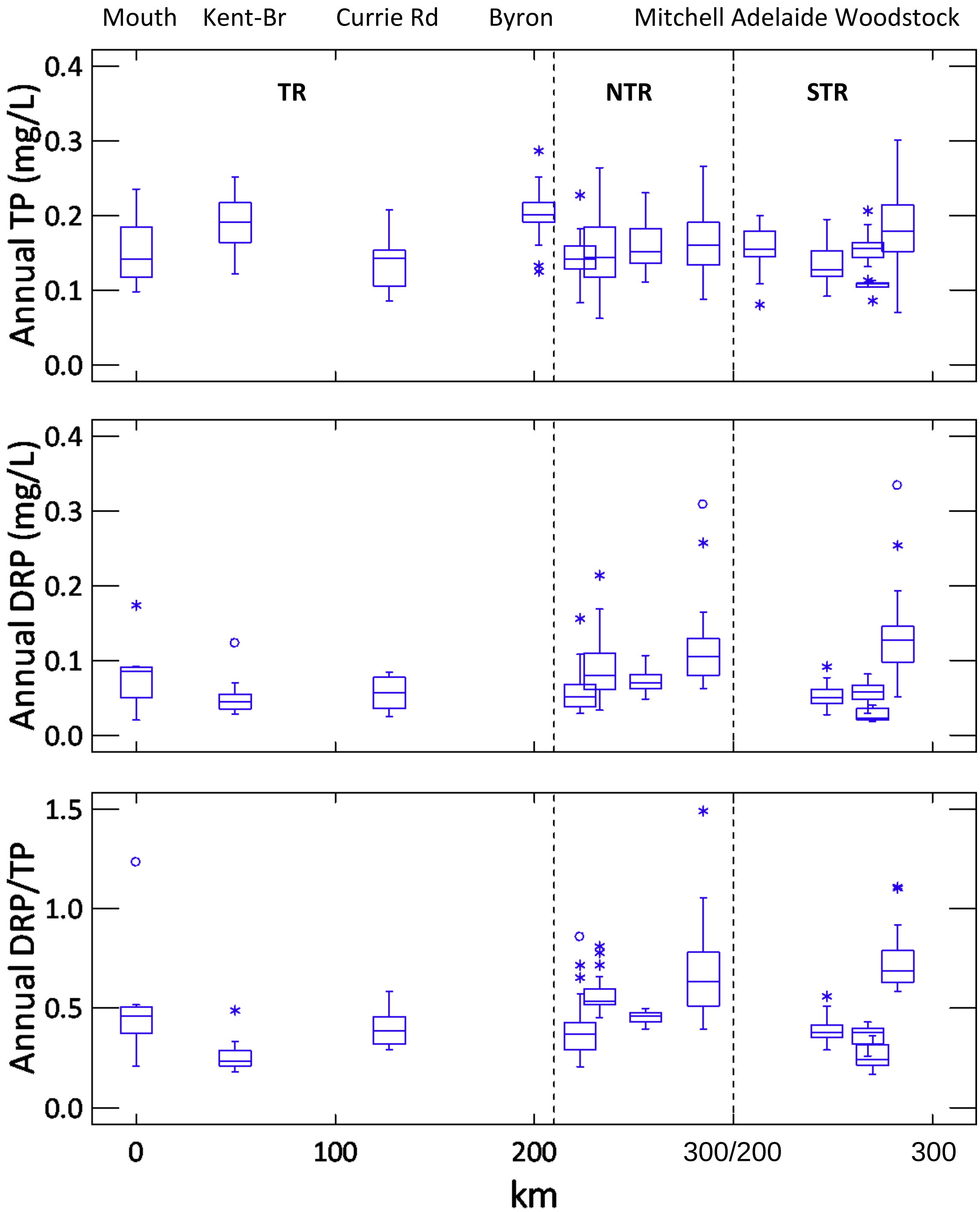
Significance levels: * p<0.05; ** p<0.01; ***p<001; p>0.05, no significant trend
+, positive trend, increase from upstream to downstream;
-, negative trend, decrease upstream to downstream;
n, no trend
lev, this station has high leverage and therefore is influential; out, this station is an outlier
Model: EGRET, the most reliable model, but not available for stations with low sample size
GAM, available for all stations that can be combined with flow data
LINEAR, Same availability as GAM, but less sophisticated

An overview of the pattern on the main stem (non-tributary) stations is presented next. In the following figures (Figure 18, Figure 19 and Figure 20), GAM modeled results were used because they include the most stations with flow-based concentration estimates and includes the Mouth. However, results for the Mouth may be imprecise because of the lack of monitoring data at that location and a strong flow-dependency of the data used (Section 2.3.3).

Annual median FWC-TP fluctuates mostly between 0.1-0.2 mg/L (Figure 18) and is especially high at the upstream station of the STR. It is also high at the first station below the forks, i.e. the most upstream station of the TR, which is based on CoL station, Byron (no DRP and TN data available). Because of a tendency of CoL TP values to be elevated (Appendix C), it is not clear whether the Byron Station values are comparable. (All other GAM data originated at least partly from the PWQMN program.) FWC-DRP is large at upstream stations of the STR and NTR and the proportion of TP that is DRP is larger as well (Figure 18), as further discussed in Sections 5.1.1 and 5.2.1. Using the GAM model and including the Mouth's computed values, FWC-TP and FWC-DRP are variable in the lower part of the TR without any apparent spatial trend.

Figure 18. Variation of annual FWC-TP, FWC-DRP and DRP/TP ratio along the Thames River (GAM-modelled)

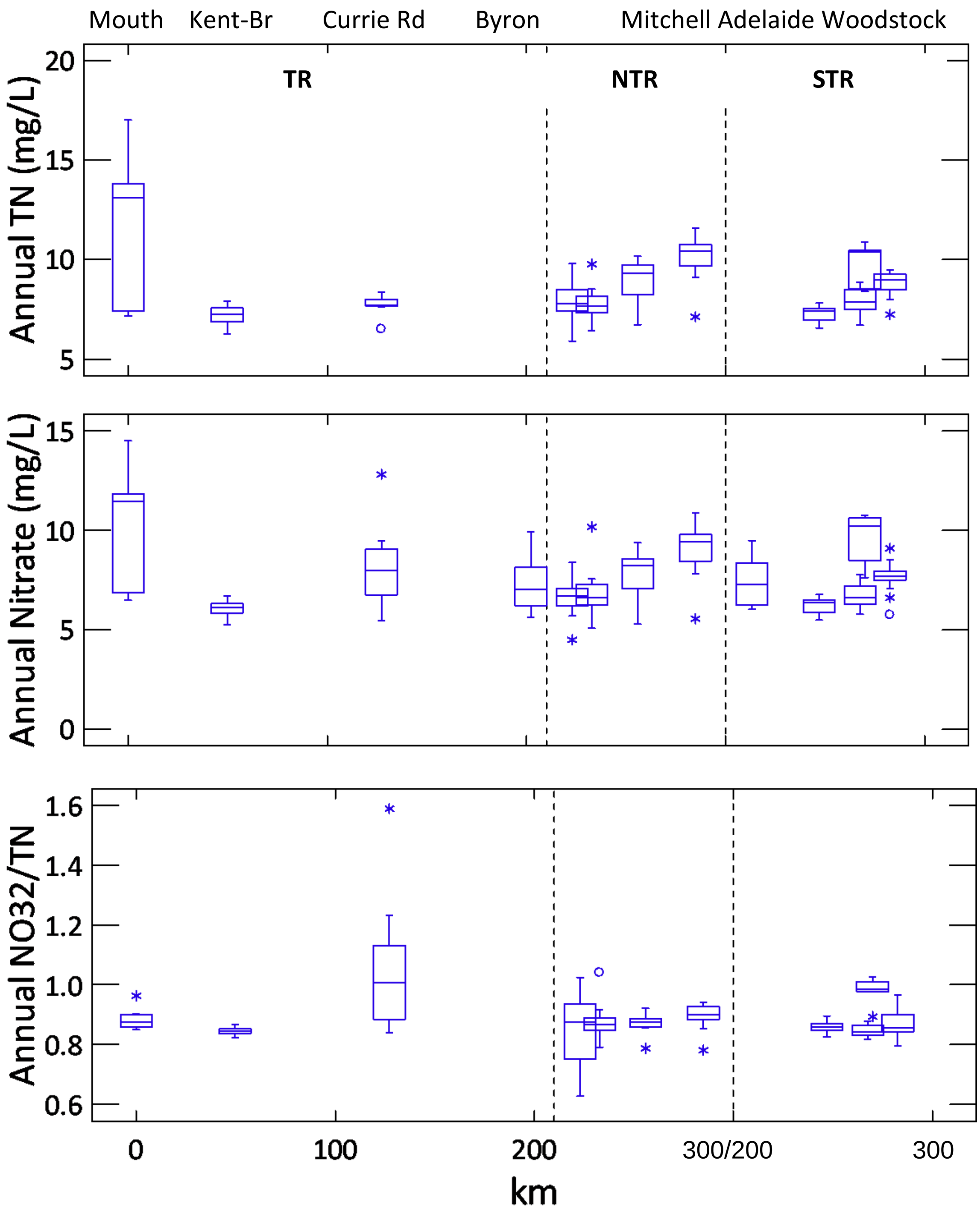
Broken vertical lines indicate the start of the different branches, names depict location of some WQ stations



Nitrogen compounds tend to be higher at the upstream stations in both branches at about 10 mg/L and are much increased at the mouth, median about 13.5 mg/L (Figure 19). FWC-TN is mostly comprised of nitrate and annual median proportion is usually around 0.9 (Figure 19, third panel).

Figure 19. Variation annual FWC-TN and FWC-NO32 along the Thames River (GAM-modelled)

Broken vertical lines indicate the start of the different branches, names depict location of some larger plants.

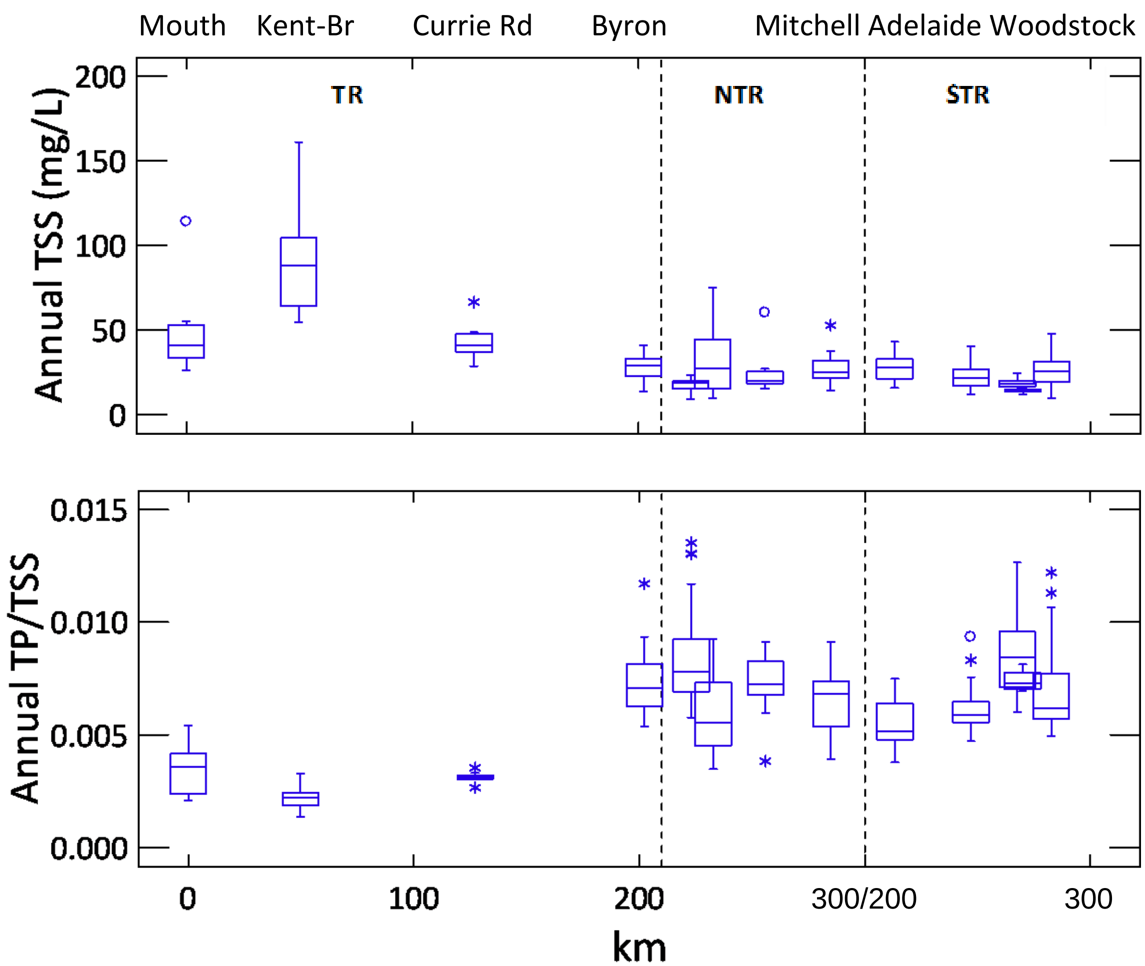


Annual median FWC-TSS fluctuates around 30 mg/L, but is increased in the lower TR (Figure 20) despite its almost constant elevation (Figure 4). As a measure of resuspension of sediments derived from fields, shoreline and river bottom TP/TSS ratios were computed. Low TP for high TSS are found in particle-rich waters and may indicate that much of TP is particulate and therefore less available to phytoplankton, as in the lower main branch, TR. On the other hand, high TP relative to TSS values indicates nutrient enrichment by a nutrient source of inorganic P, such as fertilizer or sewage. Because WWTP generally release more TP than TSS, they can have an increasing effect on the ratio.

Considering the relatively constant TSS concentration in the NTR and STR, the pattern of the TP/TSS ratio indicates potential P availability (Figure 20). The ratio is highest above 200 km in both branches. Different from the upper branches, TSS is elevated in the TR so that here the low TP/TSS ratio is probably the result of high sediment content in the river water. It seems unlikely that the distinct decrease in elevation (Figure 4) contributes to the observed TP/TSS changes in the TR and other factors, including land use changes, are probably more decisive.

Figure 20. Variation annual FWC-TSS and TP/TRR ratio along the Thames River (GAM-modelled)

Broken vertical lines indicate the start of the different branches, names depict location of some larger plants.



More detailed investigation about seasonality and spatial variability is presented in Section 5 that describes conditions along the individual river branches.

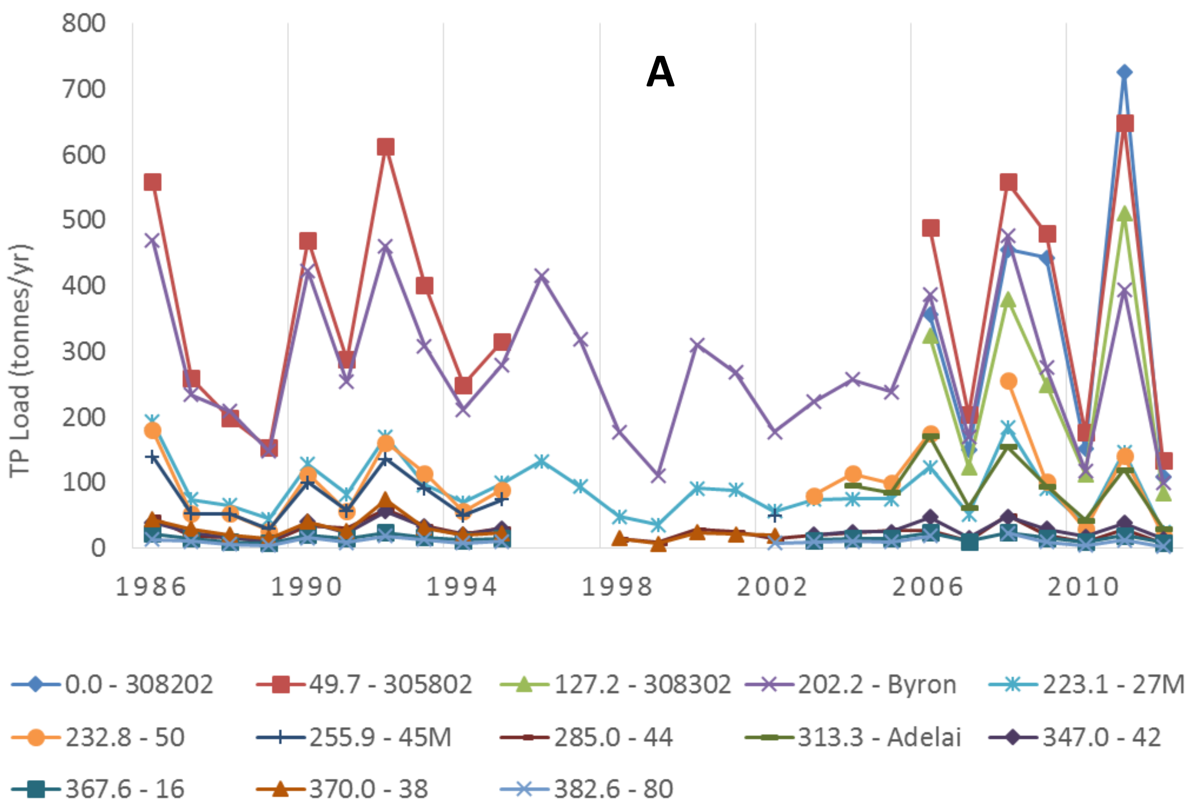
4.4 Spatial and temporal trends in loads

Unsurprisingly, annual average TP loads for the main stem stations on the Thames River increase from the upstream stations of the North Thames River (NTR: Ln 32, downstream of Mitchell, D014, 44) and South Thames River (STR: 282.6 km, Oxford County Rd 29, Innerkip, D021, WQ80) to the downstream station at Thamesville, 49.7 km above St. Clair (WQ305802), (total Thames River, Figure 21; NTR and STR only, Figure 21B). This increase parallels the increases in flow and is highly variable between years. Loads are more than double of either NTR or STR stations just above the confluence at the Forks at the first station (7 km) downstream of the confluence at Byron Stn., 202km (Figure 21A).

Figure 21. Main stem station TP loads for all years computed with GAM for the whole river (A) and above the Forks, just the stations on NTR and STR (B).

Note that the inflow to Lake St. Clair at 0km is modeled and results are discussed in Section 5.3.3)

Key	Name -TR	Key	Name - NTR	Key	Name - STR
0.0	Mouth (0km)	223.1	Clarke (223.1km)	313.3	Adelaide (213.3km)
49.7	Kent Bridge (49.7km)	232.8	Thorndale (232.8km)	347.0	Ingersoll (247km)
127.2	Currie Rd (127.2km)	255.9	St. Marys (255.9km)	367.6	Woodstock (267.6km)
202.2	Byron (202.2km)	285.0	Mitchell (285km)	370.0	Woodstock Historic (270km)
				382.6	Innerkip (282.6km)



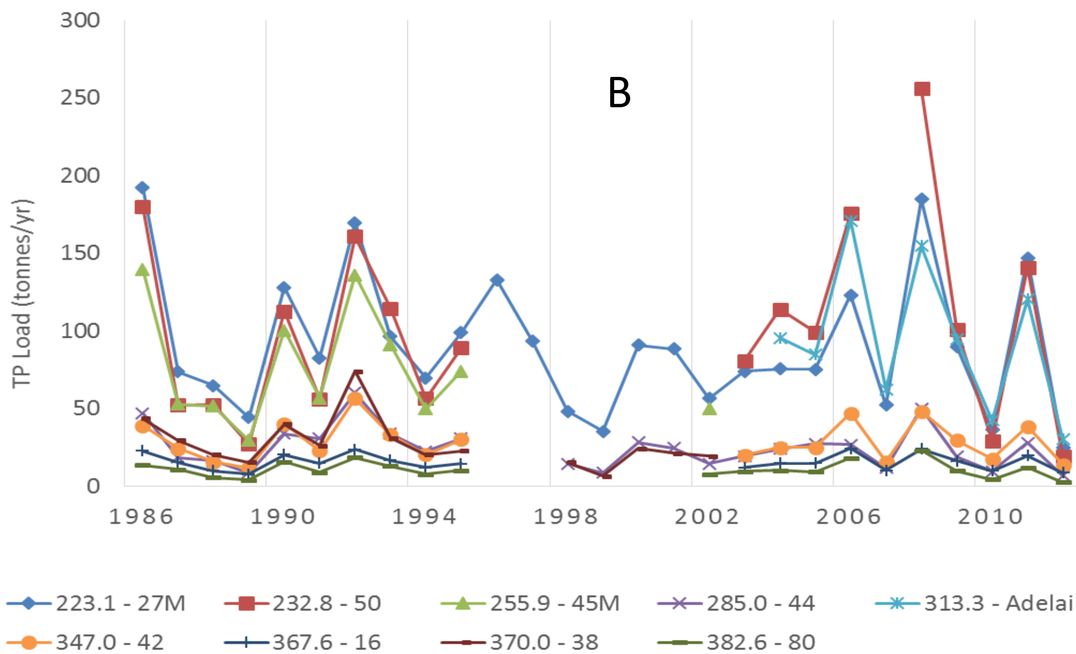
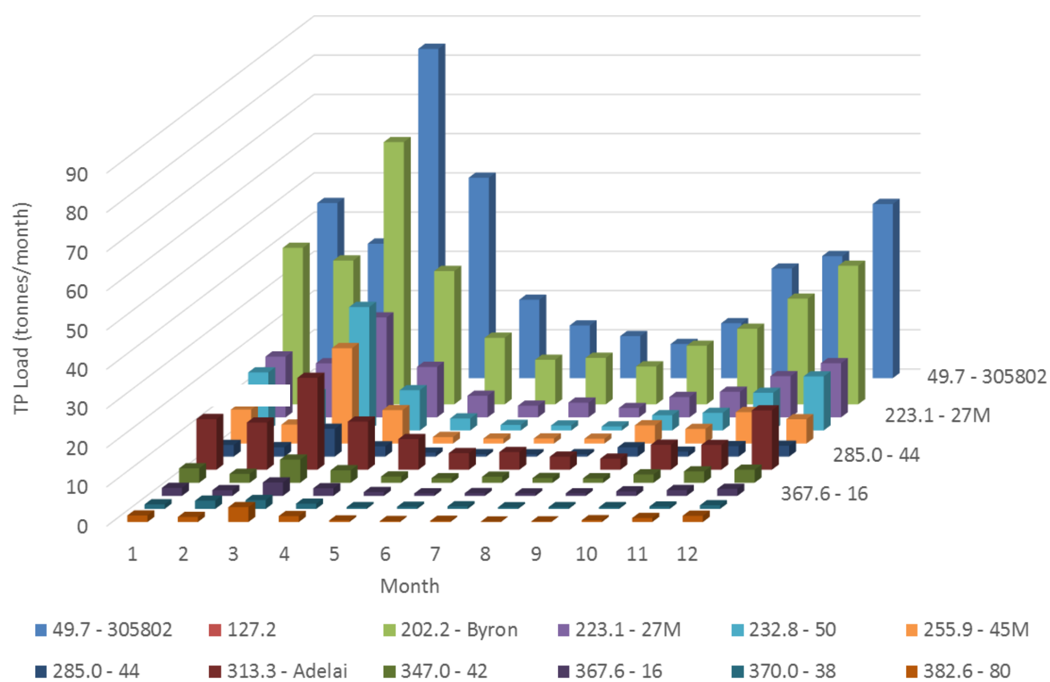


Figure 22. Monthly long-term averages for all main stem station TP loads that can be computed with EGRET.

Key	Name -TR	Key	Name - NTR	Key	Name - STR
49.7	Kent Bridge (49.7km)	223.1	Clarke (223.1km)	313.3	Adelaide (213.3km)
202.2	Byron (202.2km)	232.8	Thorndale (232.8km)	347.0	Ingersoll (247km)
		255.9	St. Marys (255.9km)	367.6	Woodstock (267.6km)
		285.0	Mitchell (285km)	370.0	Woodstock Historic (270km)
				382.6	Innerkip (282.6km)



Because loads are affected by the water volume that increases along the river and are highly variable between years, and because water quality is more directly related to nutrient concentrations, it is preferable to compare flow-weighted concentration if the goal is to determine pollution sources.

5 Detailed water quality analysis

The following sections present a detailed analysis of the water quality variables for the three branches separately starting at the most upstream stations. This investigation is to highlight any discernable sources and sinks for the measured variables. To improve clarity, numbered water quality stations have the pre-fix of "WQ" and the station location is indicated by km and name.

In general, the following report section is segmented to cover stretches from one GAM-modeled station to the next. (GAM stations are used because they are more frequent than EGRET stations, and, unlike the LINEAR model, the estimated daily concentrations are still flow based, as described in Section 2.3.2. For flow gauges at the GAM-modeled stations refer to Appendix Table 24.) FWCs and loads from the GAM-modeled stations are used in statistical tests (t-tests, with data paired by year) to determine statistically significant changes between GAM stations.

These station results are then supplemented with results from less frequently sampled locations without flow data, which are simply time averaged and not directly comparable to GAM model results. Nonetheless, in interpreting and comparing the loads and concentrations along these river sections these data can be cautiously used to illustrate trends and perhaps indicate major differences.

Loads from WWTPs (Section 3.4) and major reservoirs (Section 3.5) were added next. For additional interpretation and explanation of the obtained results, GIS-based land use information was consulted (Section 3.1). Special and important tributaries and river sections are described in more detail in separate paragraphs.

More information for land use is found in Table 4, for WWTPs in Table 5 and for the various river branches in the respective summary tables (e.g., Table 10, Table 14, and Table 18).

5.1 North Thames River Branch (NTR details)

Characteristics and results for NTR stations (between 290 km and the fork at 209 km above the mouth) are summarized in Table 10 and Figure 23A and B.

Table 10. Summary of concentrations and loads along the NTR

GAM station information on the main stem are shaded across; other GAM stations refer to tributaries

km	Common Name (River km)	WQ-Stn	Tributary	Facility	Model	Yrs	Annual average concentration (mg/L)					Load (t/yr)				
						n (TP)	TP	DRP	TN	NO32	TSS	TP	DRP	TN	NO32	TSS
209.8	Dundas (209.8km)	Dundas			Avg		0.126			5.2	13.4					
214.1	Medway Cr (214.1km)	Medway	Medway Cr		GAM	25	0.169			8.3	85.1	15.16			703	7,871
214.1	Granton WWTP (214.1km)		Medway Cr	Granton	WWTP	8	0.476				15.9	0.02				1
214.4	Richmond (214.4km)	Richmond			Avg		0.143			5.0	13.5					
216.5	Stoney Cr (216.5km)	96M	Stoney Cr		GAM		0.086	0.059	6.7	5.2	34.6	2.49	2.70	189	103	1,528
217.5	Adelaide WWTP (217.5km)			Adelaide	WWTP	13	0.484				5.3	4.60				51
223.1	Clarke (223.1km)	27_Clarke		dn Fanshawe	GAM	27	0.145	0.059	8.0	6.7	17.5	90.76	40.09	4,793	3,870	10,986
226.1	Wye Cr (226.1km)	98	Wye Cr		Avg		0.125	0.066	7.2	6.3	14.2					
232.7	Thorndale WWTP (232.7km)			Thorndale	WWTP	new										
232.8	Thorndale (232.8km)	50		up Fanshawe	GAM	19	0.153	0.090	7.8	6.8	30.5	100.95	60.16	4,731	4,103	21,169
241.9	Gregory Cr (241.9km)	95			Avg		0.050	0.029	5.5	4.9	6.5					
247.2	Dstr St. Mary's	43			Avg		0.087	0.032	6.2	5.3	16.0					
248.6	Fish Cr (248.6km)	90	Fish Cr		GAM		0.090	0.036	8.5	7.8	11.8	6.61	2.73	599	553	856
254.8	Nineteen Cr (248.6km)	45			Avg		0.084	0.029	6.5	5.6	14.2					
255.9	St. Marys (255.9km)	15_45			GAM	11	0.161	0.073	9.0	7.8	24.1	75.63	34.04	4,044	3,536	11,773
256.5	St. Marys WWTP (256.5km)			St.Marys	WWTP	5	0.231		5.4	4.6	9.6	0.34		8	6	14
256.5	Trout Cr ds (256.5km)	64	Trout Cr	ds Wildwood	GAM	25	0.077	0.012	4.5	3.8	15.0	4.30	0.64	250	207	826
256.5b	Trout Cr us (256.5km)	66	Trout Cr		GAM	24	0.224	0.118	7.7	7.0	37.0	4.07	2.18	132	121	669
262.4	Otter Cr (262.4km)	94	Otter Cr		Avg		0.048	0.023	5.0	4.5	8.2					
263.1	Flat Cr (263.1km)	89	Flat Cr		Avg		0.057	0.016	5.2	4.5	18.3					
263.6	Road 133	67			Avg		0.095	0.032	6.8	5.7	13.2					
265.9	Avon R ds (265.9km)	25	Avon R.	ds Victoria	GAM	25	0.213	0.069	7.2	5.8	36.4	11.63	3.65	372	298	2,019
265.9	Stratford WWTP (265.9km)		Avon R.	Stratford	WWTP	5	0.143		17.0	13.8	3.6	0.95		107	95	24
265.9	Avon R us (265.9km)	310302	Avon R.		GAM	7	0.172	0.065	6.5	5.4	42.2	4.70	1.78	161	133	1,174
268.0	NTR-Glengowan (268km)		Glengowan			4	0.071									
272.3	Neil Drain (272.3km)		Neil Drain	Fullarton Pond	Avg	1	0.219			1.6						
274.2	Black Cr (274.2km)	92	Black Cr		Avg	9	0.067	0.029	5.1	4.2	10.7					
285.0	Mitchell (285km)	44			GAM	25	0.167	0.118	10.2	9.2	26.5	24.62	17.47	1,417	1,272	3,997
285.2	Mitchell WWTP (285.2km)			Mitchell	WWTP	13	0.341				5.0	0.51				7
286.8	Whirl Cr (286.8km)		Whirl Cr			4	0.069									

Figure 23A. Map of NTR annual average phosphorus concentration (*UTRCA*)

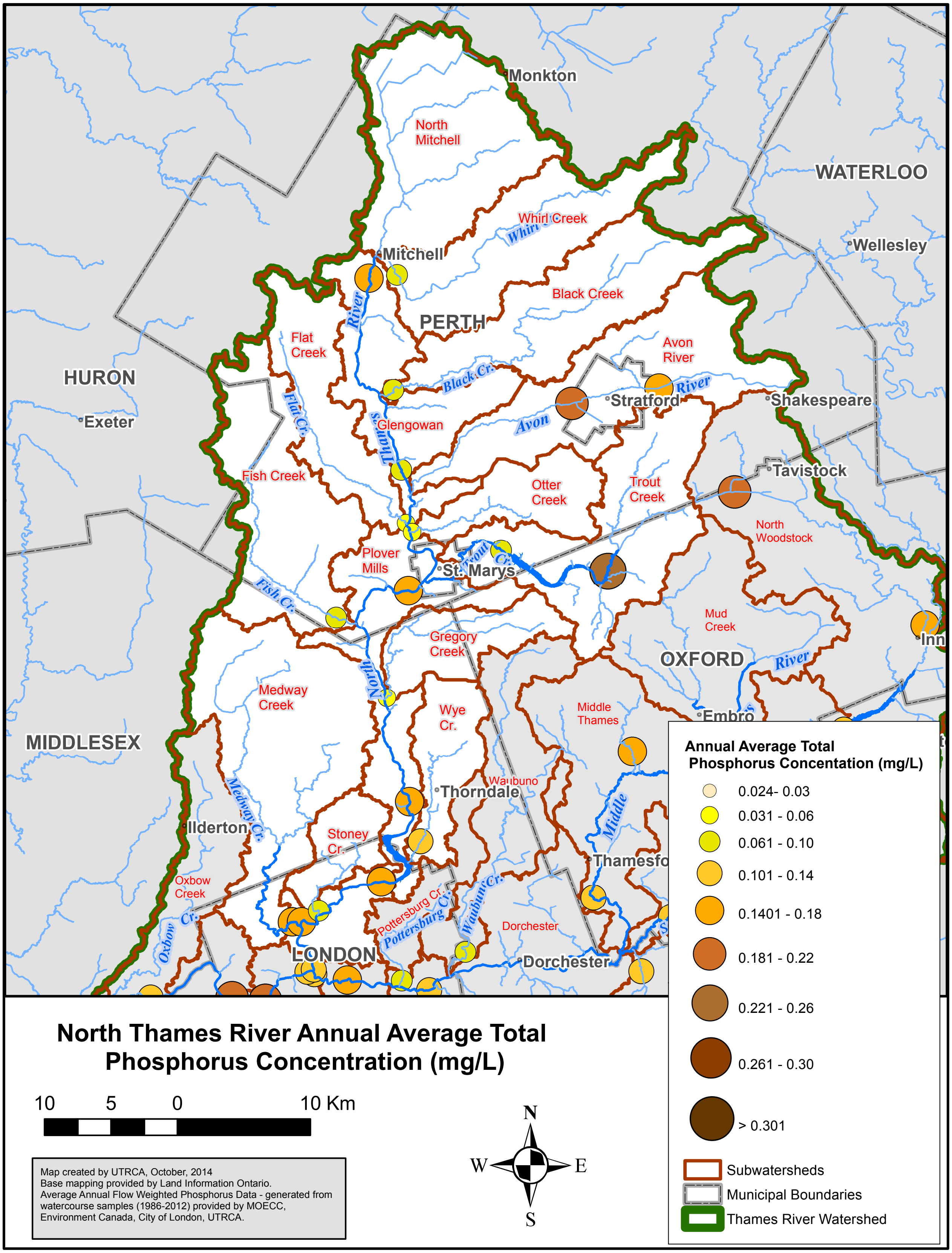
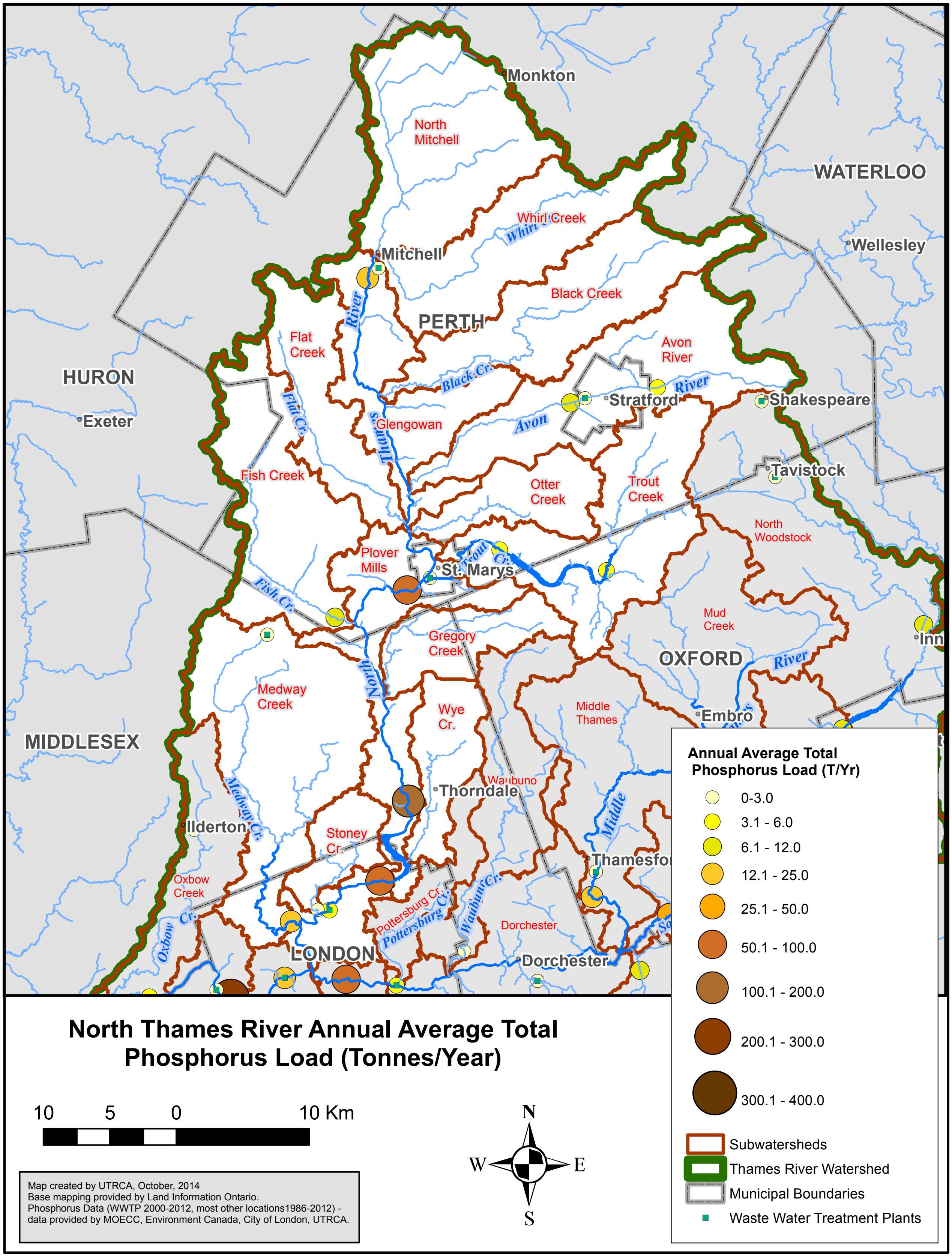


Figure 23B. Map of NTR annual average phosphorus load (*UTRCA*)



5.1.1 Rkm 285-256

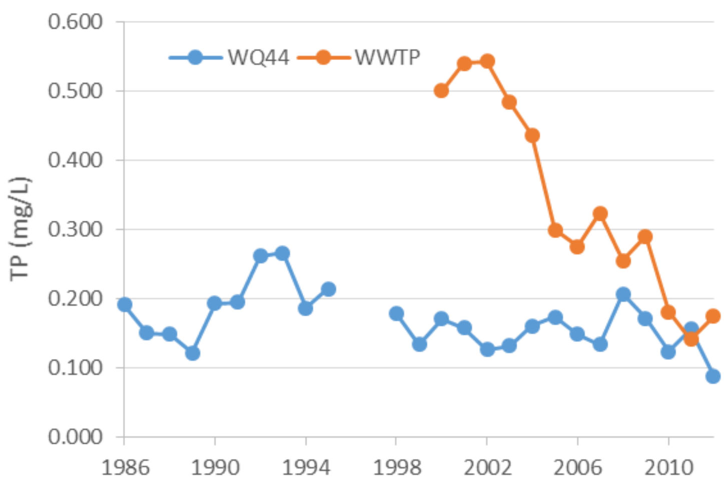
The first monitoring station on the NTR branch (GAM WQ44, Table 10) is 52 km below the start of the open channel in the headwaters, 285.0 km upstream of the mouth, downstream of eutrophic Lake Mitchell, and just below the inflow of Whirl Creek (286.8km) and the effluent of the Mitchell WWTP (285.1km). At this point the watershed is 90% agriculture with 68% tile drained, 4% urban (Town of Mitchell) and 0.04% impounded (Table 4).
All investigated nutrient variables are significantly higher (t-test, paired by year, $p<0.05$) than those at the next GAM station, WQ15_45 at St. Marys 255.9 km. This occurs despite the fact that the downstream station data are limited to earlier years (1986-1995, 2002).

The Mitchell WWTP TP load is currently relatively small at 0.51 t/yr on average for 2000-2012 compared to the long-term load at WQ44 of 24.6 t/yr (1986-2012). Concentration and loads at Mitchell WWTP have been decreasing consistently (Figure 24) and the 2010-2012 TP and TSS average were only 0.165 mg/L and 3.2 mg/L. While there is no significant correlation detectable, the long-term general decrease in the WWTP may be reflected by a decrease in the NRT concentrations at WQ44 which average 0.122 mg/L TP in three recent years (2010-2012), compared to the long-term annual average of 0.167 mg/L (Figure 24).

If the temporal trend of the Mitchell WWTP TP is extended backwards to earlier years, the earlier high values suggest that the WWTP has historically loaded adjacent river sediments and impoundments with TP. The subsequent release of this legacy sediment TP load will delay the impact of any WWTP reductions on river TP.

As observed for other WWTPs, Mitchell WWTP does not increase the TSS concentration at WQ44, which is FWC-TSS 26.5 mg/L.

Figure 24. Comparison of monthly and annual TP concentrations at Mitchell WWTP and at GAM-modeled WQ44.



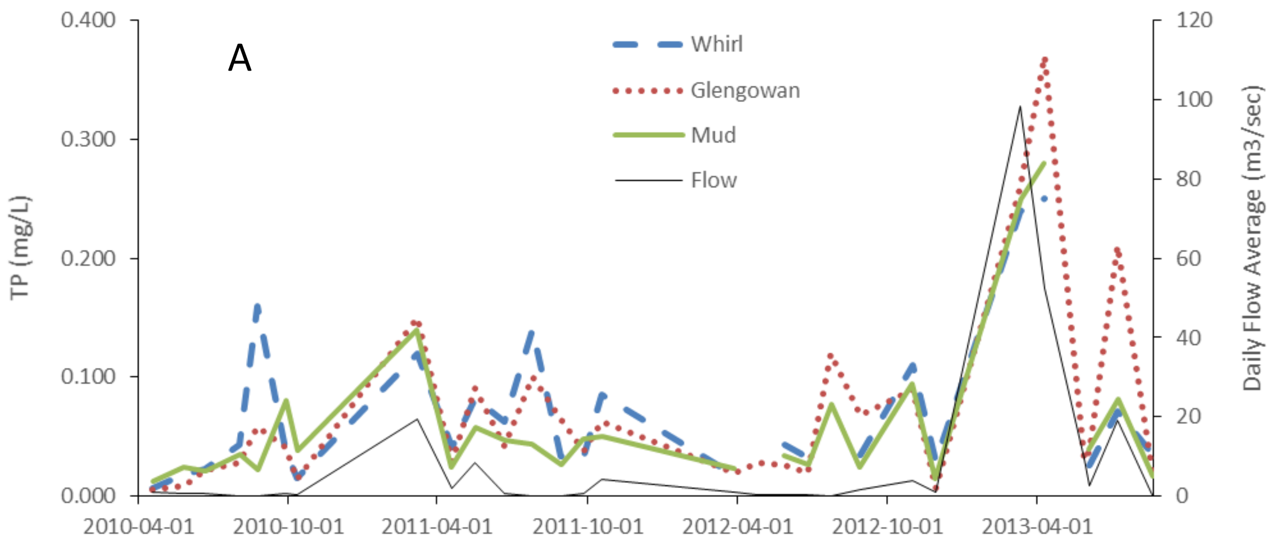
There are several tributaries joining the NTR in the section to the next GAM station WQ15_45 at St. Marys 255.9 km with variable influences on the main stem which are discussed next.

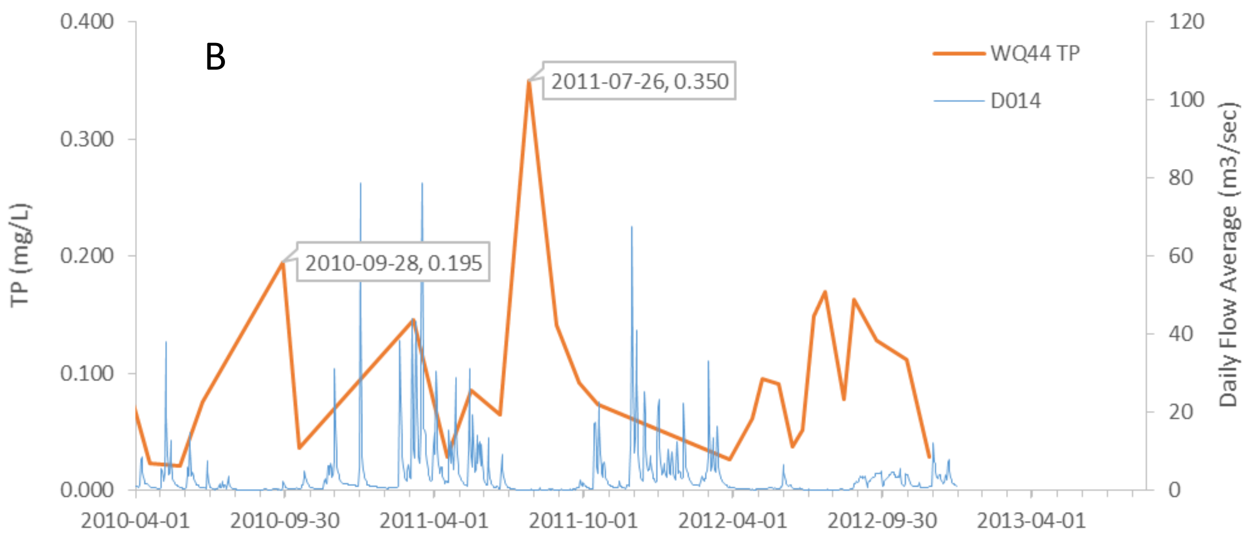
5.1.1.1 Additional creeks and NTR station monitored within the Report Card program

Three stations were repeatedly sampled for TP (and *E.coli*) in the Report Card program since 2010. Two of them are located within this section, and one (Mud Creek) joins the STR and is discussed in more detail in Section 5.2.3. Whirl Creek flows into the NTR at 286.8km about 2 km upstream of WQ44. Glengowan station is located right on the NTR at 268km. TP concentrations vary between 0.005 and 0.370 mg/L. Because no flow gauges are available at these sites, gauge D014 (284.9km) at WQ44 was used as an indicator of flow variability. Flows at WQ44 reveal a correlated pattern for most dates (Figure 25), so that high TP concentrations occur mostly at higher flows.

However, especially in Whirl Creek, extremely high TP concentrations also happen at least once each summer and fall when flow is small (25-Aug-10, 26-Jul-11, 31-Oct-12). This could indicate release of P during warm and stagnant conditions from bottom sediments as internal loading. Similar flow-TP relationships were observed in the NTR just downstream of the inflow of Whirl Creek at WQ44 (284.95km, Figure 25, B; for GAM FWC-TP see Figure 24A).

Figure 25. Observed TP concentration for special study tributaries (A, NTR tributaries Whirl and Glengowan Creek, and Mud Creek of the STR, Section 5.2.3) and for routine NTR station WQ44 (B) compared to flow at D014 (2010-2013).





5.1.1.2 Neil Drain (Fullarton Recreation Area) and Black Creek

Black Creek joins the NTR about 9 km downstream of WQ44 at 274.2km. Its monitoring station, WQ92, is located about 1.6 km upstream of the confluent. Concentrations of all studied variables are far lower than at the upstream WQ44 NTR station (Table 10) so that it has a beneficial effect on NTR water quality. Its land use is similar to that in WQ44 except that a relatively large area, 10%, is wetland, perhaps improving water quality.

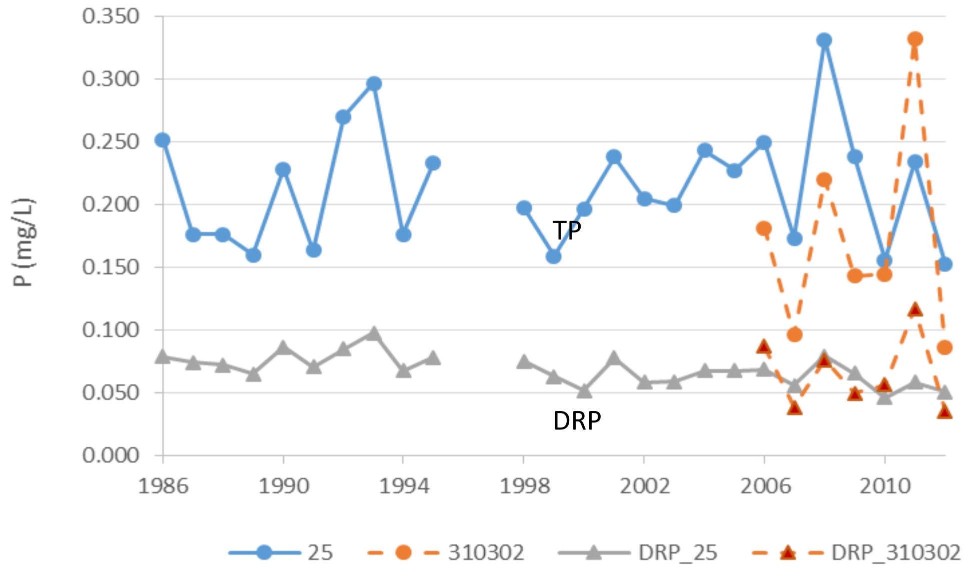
Neil Drain joins the NTR at 272.3 km. This small creek includes a small pond in the Fullarton Recreation Area 396 m upstream which has been monitored for profile data since 2005. A previous study (Nürnberg and LaZerte, 2006) determined a summer-fall TP (0.219 mg/L) and Nitrate (1.64 mg/L). Obvious signs of internal load include the high TP concentration and occasional hypoxia that occurs despite its shallowness (Section 3.5). This means that the Neil Drain contributes to the eutrophication of the NTR at and below 272.3 km.

5.1.1.3 Avon River and Lake Victoria

The Avon River joins the NTR at 265.9 km. The two GAM stations (Avon R ds and Avon R us, 265.9km) are about 7 km apart on the Avon River, one upstream (WQ310302) and one downstream (WQ25) of hyper-eutrophic Victoria Lake at Stratford. FWC-TP and FWC-DRP are quite variable. They do not show significant different concentration averages in the years 2006-2012, when both were monitored. However, upstream FWC-TP was smaller than downstream in all years but 2011 and exclusion of this year delivers a significant difference ($p<0.01$, Figure 26).

Downstream flow was more than twice that of upstream, with about 13% contributed from the Stratford WWTP (available only for 2008-2012, Table 5) at 1.2 km above that station at 18 km above the Avon River confluent with NTR. This is an unusually high amount of flow for a WWTP to contribute. 29% of the downstream nitrogen load can be attributed to the WWTP which has more than twice TN and NO32 concentration as the Avon River at WQ25. However effluent TP and TSS are below the upstream FWC (Table 10), and the elevated downstream FWC-TP at WQ25 in most monitored years can probably not be explained by effluents.

Figure 26. FWC-TP and FWC-DRP concentration upstream (310302) and downstream (25) of Victoria Lake.



It is likely that there was a large nutrient input in the past from the associated WWTPs that has now accumulated as a legacy load in the bottom sediment of the slower moving impoundment in the Avon River. This is reflected in relatively high sediment TP concentration and may lead to the incidence of internal P loading and elevated summer and fall TP concentrations.

Avon River's contribution to the NTR is large and FWC-TP and FWC-TSS are higher than at the upstream NTR station WQ44. All variables are higher than time-weighted annual averages of station WQ67 which is 2.4 km downstream of the confluent of the Avon River and NRT (Table 10). This station (1986-95) was discontinued after 1995 and cannot be directly compared to the GAM flow-weighted averages on the Avon River or upstream on the NTR (WQ44), as GAM averages always tend to be higher (Section 2.3.2).
Two smaller tributaries, Flat Creek (WQ89) and Otter Creek (WQ94) join the NTR at 263.6 and 262.4 km. Their contribution is small and probably beneficial, because all average concentrations are below that of WQ67, except for Flat Creek's TSS (Table 10).

5.1.1.4 Trout Creek with Wildwood Lake and St Marys on the main-stem

Trout Creek's (256.5km) main characteristic is a large reservoir, Wildwood Lake. There are upstream and downstream monitoring stations for both water quality and flow so that the GAM model can be applied. The upstream monitoring station (WQ66) is located at 21 km above the confluence with the NTR and the downstream station (WQ64) at 9.6 km. Different from Lake Victoria on the Avon River, Wildwood acts as a major retention facility and causes all variables to decrease, so that it improves Trout River water quality. This change is large, highly statistically significant ($p<0.0001$, all variables), and extends to spring and summer concentration averages (Table 11). In fact, TP and DRP loads remain the same (TP) or decrease (DRP) below the reservoir despite a three-fold increase in flow volume. In conclusion and as determined previously, these mass movements indicate that internal P loading in Wildwood Lake generally does not affect the Trout Creek despite occasional enhanced summer export (Nürnberg and LaZerte, 2006) and hypoxia (Figure 11).

FWCs at WQ64 in the Trout River are also significantly lower than at the NTR station St. Marys (255.9km, WQ15_45) that is 0.7 km below Trout River inflow.

There is a WWTP at 256.5km at the Town of St. Marys, with effluent concentrations exceeding that of the downstream NTR FWC of TP, but not of TN, NO32 or TSS. The whole contribution is small because its flow volume is only 3% of downstream WQ15_45. The oldest load available for St. Marys WWTP is 0.606 t/yr TP (2008) while the most recent available load of combined WQ Stations 15_45 (257km) average of 2002 was about 50 t/yr TP (EGRET: 48; GAM: 57). More recent WWTP loads have declined to <0.2 t/yr (average of 2010-2012) and TP concentrations are similar to that of the river (Table 10).

Table 11. Water quality changes around Wildwood Reservoir on Trout Creek (256.5 km)

Variable		Season	Average Values				
			NTR		Trout Creek		
			St. Marys		Downstream		Upnstream
		Stn:	WQ15_45		WQ64		WQ66
		Model	GAM		GAM		GAM
		km:	255.91		9.60		21.00
Number of years for TP:			*11*		*25*		*24*
TP	(mg/L)	Annual	0.161	***	*0.077*	***	0.224
TP		Mar-Apr	0.185		*0.067*		0.219
TP		May-Sep	0.097		*0.072*		0.179
DRP		Annual	0.073	***	0.012	***	0.118
DRP		Mar-Apr	0.094		0.015		0.104
DRP		May-Sep	0.022		0.007		0.069
TN		Annual	9.0	***	4.5	***	7.7
TN		Mar-Apr	9.0		6.0		7.2
TN		May-Sep	5.6		3.0		5.9
NO32		Annual	7.8	***	3.8	***	7.0
NO32		Mar-Apr	7.7		6.3		6.4
NO32		May-Sep	4.3		1.8		4.7
TSS		Annual	24.1	*	15.0	***	37.0
TSS		Mar-Apr	22.5		11.8		36.0
TSS		May-Sep	25.4		15.0		39.6
TP_L	(t/yr)	Annual	76		4		4
DRP_L		Annual	34		1		2
TN_L		Annual	4,044		250		132
NO32_L		Annual	3,536		207		121
TSS_L		Annual	11,773		826		669
Flow	$10^6 m^3$	Annual	450		55		17

T-test for annual FWC differences between stations located in adjacent columns: *p<0.05, ***p<0.001; ns, not significant

5.1.2 Rkm 256-233

Concentration changes from GAM station St. Marys (255.9km, WQ15_45) to Thorndale (232.8km, WQ50) are significant and decreasing ($p<0.001$) except for DRP and TSS that are increasing, but not significantly. There is a small tributary, Fish Creek; its average concentrations of TP, DRP and TSS are lower than those of the NTR just above (WQ15_45) and below (WQ50) its inflow (248.6 km), while its TN and NO32 concentrations are similar or slightly above (Table 10). Its contribution is relatively small and only 12% of the flow at the next downstream GAM station WQ50, but it could likely account, via dilution, for the approximate reduction at WQ50 by 6% of TP and 8% of DRP and TSS concentration.

5.1.3 Rkm 233-223: Fanshawe Lake

From upstream of eutrophic Fanshawe Lake (WQ50, 232.8 km) to downstream (WQ27_Clarke, 223.1 km) FWCs decrease, but these changes are significant only for DRP and TSS ($p<0.001$). FWC-TSS decreased by almost half while FWC-DRP decreased by one third. Considering that downstream WA27_Clarke includes combined data from UTRCA and CoL, with CoL tending to be higher for TP (Appendix C), TP decreases may also be “real” and caused by the retention capacity of Fanshawe reservoir on an annual basis.

However, P retention is reversed in the May-Sep period with both FWC-TP and FWC-DRP substantially elevated at the downstream station indicating an internal source (Table 12). Previous studies also found evidence of internal P load, especially because hypolimnetic TP concentration were elevated and contributed to the outflow that most of the summer happens via the bottom outlet at 8-10 m depth (Appendix E, Nürnberg and LaZerte, 2005, 2006). Those studies estimated that average internal load could be as high as one third of long-term external load to Fanshawe Lake. Internal loads are especially high in dry summers, for example in 2005 summer internal load was 3 times that of the annual external load (Section 3.5), fertilizing the reservoir and the downstream river at a time when phytoplankton can proliferate.

An approach for estimating minimum internal load in Fanshawe Lake from anoxic factors and sediment P release rates (Section 3.5) estimates 3,954 kg/summer (Table 7), which is about 4% (3.95 tonnes) of the annual load at the station below Fanshawe (91 t/yr, Table 12). Other approaches delivered 4 times higher estimates which would result in 16% of long-term downstream load.

More detailed studies, especially frequent measurements directly at the outflow (and not several km below), may be necessary to quantify the internal loading effect from Fanshawe Lake on the NTR more precisely.

There is a new WWTP in Thorndale just below WQ50 since 2012, which may influence NTR water quality in the future.

Wye Creek joins Fanshawe Lake at 226.1 km. It is monitored 2 km upstream at WQ98. Annual average concentrations are slightly below those at the surrounding GAM stations except for DRP. Because GAM results tend to be larger than simple averaged results a slightly higher DRP may indicate a DRP source. However, this influence should be marginal considering the small runoff of the small watershed area of Wye Creek of only 50.5 km^2.

Table 12. Water quality changes around Fanshawe Reservoir (233 - 223 km)

Variable		Season	Test	Average Values		
				Downstream	Wye Cr	Upstream
		Stn:		WQ27_Clarke	WQ98	WQ50
		Model:		GAM	Average	GAM
		km:		223.06	226.10	232.81
n for TP				*27*		*19*
TP	(mg/L)	Annual	ns	0.145	0.125	0.153
TP		Mar-Apr		0.142		0.173
TP		May-Sep	***	0.121		0.078
DRP		Annual	**	0.059	0.066	0.090
DRP		Mar-Apr		0.069		0.099
DRP		May-Sep	ns	0.022		0.017
TN		Annual	ns	8.0	7.2	7.8
TN		Mar-Apr		8.0		7.3
TN		May-Sep	***	5.7		4.9
NO32		Annual	ns	6.7	6.3	6.8
NO32		Mar-Apr		6.7		6.4
NO32		May-Sep	***	4.7		3.7
TSS		Annual	**	17.5	14.2	30.5
TSS		Mar-Apr		20.3		27.8
TSS		May-Sep	ns	13.8		19.7
TP_L	(t/yr)	Annual		91	small	101
DRP_L		Annual		40	small	60
TN_L		Annual		4,793	small	4,731
NO32_L		Annual		3,870	small	4,103
TSS_L		Annual		10,986	small	21,169
Flow	$10^6 m^3$	Annual		601	small	608

T-test for annual and May-Sep FWC differences between stations: **p<0.01; ns, not significant

5.1.4 Rkm 223-209.8, Medway Creek, Stoney Creek

This section is highly urbanized with two highly developed creeks and two WWTPs, Adelaide on the main stem and Granton 40.5 km up on Medway Creek.

There is no further flow gauge on the NTR above the combined flow with STR to form the TR at the fork at 209.5 km. That means that the next GAM station for comparison of flow-weighted concentrations is downstream of the fork and differences are discussed in Section 5.3.

The Adelaide WWTP effluent reaches the NTR at 217.55km and contributes 5.1% of the WQ27_Clarke TP load to the NTR at a high concentration of 0.484 mg/L. It contributes the second largest amount of TP and TSS of all WWTPs along the whole Thames River. Nonetheless, the TSS long-term average concentration is small at 5.3 mg/L.

Stoney Creek (WQ96 at 216.5km) contributions are relatively small in load and concentration. Only the long-term average TSS is high at 35 mg/L perhaps reflecting the large proportion, 17%, of urbanized catchment area.

There are two CoL monitoring stations on the main stem, Richmond at 214.4km and Dundas at 209.8. Between these stations is the influence of Medway Creek, discussed separately below.

2004-2012 time averaged annual concentrations, available for TP, NO32 and TSS, are similar at these two stations, considering that they are not flow-weighted (which makes averages less certain). They do not appear to make any unusual contributions to NTR loads.

There is a gauge at Medway Creek at 0.8 km before it flows into the NTR at 214.1 km. Medway Creek has about 14% of the flow at upstream WQ27_Clarke and presents a major tributary.

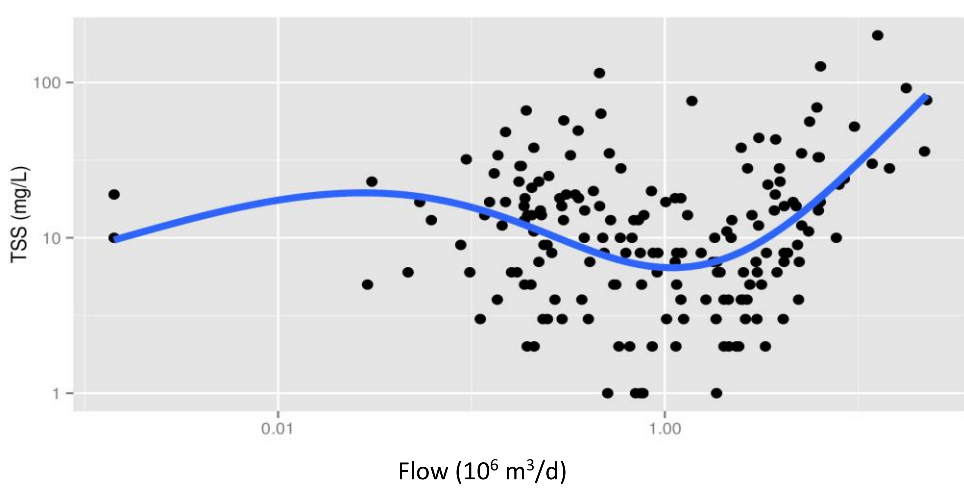
The Granton WWTP contributes high concentration (0.476 mg/L), but small loads (0.02 t/yr) of TP to Medway Creek headwater at 40.5 km upstream of the confluent. TSS is unusually high for a WWTP at 15.9 mg/L average.

FWC-TP and FWC-TSS are significantly higher in Medway Creek compared to the upstream NTR station WQ27_Clarke. NO32 is higher, but not significantly so. No TN data are available. Medway adds 17% of the TP, 18% of the NO32, and a large, 72%, of the TSS load of the upstream WQ27_Clarke station to the Thames River.

Closer inspection of (GAM) FWC-TSS of 85.1 mg/L (confirmed by EGRET estimate of 77 mg/L) on Medway Creek reveals that the GAM model inferred a complex relationship with flow, increasing dramatically at higher flow rates (Figure 27). Consequently, extrapolation to higher flows, beyond the calibration dataset, would predict large amounts of TSS. Extrapolation occurs approximately six days per year at this station (Table 2), the fourth frequent of all GAM stations. In comparison, simple time-averaged TSS is much lower at 17.3 mg/L long-term annual average.

Medway Creek frequently had some extremely high TSS concentrations (Figure 27), and the upstream WWTP has also unusually high TSS concentrations. We have no explanation for this high TSS and suggest a closer inspection of Medway Creek with respect to bank erosion and sediment resuspension. Topography cannot offer an explanation, because the overall slope of the Medway watershed is only 1.29 m/km, which is similar to other areas in the NTR and STR (UTRCA data).

Figure 27. Comparison of monitored TSS concentration with daily flow (Medway and D008)



A summary of the estimated influence of tributaries and WWTPs on the NTR is presented in Table 13.

Table 13. Evaluation of tributary and WWTP effects along the NTR for nutrients and sediment

Km	Tributary	WWTP Facility	Element	Evaluation of Effect	
				WWTP	Trib or reservoir
214.1	Medway Cr		Trib		neg: TP, DRP,TSS
214.1	Medway Cr	Granton	WWTP	neg: TP, TSS	
216.5	Stoney Cr		Trib		
217.5		Adelaide	WWTP	neg: TP	
223.1		dn Fansh	Reservoir		pos: DRP, TSS; neg: seasonal TP
226.1	Wye Cr		Trib		neg: DRP?
232.7		Thorndale	WWTP	new	
248.6	Fish Cr		Trib		pos: TP, DRP, TSS
256.5		St.Marys	WWTP	neg: TP	
256.5	Trout Cr	ds Wildwood	Reservoir		pos: all, especially TP, DRP, TSS,
256.5	Trout Cr		Trib		neg: TP, DRP, TSS
262.4	Otter Cr		Trib		pos
263.1	Flat Cr		Trib		pos
265.9	Avon R	ds Victoria	Reservoir		neg
265.9	Avon R	Stratford	WWTP	neg: TN, NO32	
265.9	Avon R		Trib		neg: TP, DRP, TSS
272.3	Neil Drain	Fullarton Pond	Trib		
274.2	Black Cr		Trib		pos
285.2		Mitchell	WWTP	improving	
286.8	Whirl Cr		Trib		pos

pos, positive; neg, negative;
Note that WWTP data are for 2000-2012 only; DRP data are never available and TN or NO32 rarely.

5.2 South Thames River Branch (STR details)

Characteristics for STR stations (between 283 km and the fork at 209 km above the mouth) are summarized in Table 14 and Figure 28A and B.

Table 14. Summary of concentrations and loads along the STR

GAM station information on the main stem are shaded across; other GAM stations refer to tributaries

Common Name (River km)	WQ-Stn	Facility or Tributary	Model/ WWTP	Years n (TP)	Annual average concentration (mg/L) TP	DRP	TN	NO32	TSS	Load (t/yr) TP	DRP	TN	NO32	TSS
York (210.2km)	York		Avg	9	0.140			5.8	25.2					
Adelaide (213.3km)	Adelaide		GAM	9	0.153			7.4	28.5	95.11			4,404	18,303
Vauxhall WWTP (214.8km)		Vauxhall	WWTP	13	0.420				6.6	2.89				46.88
Pottersburg Cr (217.7km)	Pottersbu	Pottersburg Cr	Avg	19	0.095			3.8	11.8					
Pottersburg WWTP (217.7km)		Pottersburg	WWTP	13	0.499	-	-	-	6.0	4.83				58.07
White's Br (220.3km)	Whites_51		Avg	27	0.133	0.031	6.6	5.9	19.1					
Waubuno Cr (222.5km)	97	Waubuno	GAM	8	0.087	0.063	10.2	9.2	25.9	2.91	1.97	339	307	882
Dorchester Swamp Cr (228.5km	52	Dorchester Swamp Cr	Avg	15	0.201	0.123	2.1	1.4	9.0					
Dorchester WWTP (231km)		Dorchester	WWTP	10	0.377	-	-	-	3.0	0.03				0.21
Middle Thames (240.3km)	41	Middle TR	GAM	25	0.114	0.066	10.2	9.3	20.3	15.15	9.05	1,261	1,143	2,775
Nissouri Cr (240.3km)	303402	Nissouri Creek into Middle	Avg	9	0.147	0.058	9.6	8.6	35.2					
Mud Cr (240.3km)		Mud Cr		4	0.060									
Thamesford WWTP (240.3km)		Thamesford	WWTP	11	0.213	-	-	-	3.6	0.09				1.67
Mt. Elgin WWTP (241.6km)		Mount Elgin (Subsurface)	WWTP	0						0.00				
Reynolds Cr (241.6km)	91	Reynolds1	GAM	5	0.131	0.073	7.8	6.5	35.4	8.16	4.75	464	388	2,313
	70	Reynolds2	Avg	2	0.089	0.038	6.1	5.2	11.0					
	71	Reynolds3	Avg	2	0.125	0.059	6.4	5.3	25.9					
	68	Reynolds4	Avg	12	0.129	0.048	7.1	5.9	30.0					
Ingersoll (247km)	42		GAM	20	0.134	0.053	7.3	6.2	22.5	28.71	11.43	1,502	1,287	4,872
Ingersoll WWTP (251.3km)		Ingersoll, old and new	WWTP	9	0.409	-	20.9	17.7	8.6	1.08		53.70	45.70	22.90
Halls Cr (252.5km)	310102	Halls Cr	Avg	8	0.102	0.055	7.3	6.5	20.2					
Ingersoll (253.1km)	39	Ingersoll, old and new	Avg	10	0.123	0.046	6.3	5.4	17.5					
Foldens Cr (254.9km)	69	Foldens Cr	Avg	15	0.043	0.019	7.7	7.1	10.0					
Cedar Cr (267.4km)	17	Cedar Cr1	GAM	10	0.085	0.035	6.5	5.9	17.3	2.99	1.23	215	197	620
	72-up	Cedar Cr2	Avg	10	0.108	0.060	7.0	6.1	11.1					
Woodstock (267.6km)	16	dws Pittock	GAM	20	0.156	0.057	8.0	6.8	18.4	15.63	5.74	791	670	1,907
Woodstock WWTP (269.3km)		Woodstock	WWTP	9	0.389	-	19.2	17.6	6.4	3.20		151	138	53
Woodstock Historic (270km)	38	dws Pittock	GAM	5	0.104	0.027	9.7	9.5	14.0	10.22	2.59	943	915	1,379
Innerkip (282.6km)	80	ups Pittock	GAM	20	0.179	0.136	8.8	7.7	26.7	10.79	8.33	526	455	1,695
Tavistock WWTP (282.6km)		Tavistock Lagoon	WWTP	4	0.086	-	4.7	1.8	6.9	0.05		2.71	0.97	4.11
Tavistock (282.6km)	55		Avg	15	0.156	0.056	6.6	5.5	26.1					
Tavistock Historic (282.6km)		Shakespeare	WWTP	2	0.189				3.2	0.01				0.15
Tavistock (282.7km)	310202		Avg	8	0.181	0.079	6.7	5.5	38.0					

Figure 28A. Map of STR annual average phosphorus concentration (*UTRCA*)

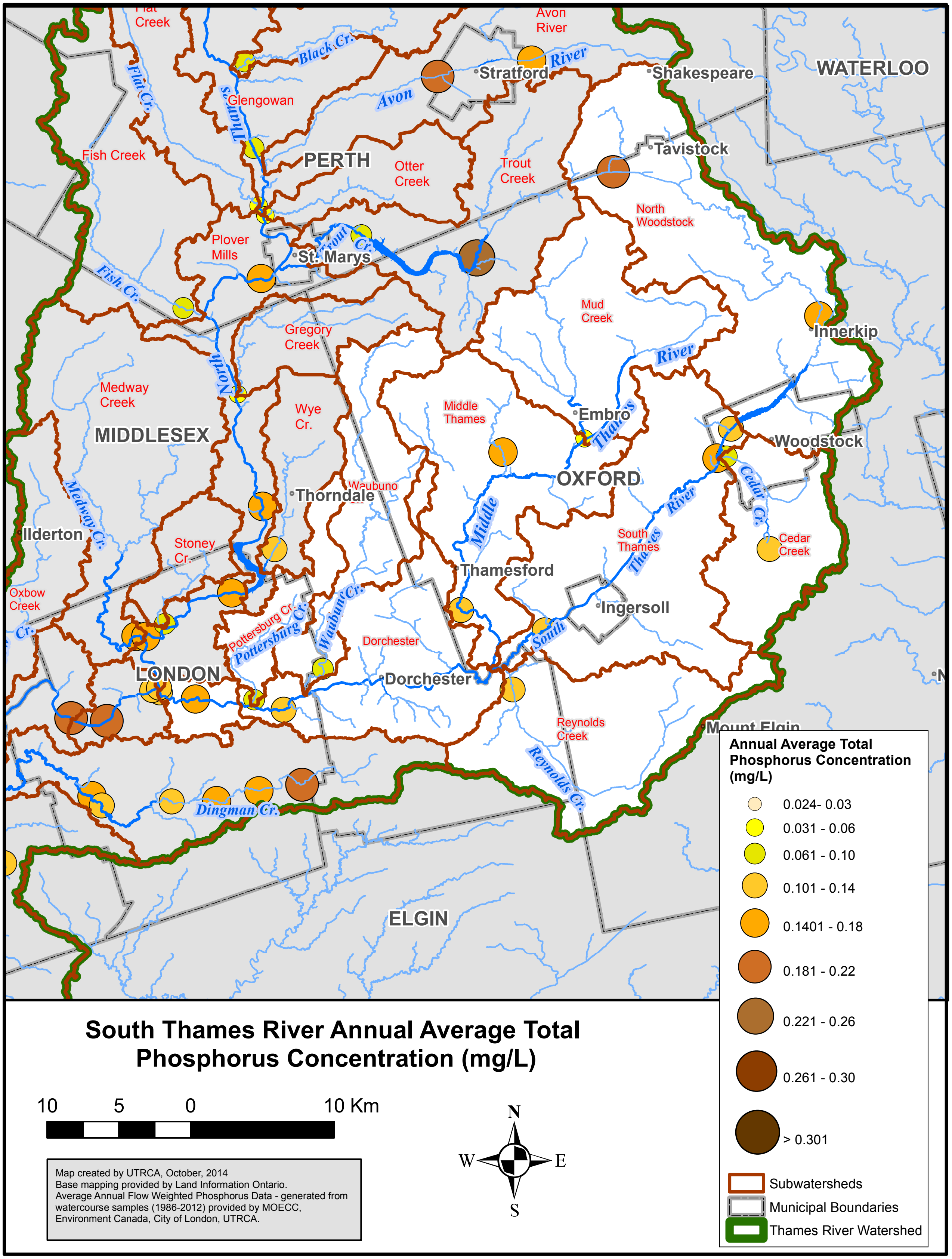
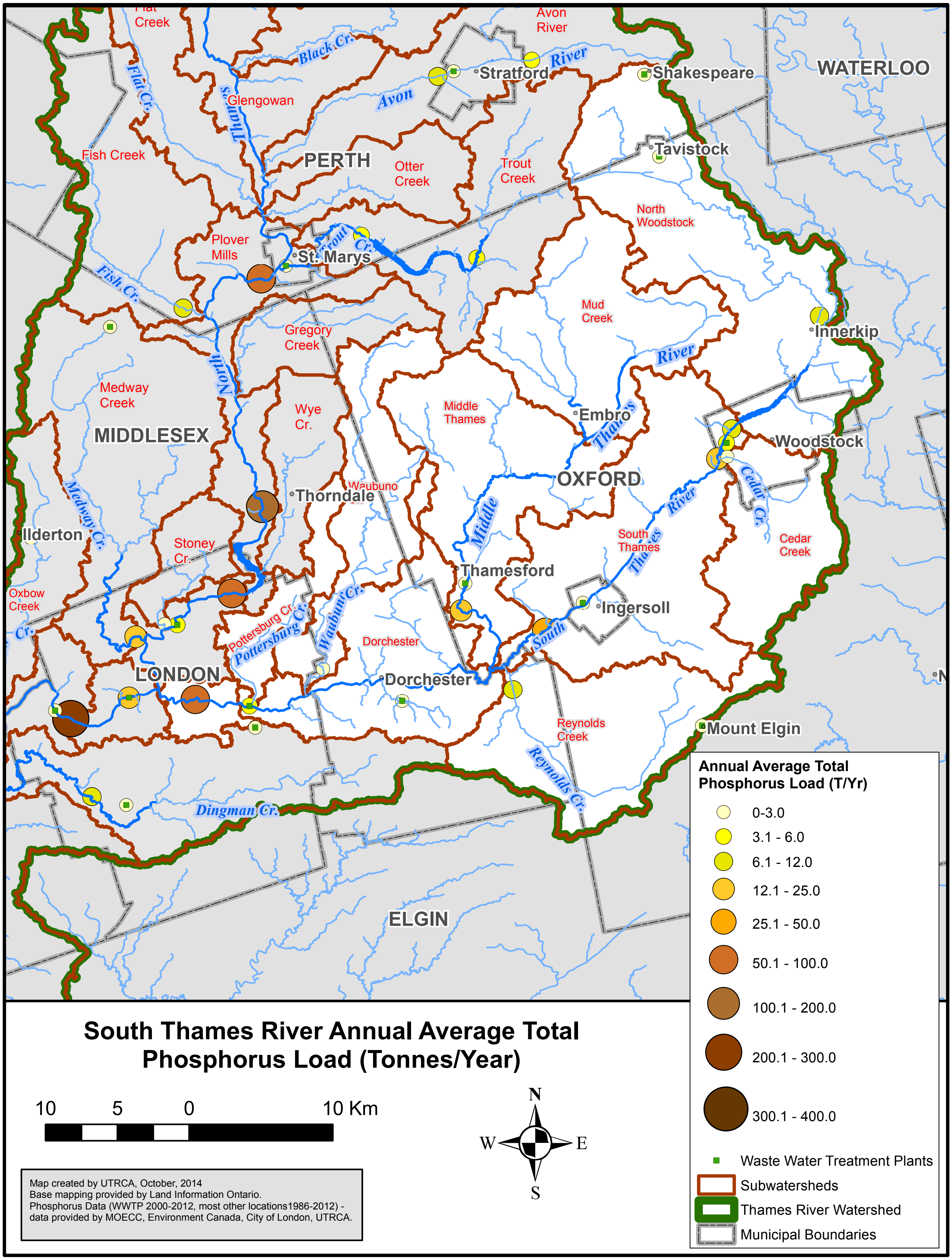


Figure 28B. Map of STR annual average phosphorus load (*UTRCA*)



The first monitoring station (WQ310202, operating since 2005) is 19 km below the start of the open channel and 282.65 km upstream of the mouth (Table 14). This is just above the inflow of Shakespeare Drain with a WWTP 34.5 km upstream operating since 2011 and the Tavistock Lagoon with effluent 28.km upstream of the tributary to the STR at 282.641km. There is a previous monitoring station (WQ55), discontinued in 2002, just upstream of these WWTP-related locations.

Annual average TP concentration (FWC values are not available for lack of a flow gauge) at the most upper STR station is 0.181 mg/L (range 0.090-0.329) of which 43% is DRP (0.079 mg/L) on average. This relatively high concentration reflects the watershed use of 84% agriculture of which is almost all (87%, the highest proportion of all sub-watersheds) tile drained (Table 4). Because its urban use is only 2.1%, the lowest proportion for the sub-watersheds of all monitoring sites, this may indicate the overriding effect of agriculture respective pollution. The nitrogen compounds are low (6.7 and 5.5 mg/L annual average of TN and NO32) compared to downstream concentrations. TSS is high at 38 mg/L perhaps reflecting the high slope of 2.47 m/km that would produce scouring of the creek bed. (This is the highest slope for sub-watersheds of all monitoring stations, UTRCA data, also Figure 4.)

The Shakespeare WWTP likely has not much influence on the STR because it is 34.5 km upstream on Shakespeare Drain and the annual TP load is small (0.01 t/yr). Average annual effluent FWC-TP for 2011 and 2012 was 0.189 mg/L which is similar to the existing STR concentration where the drain meets the STR. Tavistock WWTP is also on a tributary 28 km upstream of the confluent with STR. Effluent concentration averaged 0.360 mg/L TP during several months with available data in 2000-2003 and decreasing annual averages since then (0.134 mg/L in 2004, 0.085 in 2010, 0.073 in 2011 and 0.051 in 2012) for an average 0.086 mg/L. This concentration is well below the river FWC-TP and therefore the WWTP effluent dilutes the river concentration respective TP. Its contribution to the load is 0.05 t/yr which is only 0.05% of the load of 10.8 t/yr at 282.6 km (WQ80, D021). Neither of the two WWTPs had significant bypass events and other measured pollutant besides TP are not extreme. Consequently their influence on the STR can be described as little negative effect to slightly beneficial.

5.2.1 Rkm 283-267: Headwaters and Pittock Reservoir

Comparison of the annual averages of the most upstream station (WQ310202) with annual flow-weighted averages at WQ80, just above Pittock Reservoir at 282.64 km, suggest no difference for TP and TSS, but TN and NO32 were possibly higher at WQ80. Because the two WWTP on the tributaries between these locations cannot account for this increase as shown above, the large increase in watershed area due to the tributaries, including a doubling of urban area proportion, and slight increase in agriculture (albeit small as total area), must include the cause for elevated N-compounds.

WQ80 is the furthest upstream station on STR that can be modeled with GAM. This station was used in combination with two stations below Pittock reservoir, to evaluate the influence of a large reservoir on nutrient and pollutant concentration (Table 15). There was the discontinued station (WQ38, available data 1998-2002, GAM-modeled) close to Pittock Reservoir outflow and a long-term GAM-modeled station 3 km further downstream at 267.6 km (WQ16). At both these locations downstream of Pittock Reservoir annual average FWC are drastically and significantly decreased (TP: $p<0.01$; all other tested variables: $p<0.001$). The decrease in DRP was especially extreme from 0.136 mg/L at WQ80 to 0.057 at WQ16), as was TSS.

However, the effect varies dramatically over the season. The upstream station exhibits particularly high spring concentrations that are much reduced below the dam (WQ16, not available for WQ38). Conversely, May-Sep concentrations are quite low at WQ80 and higher at downstream WQ16. There are two explanations for these patterns. (1) The particle retention effect of the reservoir decreases TP and TSS concentrations especially during spring runoff when loads are the highest. (2) Conversely, internal loading in the summer in Pittock Reservoir increases TP and DRP in the outflow as evidenced in the previous studies (Nürnberg and LaZerte, 2006, 2005). Further, sediment constituents, possibly elevated because of WWTPs in upstream tributaries and retention by the impoundment, decreased substantially below Pittock dam (Table 15, lower panel).

The effluent of Woodstock WWTP at 369.3 km, between WQ16 and Pittock outlet, is substantial and delivers seasonally constant high concentrations and loads. WWTP loads are about 20% on average for TP, TN and NO32, but only 3% for TSS of the loads flowing through downstream WQ16. But they are similar to half of the river load in the summer (May-Sep). WWTP-TP, -TN and -NO32 are about 2.5 times higher than river FWC (i.e., FWC-TP: 0.389 mg/L compared to 0.179 at upstream WQ80 and 0.156 at WQ16, Table 15).

While there is no information on DRP for the WWTP, it is conceivable that much of effluent TP is in highly available form of DRP. The WWTP effluent, in addition to internal loading from sediments could explain the summer DRP increase from 0.032 at WQ80 to 0.050 at WQ16 as well as the TP increases.

An additional explanation for general changes in the water quality variables along this part of the river includes land use changes, as the proportion of urban areas increases at the expense of agriculture (Table 4).

Table 15. Water and sediment quality around Pittock Reservoir (267 - 283 km)

Variable		Season[1]	Average Values			Contribution of WWTP (WWTP/WQ16)
			Downstream GAM-WQ16 267.58 km	Woodstock WWTP 269.30 km	Upstream GAM-WQ80 282.64 km	
Number of years for TP:			*20*	*9*	*20*	
TP	(mg/L)	Annual**	0.156	0.389	0.179	249%
TP		Mar-Apr	0.163		0.205	
TP		May-Sep	0.160		0.082	
DRP		Annual***	0.057		0.136	
DRP		Mar-Apr	0.066		0.130	
DRP		May-Sep	0.050	n.a.	0.032	n.a.
TN		Annual***	8.0	19.2	8.8	241%
TN		Mar-Apr	8.7		8.8	
TN		May-Sep	7.1		7.1	
NO32		Annual***	6.8	17.6	7.7	261%
NO32		Mar-Apr	7.8		8.6	
NO32		May-Sep	5.9		5.5	
TSS		Annual***	18.4	6.4	26.7	35%
TSS		Mar-Apr	17.4		27.1	
TSS		May-Sep	19.6		16.4	
TP_L	(t/yr)	Annual	16	3	11	20%
DRP_L		Annual	5.7		8.3	
TN_L		Annual	791	151	526	19%
NO32_L		Annual	670	138	455	21%
TSS_L		Annual	1,907	53	1,695	3%
TP_L	(t/yr)	May-Sep	3.4	1.5	0.9	45%
DRP_L		May-Sep	1.0		0.4	
TN_L		May-Sep	155	76.1	70	49%
NO32_L		May-Sep	129	69.9	54	54%
TSS_L		May-Sep	456	23.7	192	5%
Sediment						
TP	(mg/g dry weight)		0.856		1.005	
Aluminum	(mg/g)		4.67		10.01	
Fe	(mg/g)		10.04		17.86	
LOI (% organic)			4.1%		8.5%	

[1]Paired T-test for annual FWC differences between downstream and upstream stations: **p<0.01, ***p<0.001
Sediment data from OGS - Ministry of Northern Development and Mines (Ontario Geological Survey)

5.2.2 Rkm 267-247

Along the 20 km section to the next GAM-modeled station at 247 km (WQ42), three creeks and one WWTP join the STR (Table 14). All annual average nutrient FWCs are lower at the downstream Ingersoll station and significantly so (paired t-test p<0.001), except for DRP

($p=0.09$). But FWC-TSS is significantly higher at the downstream station ($p<0.01$). It is especially elevated during summer (26.1 mg/L, compared to 19.6 mg/L at upstream WQ16).

The FWC-TP decrease of about 14%, which also occurs during spring and summer, is perhaps caused by relatively nutrient poor tributaries. Cedar Creek (267.4 km at STR) FWC-TP is 0.085 mg/L about 0.7 km upstream of the inflow, Foldens Creek 0.043 (time-weighted average of years before 2002, 4.8 km upstream), and Halls Creek 0.102 mg/L TP (5.1 km upstream).

Cedar Creek's beneficial influence is also obvious throughout the seasons, and most FWCs are smaller than at the next upstream and downstream GAM stations (not shown). The only exception is the summer TSS, which is elevated in Cedar Creek at 37.4 mg/L compared to 19.6 at upstream WQ16 and 26.1 at downstream WQ42. Another monitoring station about 10 km upstream of the confluent with STR on Cedar Creek (WQ72) shows similar concentration as the GAM-modeled station. (WQ17). Therefore, Cedar Creek positively influences water quality in the STR.

The monitoring station (WQ39, 253.1km) between Foldens and Halls Creek on STR has an annual average of 0.123 mg/L TP. WQ39 concentration averages of TN, NO32 and TSS are also slightly smaller than the GAM modeled FWC at WQ42 and WQ16. Because GAM modeled values may include higher concentrations during periods of high flow, the concentration averages at WQ39 may not be much different from those at WQ42.

The Ingersoll WWTP effluent of the new and the old facility each have high nutrient (TP, TN, and NO32) concentration averages that are about 3 times that of the downstream station WQ42, but only a third for the TSS concentration. The combined WWTP plants contribute 1.08 t/yr TP, while Cedar Cr 2.99 t/yr (the only creek in this section with available flow and calculable GAM load estimates) compared to 28.7 t/yr TP at WQ42. This shows that the WWTP contribution, although highly concentrated, is small compared to the overall load, and the dilution effect of Cedar and the other tributaries in this section override the WWTP contribution.

Because the Ingersoll WWTP overall load is small for the available recent years, less than 4% for all nutrients, its present contribution to that part of the STR can be considered minor. However, Ingersoll WWTP likely enriched the bottom sediments with organic material and nutrients in the past. Sediment analysis confirms such high concentrations (organic content, LOI, 8% and TP, 0.91 mg/g dry weight, which are about as large as upstream of Pittock Reservoir, Table 15).

5.2.3 Rkm 247-210, above the Forks

The STR is joined along the 30 km section to the next GAM-modeled station at 213 km (Adelaide, CoL) by six creeks with monitoring stations, one, Mud Creek, monitored for the Report Card effort, and six WWTP effluents (Table 14). FWCs between the two GAM stations Ingersoll (247km) and Adelaide (213.3km) are significantly different and higher at the CoL station Adelaide for TP and TSS, but not significantly different for NO32. No data are available for TN and DRP at Adelaide (213.3km).

5.2.3.1 Reynolds Creek

A major inflow is Reynolds Creek 5.5 km below Ingersoll (WQ42, 247km) at 241.6km. GAM modeled FWC at its station just 2.1 km above the inflow into STR exhibits similar FWC of TP, TN, NO32 but slightly elevated FWC of DRP and TSS compared to the closest GAM modeled station, WQ42. Its loading contribution to the STR is substantial at about 30% of the TP, TN, and NO32 load at WQ42; contributions are even higher for DRP and TSS at 42% and 48%. Some upstream TP concentration averages are available, but because they are from only two early years for two intermediate stations and similar at the most upstream station, there is no spatial trend that could convey any cause or any special pollutant sources. (Table 16, GAM values are higher because they are flow-weighted and consider high-flow periods, but are more recent.)

While there is a WWTP about 23 km upstream that uses septic fields and serves 370 people in the village of Mount Elgin, it is unlikely the source of the relatively high DRP concentration in Reynolds Creek. The WWTP is approximately 560 m away from the creek with a wetland and woodlot in between. Also, data from 3 monitoring wells that are sampled regularly show no elevated nitrates and no change from the initial samples taken at installation (Karla Young, pers. comm.).

It is unclear what is responsible for the slightly elevated FWC of DRP and TSS in Reynolds Creek and further studies are warranted.

Table 16. Reynolds Creek: annual averages along the creek stations, km upstream of confluent with STR

Station	WQ91		WQ70	WQ71	WQ68
km	2.12		3.07	11.02	23.11
Method	GAM	---------------------	Average	-------------------------------	
TP	0.131	0.111	0.089	0.125	0.129
DRP	0.073	0.060	0.038	0.059	0.048
TN	7.8	5.1	6.1	6.4	7.1
NO32	6.5	4.0	5.2	5.3	5.9
TSS	35.4	22.8	11.0	25.9	30.0
Period	2008-2012		1986-1987		1986-95,2003-4
n, years	5		2		12

5.2.3.2 Middle Thames River and Mud Creek

The Middle Thames River joins the STR about 1.3 km below Reynolds Creek at 240.3km. There is a GAM modeled station 6.9 km upstream (WQ41) and the Thamesford WWTP 9.3 km upstream of the confluent with the STR and another station on the Nissouri Creek that flows into Middle Thames River at 22.7km.

All FWC except for the N-compounds are lower than those for Reynolds WQ91 and TP and TSS are lower than at the upstream GAM Stn WQ42. However, DRP, TN and NO32 are elevated in comparison and contribute a large load to the STR. For a flow that is 59% of that at WQ42, Middle Thames loads at 6.9 km upstream of the confluences are 53% TP, 79% DRP, 84% TN 89%, NO32 and 57% TSS. Generally, spring FWC are higher than summer FWC except for TSS, which is about the same.

The Thamesford WWTP located about 2.4 km further upstream on the Middle Thames River with an average effluent TP concentration of 0.213 mg/L which is almost twice that of the Middle Thames at WQ41. Its TSS concentration is very low at 3.6 mg/L which means it would dilute the tributary with respect to TSS. Because the total TP loads are small (0.09 t/yr compared to 15.2 t/yr at WQ41) and there is only very little bypass, Thamesford WWTP cannot be considered a major pollution source to the Middle Thames River. However, no TN or NO32 data are available to estimate the contribution of the WWTP for these nutrients to the tributary.

The Nissouri Creek further upstream contributes high concentrations of TP and TSS to the Middle Thames River, while the other variables are probably similar to those determined by GAM at WQ41 (Table 14). Its small watershed contributes only a small load which the Middle Thames seems to be able to assimilate since TP and TSS are not elevated compared to the STR.

The negative effect of the Middle Thames River on the STR respective to DRP, TN, and NO32 points to non-point sources.

Mud Creek is a small creek that joins the STR at 240.27km. It has only been monitored recently within the Report Card program about 30 km upstream of confluence with the STR. 5-8 samples per year reveal large fluctuations that were simply averaged to obtain the long-term annual average of 0.060 mg/L. If this value is representative, Mud Creek does not contribute to the large TP concentration in the STR.

5.2.3.3 Dorchester WWTP and Dorchester Swamp Creek

The Dorchester WWTP is located at 231km on the STR with effluent concentration of 0.377 mg/L TP which appears to be decreasing in recent years (Figure 29). Its total TP load is small compared to that at upstream Ingersoll STR station WQ42 (0.3 t/yr TP vs 28.7 t/yr), and its TSS concentration is low at 3 mg/L so that its impact on the water quality is probably minor.

At a discontinued monitoring station (WQ52) 2 km up the Dorchester Swamp Creek, high TP and DRP concentration were monitored in four years (1998-2001), while in 11 previously monitored years annual averages were below 0.060 mg/L TP and 0.028 mg/L DRP (Figure 30).

Because of Dorchester Swamp Creek's small watershed (18 km^2) the loads should be small regardless, however it may be useful to investigate TP and DRP at this station further to determine any unconsidered P sources to the STR. This watershed is unusual for the Thames River as it is comprised of only 49% agriculture, but 12.7 % urban and a large proportion, 30%, swampy wetlands.

Figure 29. Dorchester WWTP annual average TP concentration

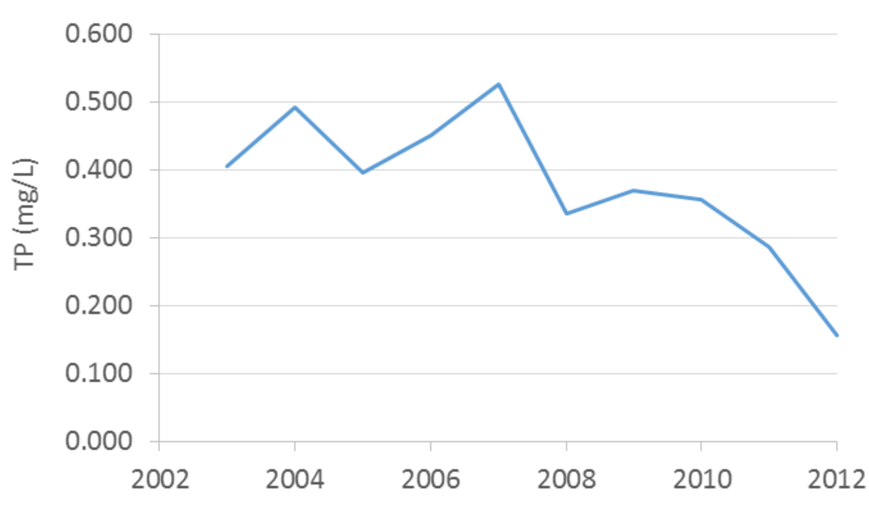
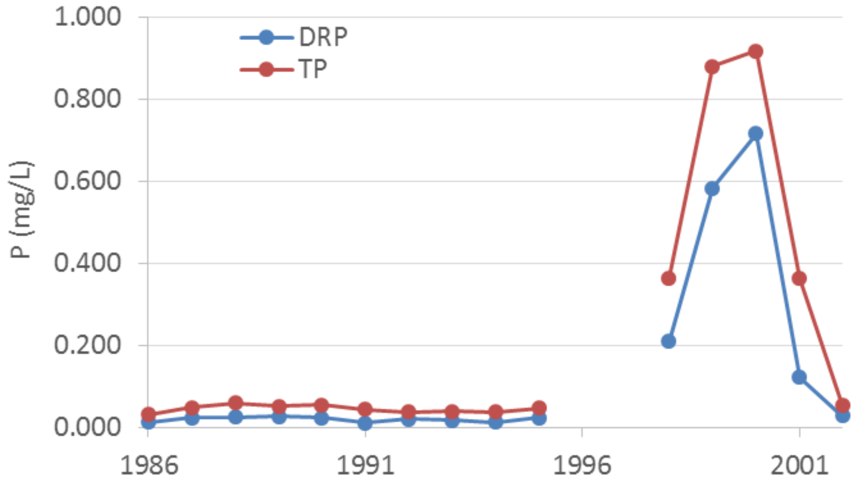


Figure 30. Dorchester Swamp Creek P compound annual averages for discontinued monitoring station WQ52



5.2.3.4 Waubuno Creek to Adelaide

Waubuno Creek joins the STR at 222.5km. Except for FWC-TP which is significantly smaller, FWC of DRP, TN, NO32 and TSS are similar (WQ97) to those from the Middle Thames, which is the closest tributary with GAM-modeled results. Also its proportionate land use is quite similar to that of the Middle Thames, except that there are no impounded areas. Because its watershed is only about a third of that of Middle Thames, its adverse influence respective DRP and the N-compounds on the STR is proportionally less.

There is a discontinued station WQ51 (1986-95) that was combined with CoL station “Whites” at STR 220.29km. Annual averages tend to be between the GAM-modeled FWC at the upstream station WQ42 and the downstream Station “Adelaide”, or slightly below. Slightly smaller values are expected when computed as simple time-based averages compared to flow-weighted GAM values.

There are two substantial WWTPs in this last reach before WQ-Adelaide. The Pottersburg WPCP and Vauxhall WPCP effluents pour directly into STR at 217.7 and 214.8km. Both have high TP effluent concentrations of almost 0.5 mg/L TP but low TSS concentration of about 7 mg/L. Annual calculated effluent TP loads are 4.8 t/yr at Pottersburg and 2.9 t/yr at Vauxhall and contribute at least 5 and 3% to the load at downstream Adelaide station of 95.1 t/yr. Because in addition there is consistent and substantial bypass (Table 6) these values are underestimates for the complete contribution of these WWTPs to the P- and possibly N-compounds in the STR. DRP and N-compounds were not measured.

The Pottersburg Creek joins STR at 217.7 just below the WWTP effluent. There is a monitoring station 0.5 km upstream (Pottersburg). Long-term average water quality is relatively good with long-term average TP of 0.095 mg/L, NO32 3.8 mg/L and TSS 11.8 mg/L. It appears that Pottersburg Creek with its small watershed area of 45 km^2 may actually benefit STR water quality with respect to nutrient load.

Adelaide at 213km is the closest station with flow gauges to the confluence at 209.5km. At this point in the Thames River watershed land use is still dominated by agriculture at 76%, but the urban percentage is higher at 9.6%. Compared to upstream GAM modeled WQ42, at Adelaide station FWC-TP has significantly increased from 0.134 to 0.153 mg/L ($p<0.01$), NO32 increased from 6.2 to 7.4 mg/L (n.s.) and TSS increased significantly from 22.5 to 28.5 mg/L ($p<0.01$). No DRP and TN data are available.

A summary of the estimated influence of tributaries and WWTPs on the STR is presented in Table 17.

Table 17. Evaluation of tributary and WWTP effects along the STR for nutrients and sediment

km	Facility or Tributary		Evaluation of Effect		
			WWTP		Tributary
				Bypass	
214.8	Vauxhall	WWTP	neg	large	
217.7	Pottersburg Cr	Trib			pos
217.7	Pottersburg	WWTP	neg	consistent	
222.5	Waubuno Cr	Trib			neg: DRP, TN, NO32
228.5	Dorchester Swamp Cr	Trib			not clear: TP, DRP
228.5	Dorchester	WWTP	no		
240.3	Middle TR	Trib			neg: DRP, TN, NO32
240.3	Nissouri Creek into Middle Thames	Trib			neg: TP, TSS
240.3	Mud Cr	Trib			pos
240.3	Thamesford	WWTP	no-p	little	
241.6	Mount Elgin, Subsurface	WWTP	no		
241.6	Reynolds	Trib			neg: DRP, TSS
251.3	Ingersoll, old and new	WWTP	no-n		
252.5	Halls Cr	Trib			no-pos
254.9	Foldens Cr	Trib			pos
267.4	Cedar Cr	Trib			pos
267.6	dws Pittock Reservoir	Trib			pos
269.3	Woodstock	WWTP	neg	some	
282.6	Tavistock Lagoon	WWTP	no-p	some	
282.7	Shakespeare	WWTP	no		

pos, positive; neg, negative; no-p; not much, leaning towards positive; no-n, not much, leaning towards negative
no, flow is so small that little effect is expected
Note that WWTP data are for 2000-2012 only; DRP data are never available and TN or NO32 rarely.

5.3 Thames River: Rkm 209-0

Characteristics for TR stations (between the fork at 209 km and the mouth) are summarized in Table 18 and Figure 31A and B.

Table 18. Summary of concentrations and loads along the TR

GAM station information on the main stem are shaded across; other GAM stations refer to tributaries

Common Name (River km)	WQ-Stn	Tributary	Facility	Model/ WWTP	Yrs n (TP)	Annual average concentration (mg/L) TP	DRP	TN	NO32	TSS	Load (t/yr) TP	DRP	TN	NO32	TSS
Mouth (0km)	Mouth			GAM	7	0.154	0.080	11.4	10.0	50.6	341.60	186.48	24,102	20,978	112,980
Tilbury WWTP (1.3km)			Tilbury	WWTP	9	0.528		10.4	6.8	7.2	0.52		11	7	7
Jeannettes Cr (3.5km)	311002	Jeannettes Cr		Avg		0.173	0.032	2.2	0.8	43.5					
Merlin WWTP (3.5km)		Foxton Dr	Merlin PV	WWTP	0	0.412		4.3	1.7	10.8					
Jacob Rd (14.8km)	308202			Avg	0	0.106	0.028	5.1	4.2	35.6					
Chatham WWTP (25km)			Chatham	WWTP	12	0.453		16.7	14.8	6.2	3.75		146	130	50
McGregor Cr (29.7km)	308102	McGregor		GAM	7	0.200	0.063	10.0	8.2	183.4	15.32	4.82	753	617	13,756
Blenheim WWTP (29.7km)		Cameron Drair	Blenheim Lagoo	WWTP	7	0.279		6.9	4.0	2.2	0.24		6	4	2
Ridgetown WWTP (29.7km)		Gawne Drain	Ridgetown	WWTP	8	0.248		10.7	13.9	5.3	0.21		7	22	5
Chatham (30.8km)	2GC1700			Avg	1	0.046	0.009	5.3	4.8						
Kent Bridge (49.7km)	305802_Kent Br			GAM	17	0.191	0.050	7.2	6.1	92.7	364.22	96.99	13,256	11,184	185,249
White Ash Cr (65km)	305702	White Ash Cr		Avg	2	0.055		4.3	3.7	29.2					
Thamesville (65.2km)	2GE1000		Thamesville	Avg	1	0.081	0.024	5.2	4.5						
Thamesville WWTP (65.2km)			Thamesville	WWTP	13	0.489		12.2	9.9	7.2	0.04		1	1	1
Fleming Cr (89.8km)	310902	Fleming Cr		Avg	2	0.161	0.106	4.7	3.8	44.7	high TP, DRP, TSS on Nov 2011				
Wardsville WWTP (93.4km)			Wardsville	WWTP	5	0.223				3.3	0.01				0
Glencoe WWTP (115.2km)		Newbiggin Cr	Glencoe	WWTP	1	0.267			9.5	4.8	0.06			2	1
Newbiggen Cr (115.2km)		Newbiggin Cr		Avg	17	0.279	0.127	7.9	6.0	58.4					
Currie Rd (127.2km)	308302			GAM	7	0.136	0.056	7.7	8.3	43.6	254.66	109.33	13,405	14,636	80,125
Giles (173.0km)	Giles			Avg		0.118			4.7	21.3					
Mt. Brydges WWTP (185km)			Mount_Brydges	WWTP	1	0.255				2.5	0.01				0
Komoka Cr (185.7km)	63	Komoka Cr		Avg		0.024	0.005	1.8	1.2	7.3					
Dingman Cr (186.5km)		29	E005	GAM	25	0.162	0.067	5.2	3.9	62.2	7.76	3.19	241.98	180.77	3,043
Dingman-Lambeth (186.5km)	Lambeth		2.1	Avg	9	0.133				33.8					
Southland WWTP (186.5km)		Dingman Cr	Southland Park	WWTP	13	0.296				6.9	0.02				1
Dingman-Dingman Dr (186.5km)	Dingman Dr.	Dingman Cr	2.2	Avg	9	0.135				32.4					
Dingman-Wellington (186.5km)	Wellington	Dingman Cr	2.3	Avg	9	0.146				35.2					
Dingman-Wel.historic (186.5km)	Wellington	Dingman Cr	2.3	Avg	11	0.143	0.050	4.7	3.6	24.8					
Dingman-Highbury (186.5km)	Highbury	Dingman Cr	2.4	Avg	9	0.156				38.4					
Dingman-Old Victoria (186.5km)	Old Victoria	Dingman Cr	2.5	Avg	9	0.182				23.1					
Th. Komoka (189.1km)	47_Komoka			Avg	21	0.140	0.065	6.4	5.5	20.5					
Komoka WWTP (189.1km)			Komoka	WWTP	4	0.140				2.8	0.03				1
Kilworth WWTP (192.1km)			Kilworth Height	WWTP	4	0.117				1.7	0.03				0
Oxbow Cr (194.2km)		Oxbow Cr		GAM	10	0.117	0.101	7.3	6.6	19.6	4.86	4.47	277	251	785
Ilderton WWTP (194.2km)		Oxbow Cr	Ilderton	WWTP	8	0.198		13.2	11.2	4.1	0.04		2	2	1
Oxford WWTP (200.9km)			Oxford	WWTP	13	0.438				3.9	1.24				11
Byron (202.2km)	Byron			GAM	27	0.202			7.2	27.6	274.47			9,088	38,871
Suspension Br (204.9km)	Springbank F(	May-Oct before 2006		Avg	9	0.184			5.6	17.0					
Greenway WWTP (207.5km)			Greenway	WWTP	12	0.409				6.2	19.12				106
Coves (208km)	Coves			Avg	9	0.254			0.8	55.5					
Wharncliffe (209.3km)	Wharncliffe			Avg	9	0.132			5.4	18.0					

Figure 31A. Map of TR annual average phosphorus concentration (*UTRCA*)

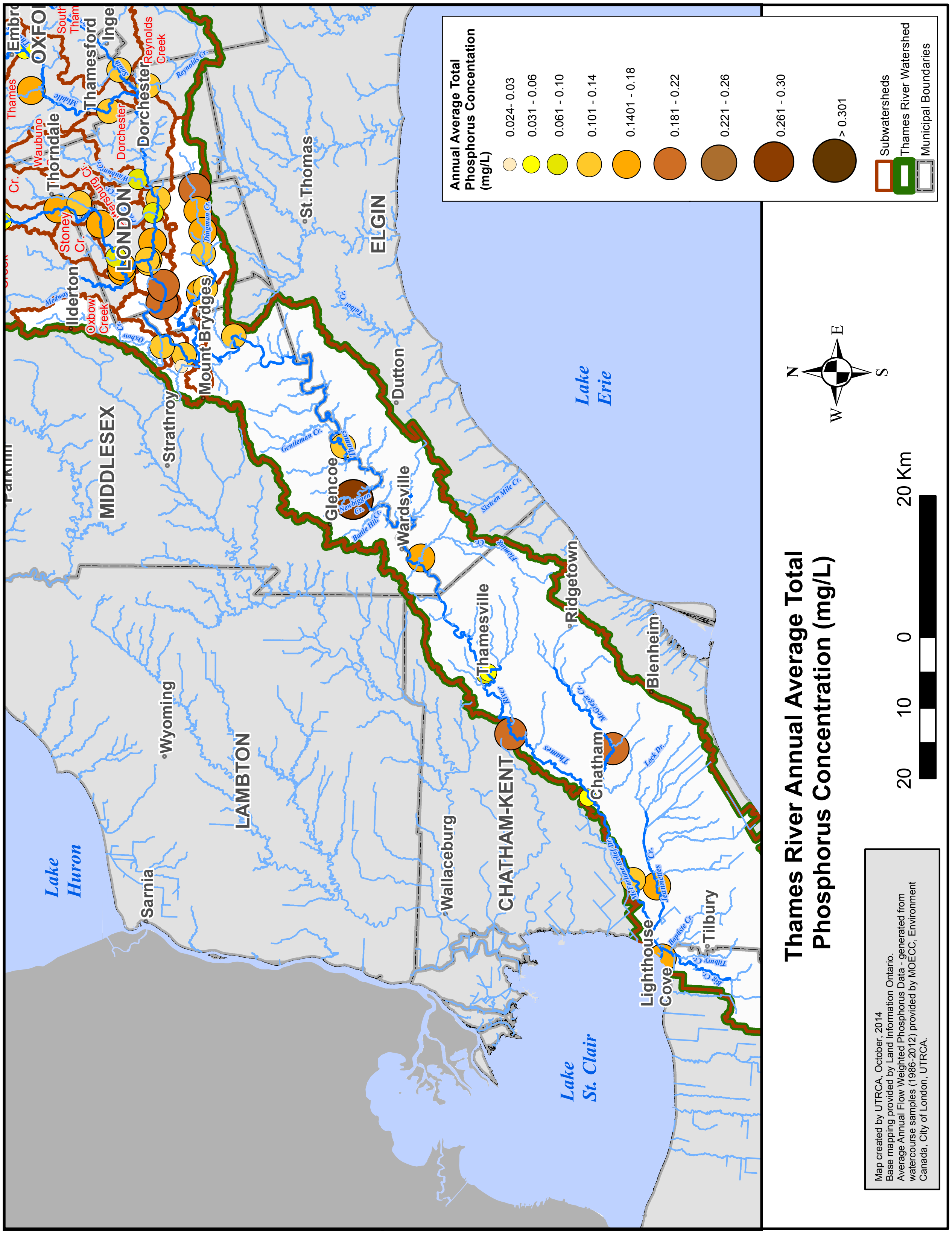
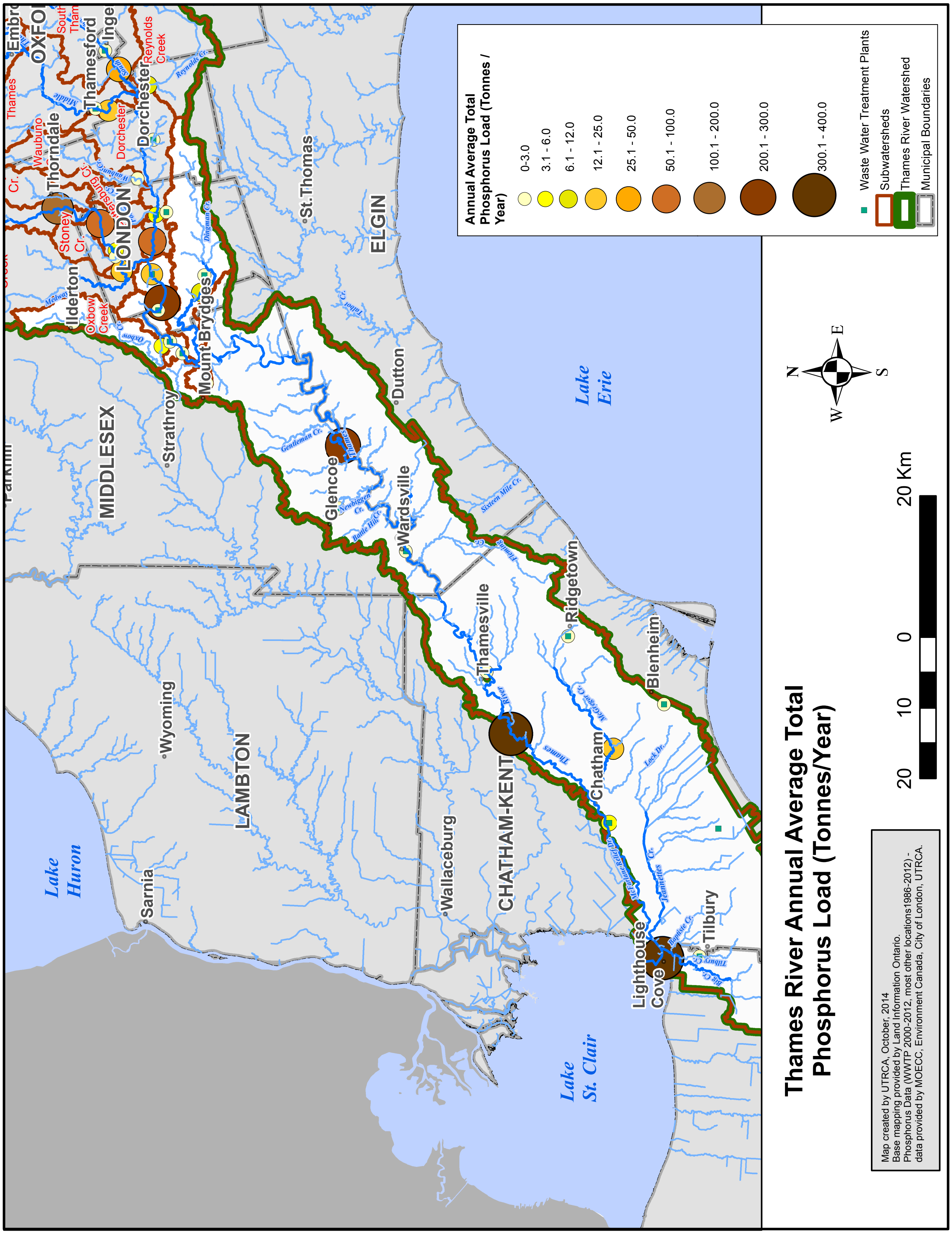


Figure 31B. Map of TR annual average phosphorus load (*UTRCA*)



5.3.1 From river branches around the Forks to Byron on TR

At 209.5km the NTR and STR meet and form the TR at the Forks. Both branches are quite similar in watershed area, land use and loads although the NTR watershed area is about 6% greater than the STR and thus contributes about 6% more to the flow at Byron (Table 1). One difference in land use is the high proportion of tile drained agricultural lands, 73%, in the NTR compared to only 47% in the STR and the slightly higher percentage of urbanization in the STR of 9.6% compared to 7.1% (comparison of Dundas with York in Table 4).

The GAM station results just upstream of the Forks are similar to each other for TP and NO32, but NTR TSS is lower compared to STR (Table 19). FWC-TP of both branches are significantly lower than of Byron. NTR FWC-TSS is also significantly lower than Byron FWC-SS.

The discrepancy in annual average FWC-TP between Byron and the upstream branch GAM sites may be explained by several load additions along the 14 km stretch below GAM station WQ27_Clark on the NTR to the fork (Section 5.1.4.) and along the 7 km stretch between the Forks and GAM station Byron on TR. There are only 4 km below the STR station Adelaide to the Forks so that contributions from this stretch are probably minor.

Table 19. Comparison of GAM model results for the two branches just upstream of the Forks with Byron station below on the TR

Location km	Station	Water-shed Area (km^2)	TP	DRP*	TN*	NO32	TSS	Flow (10^6 m^3)	Number of years (TP)
			Annual average concentration (mg/L)						
STR 213.3	Adelaide	1,346	0.153***			7.4	28.5	591	9
NTR 223.1	27_Clarke	1,427	0.145***	0.059	8.0	6.7	17.5***	601	27
TR 202.2	Byron	3,089	0.202			7.2	27.6	1,322	27
			Load (t/yr)						
STR 213.3	Adelaide	1,346	95			4,404	18,303	591	9
NTR 223.1	27_Clarke	1,427	91	40	4,793	3,870	10,986	601	27
TR 02.2	Byron	3,089	274			9,088	38,871	1,322	27

*There are no DRP and TN results available for CoL stations.
Significance levels for comparison between branch stations with Byron are indicated: *** for p<0.001

The CoL monitoring station Wharncliffe at 209.3km provides annual averages that reveal NO32 concentrations similar to the upstream NTR station *Dundas* and STR station *York*, while TP and TSS are in-between average concentrations of these stations (Table 18).

The Coves (208km, 184 m above inflow), which are former oxbow lakes of the Thames River, contribute to TP and TSS load as the long-term averages are higher than those even at downstream Byron. The three ponds that comprise the Coves had elevated TP concentration between 0.28-0.44 mg/L determined in a previous study (Nürnberg, 2007a). Long-term outflow determined in this study at the Coves monitoring station is 0.254 mg/L TP and 55.5 mg/L TSS (2004-2012). NO32 was extremely low, usually below 1 mg/L, which may indicate that an overabundance of phytoplankton was utilizing nitrate (Nürnberg, 2007b). No TN data were available to test this hypothesis.

The most obvious TP source is the Greenway WWTP at 207.5km. It is the largest WWTP on the whole Thames River and contributes a large amount of TP to the TR at Byron, 7.0 % (19.1 t/yr) of the annual load and 13% (7.6 t) of the May-Sep load at an annual average concentration of 0.409 mg/L. (These values are averages for 2001-2012 and do not include the 2000 summer extreme flows.) In addition, there was some primary bypass and large volumes of secondary bypass that is not included in the effluent load values (Table 6). TSS concentration is much lower than that of the TR at 6.2 mg/L as typical for WWTPs. No nitrogen data are available but would be important to consider.

Another CoL station is Springbank Footbridge (204.9km) on the former Spring Banks Reservoir that was operated from May-Oct until 2006. Its TP concentration is elevated at 0.184 mg/L compared to that of Wharncliffe while NO32 and TSS are similar.

5.3.2 Byron (202.2km) to Currie Rd, Dutton (127km)

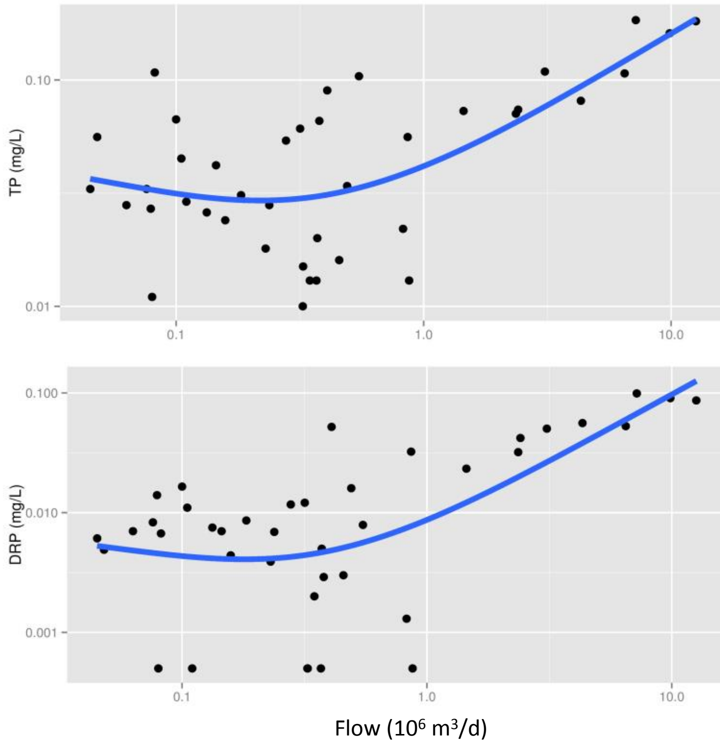
The next downstream GAM-modeled station (WQ308302, Currie Rd) is 75 km below Byron at 127.2km. Comparison of 2006-2012 data indicates significant decreases of FWC-TP ($p<0.01$) but significant increases of FWC-TSS ($p<0.01$). The FWC-NO32 increase is marginally significant ($p<0.05$). Since higher NO32 often indicates the lack of cyanobacteria proliferation in the Thames River (Nürnberg, 2007b), the changes probably indicate a better water quality with respect of phytoplankton and TP, but enhanced turbidity due to higher TSS. Flow-weighted long-term averages for DRP and TN, not available for Byron, are smaller than at upstream NTR station WQ27_Clarke ($n=6$, $p<0.05$ for DRP, $p<0.001$ for TN), also indicating improved water quality. Land use proportions are quite similar at these two stations (Byron and 308302) and cannot explain the differences (Table 4).

1.5 km below Byron is the smaller Oxford WWTP which contributes 0.5 % of Byron's TP load to the TR at a high concentration of 0.438 mg/L. TSS concentration is much lower than that of the TR at 3.9 mg/L as typical for WWTPs. Its influence on the TR is probably negligible.

Oxbow Creek joins the TR at 194.2km. There is a flow gauge and a monitoring station (WQ86) 1.9 km upstream of the confluence. FWC-TP and FWC-TSS are significantly (TP, $p<0.001$; TSS $p<0.01$) lower than at Byron positively influencing water quality by diluting the TR. FWC-NO32 was not significantly different from Byron's.

Oxbow Creek's FWC-DRP (not available for Byron) is quite high at 0.110 mg/L compared to 0.117 FWC-TP, so that FWC-DRP is 87% of FWC-TP. This could be an artefact, however, as there were not enough samples for EGRET modelling (to verify the GAM model). And there is a steeply increasing relationship between high flows and concentration for both TP and DRP (Figure 32), perhaps leading to a distortion the ratio between TP and DRP. In comparison, raw monitoring ratios of DRP versus TP (not FWC) range from 2-70%.

Figure 32. Oxbow Creek monitored TP and DRP versus flow (E008)



The Ilderton WWTP is 36 km upstream of the confluence on Oxbow Creek and may contribute to a larger variation in the DRP/TP ratio. However, water quality is likely only marginally affected because the Ilderton WWTP is quite small with little effluent (0.2 10^6 m^3) and relatively low TP (0.198 mg/L), even though TN (13.2 mg/L) and NO32 (11.2 mg/L) concentrations are relatively high.

There are two similar-sized small WWTPs along the TR, Kilworth Heights WWTP at 192.1km and Komoka WWTP at 189.1km. Average TP concentration is small at 0.117 and 0.140 respectively and TSS is below 3 mg/L. These WWTPs likely do not negatively affect water quality of the TR, although N-compounds should be considered and monitored.

A monitoring station (WQ47_Komoka) just below the inflow of Komoka WWTP yields time-averaged concentrations that are similar to the other nearby TR stations (Table 18)

5.3.2.1 Dingman Creek

Dingman Creek joins the TR just south of London at 186.5km. Along the creek are six monitoring stations between 23.6 and 45.5 km upstream of the confluence with the Thames (Table 20). One station (WQ29) with daily flow data is monitored by PWQMN and five further stations are monitored by CoL. In addition, there is a WWTP, Southland Park, with secondary treatment that serves 537 people on Dingman Creek at 30.5 km.

Annual average TP concentration (the only estimate possible for the CoL stations that lack corresponding flows) are variable between years but tend to decrease along Dingman Creek (Table 20, Figure 33). There is no spatial pattern for TSS detectable.

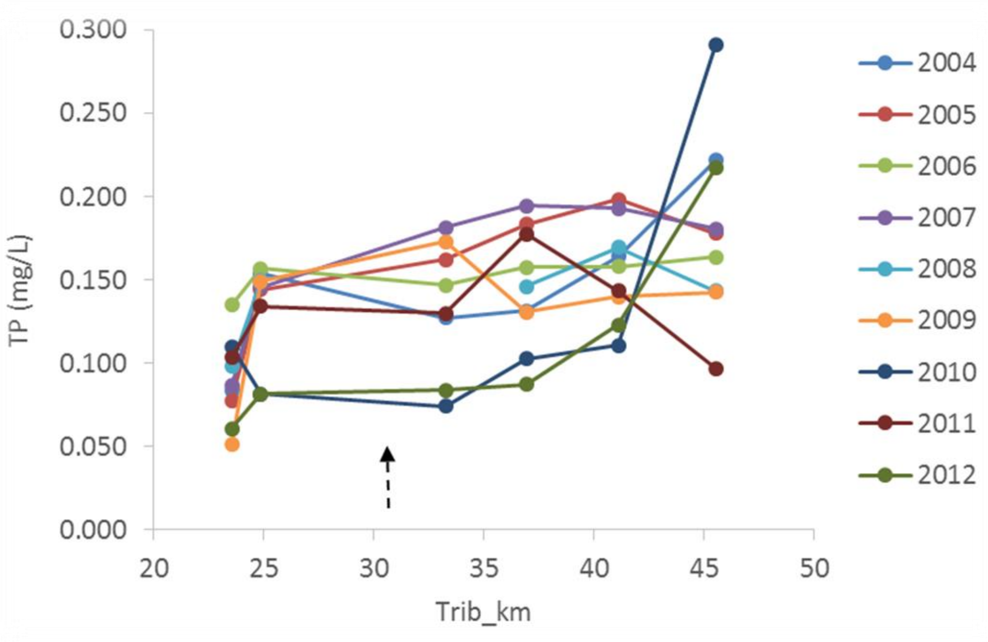
The WWTP does not have any obvious effects on either variable's average concentration despite its higher effluent TP concentration. This is understandable, because the WWTP load is only 0.3% and 0.02% respectively of the TP and TSS load 7 km downstream (at km 23.57, Stn 29 and Gauge E005, EGRET).

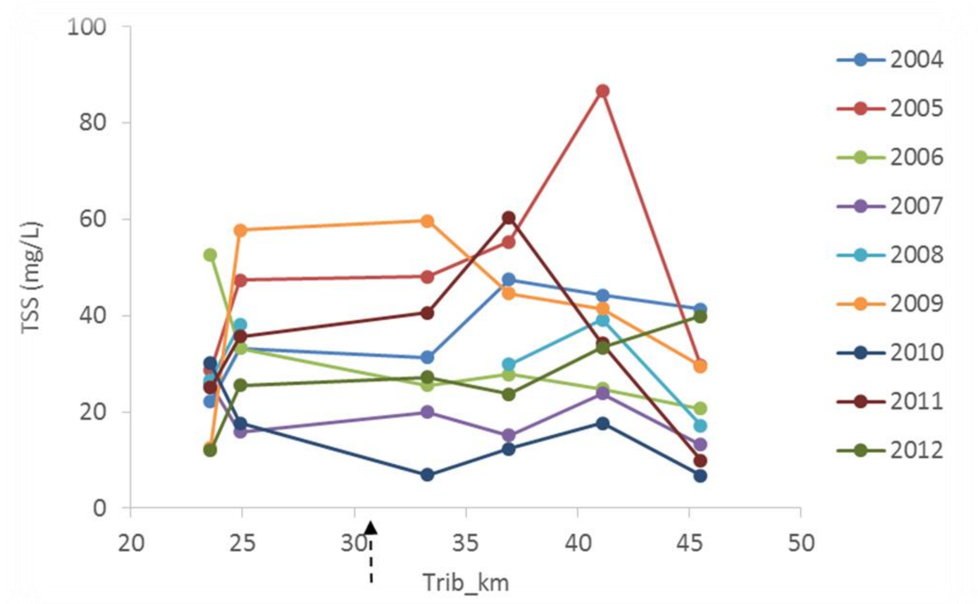
Table 20. Dingman Creek 2004-12 annual time-weighted average of TP and TSS concentration.

Agency	Site	Distance from Thames R. (km)	Concentration		Load	
			TP (mg/L)	TSS (mg/L)	TP (kg/yr)	TSS (tonnes/yr)
PWQMN	29, E005	23.57	0.090	26.2	7,876	2,594
CoL	Lambeth	24.89	0.133	33.8		
WWTP	Southland Park[1]	30.51	0.263	5.7	24	0.53
CoL	Dingman Drive	33.28	0.135	32.4		
CoL	Wellington	36.93	0.146	35.2		
PWQMN	Wellington[2]	36.93	0.143	24.8		
CoL	Highbury	41.12	0.156	38.4		
CoL	Old Victoria	45.53	0.182	23.1		

[1]WWTP concentrations refer to the effluent and are not comparable to the creek's values
[2]Discontinued monitoring station, available for: 1986-1996

Figure 33. Dingman Creek, stations of the City of London, except the most downstream station at 23.6 km of the PWQM program. The arrow indicates the location of the WWTP. TP, top panel, TSS, bottom panel

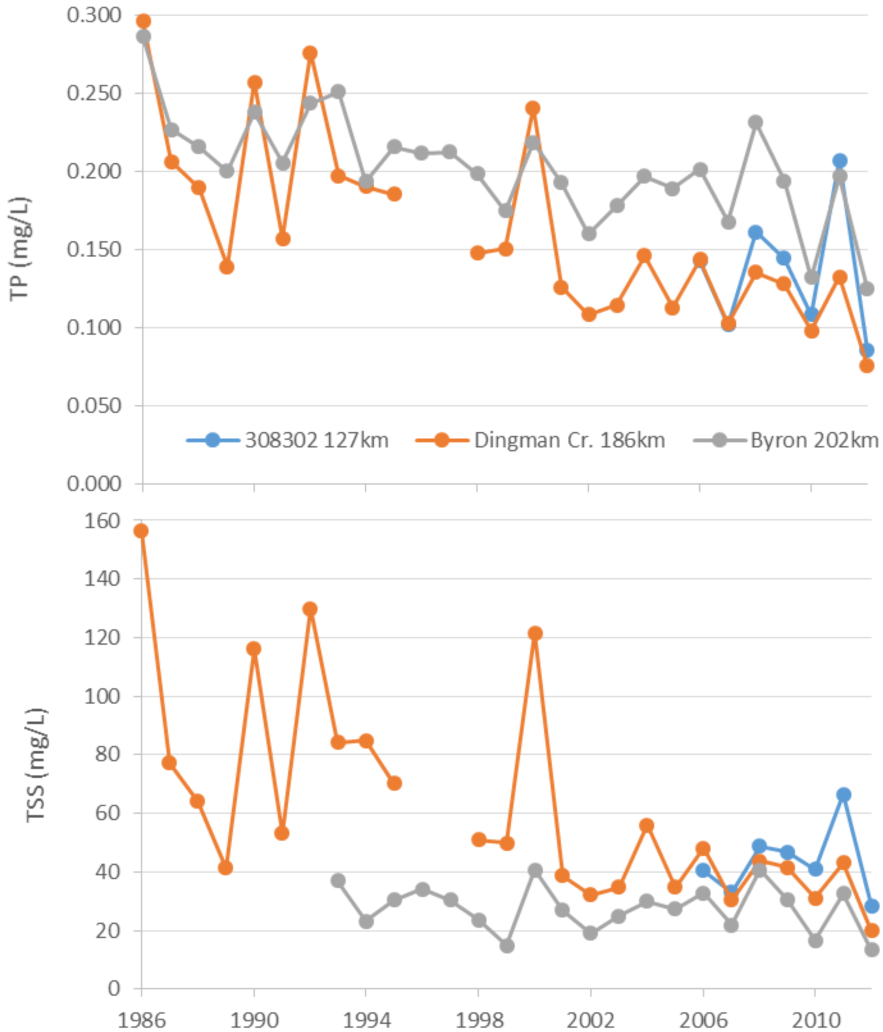




Dingman Creek probably contributes to the increasing TSS concentration along the TR. Its annual flow-weighted concentrations for WQ29 are not significantly different from those at the next main-stem GAM station (WQ308302, described above), but are much higher than those at upstream Byron station (Figure 34). Both TP and TSS are decreasing with time in the creek, which is the general trend in the main-stem TR stations for TP, but not for TSS (Table 8), which was extreme in 1986-2000 and has decreased since (Figure 34). TN and NO32 are significantly lower in Dingman Creek compared to the main stem, which could be influenced by biological uptake.

Land use is only 64% agriculture with 44% tile drained and a high 24% urban (Table 4). Potentially extended imperviousness in the proportionally large urban area may explain the high TSS values.

Figure 34. Comparison of flow-weighted TP and TSS for GAM stations above and below Dingman Creek inflow.



5.3.2.2 Dingman Creek inflow to Currie Rd, Dutton (127km)

Within one km downstream of the Dingman Creek, Komoka Creek (WQ63) joins the TR. This creek, which drains a small area of only 17.9 km^2 has extremely pristine water quality indicating almost oligotrophic conditions (Table 18). Distinguishing land use is a small agricultural area of 64% and 10% urban, and the largest recorded wetland area of 14.3%, besides 2% of open water.

However, only the wetland proportion is distinctly different from other areas with much higher concentration. Obviously, Komoka Creek has a beneficial and diluting influence on the TR.

Mount Bridges WWTP is just below the inflow of Komoka Creek at 185.0km. It is a very small plant with only 0.06 10^6 m^3 annual effluent volume and relatively low TP concentration of 0.255 mg/L in the one year of operation (2012). Only a marginal effect on the TR can be expected, if the effluent characteristics remain similar.

The CoL monitoring station Giles at 173.0km on TR exhibits lower annual average TP, NO32 and TSS concentrations than the GAM station WQ308302, which is expected when comparing non-flow-weighted temporal averages to GAM averages.

5.3.3 Lower Thames River: Currie Road, Dutton (127 km) to Mouth (0 km)

Stations in this section, starting north-west of Dutton, are monitored by LTVCA and EC.

The next GAM-modeled station below Currie Road 127km (WQ308302, discussed above) is at Kent Bridge at 49.7km (WQ305802). This is the station furthest downstream (before the inflow into Lake St. Clair) with complete daily flow (pro-rated from E003 at 65.2km, (Appendix B, Table 3) and water quality data.

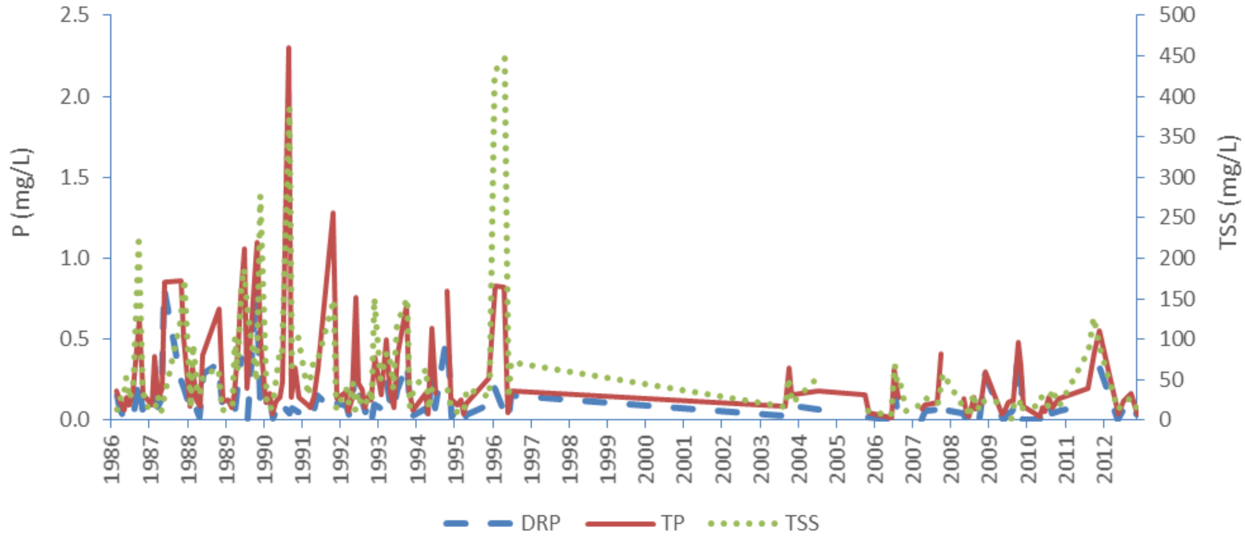
Flow-weighted average concentrations (2006-2012) change significantly along this 78 km stretch of the TR, except for DRP. TP and TSS increase extensively and TN and NO32 decrease slightly, but significantly (Table 18; FWC-TP and FWC-TN, $p<0.001$; FWC-NO32, $p<0.05$; FWC-TSS, $p<0.01$). The large increase in TP occurs especially in the spring (Mar-Apr) when all other variables tend to be larger as well.

The most obvious sources are the three tributaries, the Newbiggen Creek that enters the TR at 115.2km, Fleming Creek at 89.8km and White Ash Creek at 65km. These creeks increase in size as they drain different sized watersheds with Newbiggen the smallest (46 km^2), White Ash (76 km^2) intermediate and Fleming the largest watershed (113 km^2). Proportional land use is quite similar except that Newbiggen is the most urban (Table 4).

There are three WWTPs that could theoretically increase nutrient load, but their combined TP loads of 0.11 t/yr is only 0.01% of the Kent Bridge station (WQ305802) load. The Glencoe WWTP is on Newbiggen Creek, 12 km upstream of the confluence at 115.2km, the Wardsville WWTP effluent joins the TR at 93.4 and the Thamesville WWTP at 65.2km. The Thamesville WWTP delivers the largest average TP concentration of these three plants (0.489 mg/L), high TN and NO32 (12.2 and 9.9 mg/L) and a TSS concentration (7.2 mg/L) that is high for a WWTP, but still much below ambient concentration in the TR.

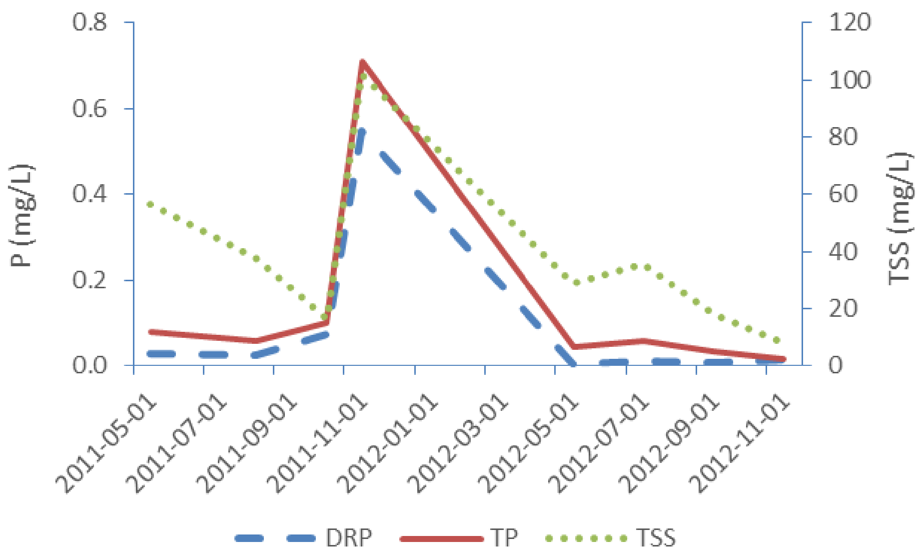
Water quality of Newbiggen Creek has been monitored for 17 years 6.9 km upstream of the confluence (WQ 307302). Quite often high TP concentration coincides with high TSS, indicating frequent sediment loading and possible bottom sediment mixing (Figure 35). Most extreme values occurred before 1996, but there are still measurements of TSS above 100 mg/L (127 mg/L) coinciding with high TP (0.555mg/L) and DRP (0.322 mg/L) concentrations (e.g., 3. Oct, 28 Nov 2011). The fact that not only TP is elevated but also DRP at high TSS indicates that it isn’t just particulate biologically-inert P fractions that are elevated, but bioavailable P as well. Further, the observation that a lower proportion of TP is DRP in the spring and higher proportions occur in the summer and fall may indicate that the source is anoxic sediment release of phosphorus and should be investigated further.

Figure 35. Newbiggen Creek TP, DRP and TSS improvement over time



Fleming Creek enters the TR at 89.8km between the two WWTP on the TR. Two years (2011-2012, WQ310902) of monitoring 2.6 km upstream from confluence suggest that while concentrations are generally lower than those of Newbiggen there are episodes of extreme TSS concentration associated with high TP and DRP concentration at the same time as in Newbiggen (e.g., 28 Nov 2011, Figure 36). It can be assumed that such episodes are more frequent in Fleming Creek as well.

Figure 36. Fleming Creek TP, DRP and TSS monitoring data

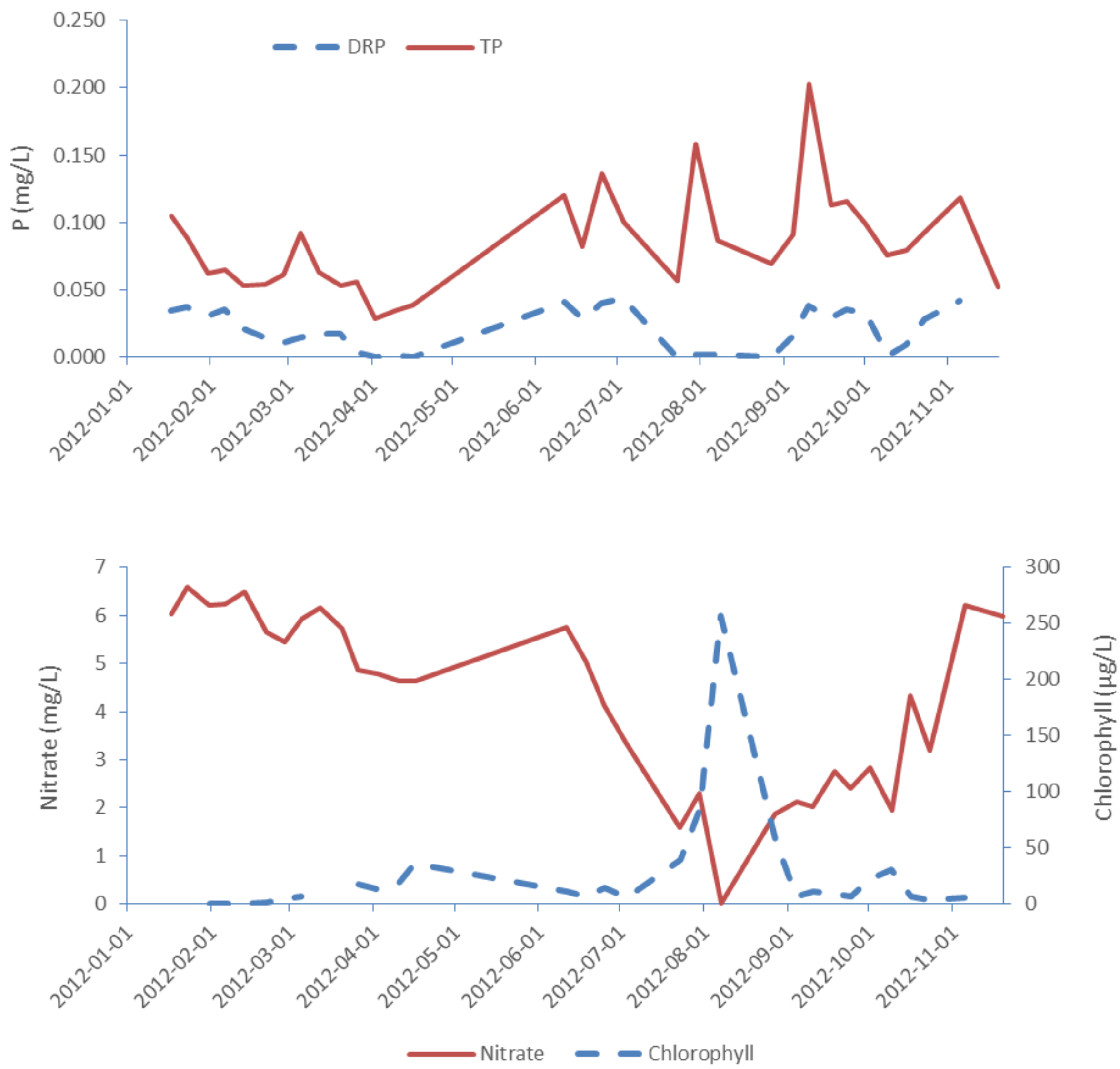


White Ash Creek enters the TR about 25 km downstream of Fleming Creek at 64.96km. Infrequent (4/year) monitoring started in 2011 (WQ305702) at 0.2 km upstream of the inflow. Similar to Fleming Creek, concentrations were highly variable with large ranges, e.g., 0.013-

0.137 mg/L TP and 4.0-68 mg/L TSS. Potential sources, such as the bottom sediments, should be investigated.

EC introduced a monitoring station in 2011 at 65.1km in Thamesville (2GE1000) at flow gauge E003, just above the inflow of White Ash Creek and below the effluent from the Thamesville WWTP. A year of high frequency sampling of about 3 times per month also reveals the extreme variability of TP and DRP in this section of the river especially in the summer and fall (Figure 37). NO32 values were similar to TN (not shown) and TSS data were not available.

Figure 37 Water quality variables at the frequently sampled EC station 2GE1000 at 65.2km



EC has been using the Thamesville 65.1km (2GE1000) station data in a comparison study to estimate loading from various rivers and creeks to Lake Erie (Dove et al., 2014). The EC study presents results for one overlapping year, 2012, that can be compared to this study (Table 21). TP load results are comparable, considering the errors involved. DRP is variable but GAM and

EC estimates are close. The EC TSS load is almost twice as high as this study's estimates. Grand River TP loads were available for 2011-2013 in the EC report and values were similar to EC loads for the Thames River (Table 21). The 2011 Grand River load is quite similar to loads estimated in this study for 2011 and further supports the conclusion that at least TP loadings are similar to EC loading estimates.

Table 21. Comparison of TP, DRP and TSS loads at Kent Bridge with EC study results at Thamesville

Study	2011	2012	2013
Number of samples per year			
EC	-	32	22
This Study	8	8	
TP (t/yr)			
EC	-	90	441
*EC-Grand River**	*712*	*91*	*416*
This study: GAM	648	133	-
This study: EGRET	671	113	-
DRP (t/yr)			
EC	-	25.9	220.8
This study: GAM	153	38	-
This study: EGRET	425	52	-
TSS (t/yr)			
EC	-	101,126	239,541
This study: GAM	481,121	60,138	-
This study: EGRET	329,835	46,384	-

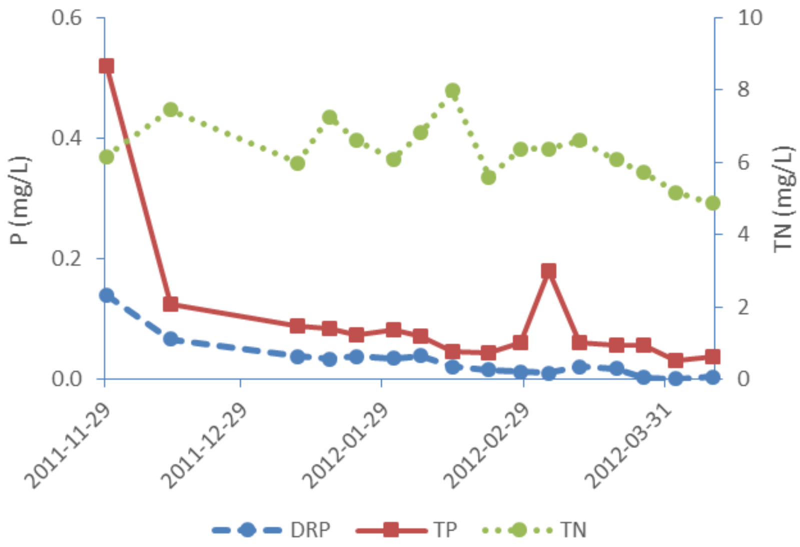
*Grand River TP load was comparable to Thames River load, but not DRP nor TSS loads.

Note that the WQ station (Kent Bridge, 49.7km, WQ305802) used in this study is below the EC station, which is located at 65.1km, close to E003. Therefore, this study uses slightly higher flows, prorated according to watershed differences (Factor 1.0299, Appendix B and Table 3).

There is no GAM-modeled station below Kent Bridge at 49.7km (WQ305802).

EC introduced another monitoring station in 2012 at 30.8km in Chatham (2GC1700), 1 km above the inflow of McGregor Creek. Frequent monitoring in the winter (Nov 2011-Apr 2012) underlines the variability of the study variables, but reveals relatively low and slightly decreasing TP and DRP and constant TN values (Figure 38). N compounds follow the tendency of decreasing concentration from spring to fall (as described before in Section 4.2). The theory of low nitrate at high phytoplankton biomass is supported by chlorophyll measurements, when the maximum (257 µg/L on 7 Aug 2012) coincides with the minimum nitrate (0.01 mg/L).

Figure 38. EC station 2GC1700 TP, DRP and TN winter monitoring (Nov 2011-Apr 2012)



A major contributor to the lower TR is McGregor Creek that enters at 29.74km. At the monitoring station and flow gauge 10.8 km upstream its flow represents 4% of the flow at the Kent Bridge, but its contribution is larger at the inflow as it drains a relatively large watershed (203 km^2). FWC-TN and FWC-NO32 are significantly higher than at Kent Bridge while the other variables are not significantly different, even though TSS long-term average is almost twice as high, a result of the large seasonal variability.

Two WWTP located on tributaries contribute to the load of McGregor Creek. The Ridgetown WWTP effluents flow into the Gawne Drain that meets McGregor at 34 km and the Blenheim WWTP effluents flow into the Cameron Creek which enters McGregor at 24 km above the confluent with the TR. Both are small plants (combined flow is less than 0.08% of the Kent Bridge flow) and probably do not contribute much to McGregor's or TR's water quality. Long-term TP averages are below 0.3 mg/L, but the Ridgetown WWTP has elevated TN and NO3 concentration of about 11 and 14 mg/L respectively.

About 5 km below McGregor Creek inflow the Chatham WWTP effluent enters the TR at 25km. This is a substantial WWTP (its flow presents 0.46% of the Kent Bridge flow) with high TP, TN and NO32 effluent concentration throughout the period of record (2000-2011, Figure 8, Figure 9). Better elimination of nutrients at this WWTP would benefit the lower TR.

Monitoring station WQ308202 at 14.8km offers occasional observations since 2002 and more consistent monthly May-Oct data starting 2006, but no flow data are available. Time-weighted averages since 2006 are comparably low (Table 18) probably because of a lack of spring and winter high flow sampling events.

Merlin PV Lagoon is a small WWTP that drains into Foxton Drain, 29km above the confluence with the TR at 3.8km, just above Jeannette Creek. The few available data suggest that it contributes high TP but little NO32 and TN to Foxton Drain.

Jeannettes Creek likely exerts a larger influence on the TR as it drains a large 330 km^2 area. Sparse 2011 and 2012 monitoring results at 9.3 km upstream (WQ311002) of its inflow at

3.47km of the TR reveal elevated time-weighted averages of TP and TSS, but low TN and NO32 concentrations (Table 18). More data is required to fully evaluate this creek.

Tilbury WWTP's effluent reaches the TR at 1.3 km above the Mouth. Even though its TP concentration is high at 0.528 and TN at 10.4 mg/L, it probably does not significantly affect the TR because of its small flow (0.05% of the NT at the Mouth). Nonetheless, the TP average of 0.528 is high for a WWTP along the TR and further elimination of nutrients is recommended.

5.3.4 Lower Thames internal load and pumping stations

As described for Newbiggen Creek (6.9km, WQ 307302) above, elevated DRP despite high TSS and the timing of summer and fall indicates that the DRP source is the anoxic sediment release of phosphorus as internal P load.

In the lower Thames waters increased DRP concentration and low DO (<3 mg/L) were measured at several occasions and hypoxic events were obvious in the fall of 2011 and summer and fall 2012.

For example, dissolved oxygen concentrations at 3 mg/L or below were measured, in the Thames River at Thamesville, Stn. 305802 on 28-Jan-91, 26-Nov-91, 26-Oct-92, 24-Jan-94, 15-Oct-12, and 19-Nov-12 (data since 1976); in the Thames River, Jacob Rd, Kent Cnty Rd 35, Prairie Siding Stn. 04001308202 and in McGregor Creek at Stn. 04001308102, Communication Rd, SE of Chatham on14-Nov-11, 13-Aug-12, 15-Oct-12, 19-Nov-12 (data since 2003).

The extent of internal loading is probably enhanced by the local practice of pumping. There are small, but widespread (about 150 including those connected directly to Lake St Clair, Jason Wintermute, pers. comm.) impoundments created by municipal pumping schemes located throughout the lower reaches of the Thames River. These pumps operate mainly in the fall and spring, when wet, but not freezing weather has produced enough runoff to fill the small reservoirs associated with the pumps. It is likely that these reservoirs accumulate nutrient-rich sediment that under warm stagnant condition in the summer and fall release phosphorus as internal loading from the bottom sediments into the overlaying water. Pumping may then distribute this phosphate-rich water throughout the water channels in the vicinity. Therefore, the pumping activity in the lower Thames River may adversely affect the water by increasing DRP.

A summary of the estimated influence of tributaries and WWTPs on the TR is presented in Table 22.

Table 22. Evaluation of tributary and WWTP effects along the TR for nutrients and sediment

km	Tributary	WWTP	Evaluation of Effect	
			WWTP	**Trib**
1.3		Tilbury	neg: TP,TN	
3.5	Jeannettes Cr			no-n
3.5	Foxton Dr	Merlin PV	no-n	
25.0		Chatham	neg: TP,TN,NO32 consistent bypass	
29.7	McGregor Cr			neg: TP, DRP,TSS
29.7	Cameron Drain	Blenheim Lagoon	no-n	
29.8	Gawne Drain	Ridgetown	neg: TN,NO32	
65.0	White Ash Cr			neg?: TP, DRP,TSS (occasionally)
65.2		Thamesville	neg: TP,TN,NO32 bypass	
89.8	Fleming Cr			neg: TP, DRP,TSS
93.4		Wardsville	no	
115.2	Newbiggen Cr	Glencoe	neg: NO32	
115.2	Newbiggin Cr			neg: TP, DRP,TSS
185.0		Mount_Brydges	no	
185.7	Komoka Cr			pos: TP, DRP, TN, NO32, TSS
186.5	Dingman Cr	Southland Park	neg: TP	
186.5	Dingman Cr			neg: TSS
189.1		Komoka	no	
192.1		Kilworth Heights	no	
194.2	Oxbow Cr			pos:TP; neg: DRP,TSS
194.2	Oxbow Cr	Ilderton	no	
200.9		Oxford	no-n	
207.5		Greenway	neg: TP lg. sec. bypass	
208.0	Coves			neg: TP,TSS

pos, positive; neg, negative; no-p; not much, leaning towards positive; no-n, not much, leaning towards negative
no, flow is so small that little effect is expected
Note that WWTP data are for 2000-2012 only; DRP data are never available and TN or NO32 rarely.

5.4 Export into Lake St. Clair

The location at the mixing zone between the Thames River and Lake St. Clair includes characteristics of both, river and lake, and present a class of environment by itself (Larson et al., 2013). The complexity including possible mechanisms and processes are indicated in Figure 39.

Water levels between lake and river are so close as to permit exchange (e.g., 3 Oct 2013: Lake St. Clair, 174.88 m asl; Chatham, 174.95 m asl; compared to the closest Thames River gauge at

Thamesville E003 at 179.27 m asl). Because of water level fluctuations there is no confined actual inflow location but it is rather a large mixing area or freshwater estuary.

Consequently, flows and loads are hard to separate and we based our export on combining known and modelled contributions from various tributaries with the gauged and monitored Thames River station closest to the mouth. There are no monitoring data available close to the mouth probably because of these inconsistent flow patterns.

Figure 39. Conception model of processes at the river/lake interface

Copied from (Larson et al., 2013)

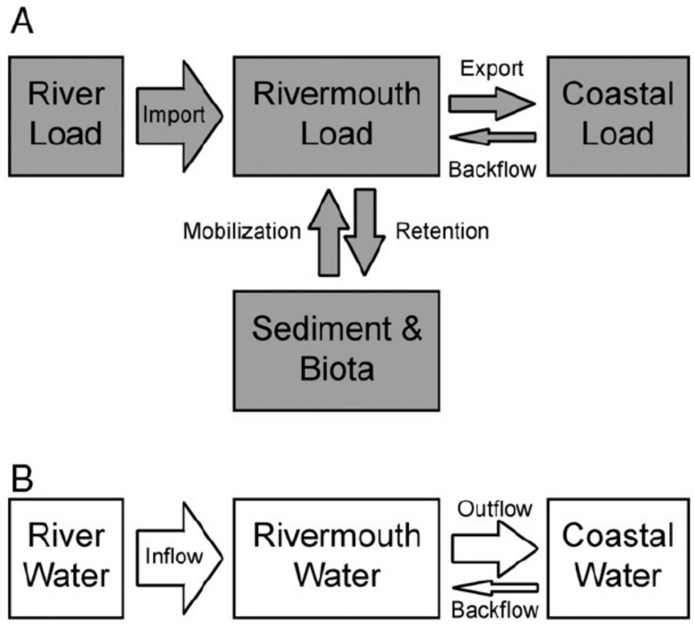


Fig. 3. Diagram, not drawn to scale, showing typical transport dynamics of sediment and nutrient loads (A) and water (B) in rivermouth ecosystems. Boxes represent storage (mass or volume, respectively), and arrows represent fluxes (mass/time or volume/time, respectively). Nutrient processing by biological uptake and deposition in the rivermouth reduces delivery to coastal waters by creating a biologically active storage environment. We generally expect longer water retention in the rivermouth to result in greater nutrient trapping and uptake, but mobilization within the rivermouth may at times exceed retention.

The station furthest downstream that has complete daily flow values and water quality data is at station “Thames River at Kent Bridge” (WQ305802, and pro-rated E003), about 50 km upstream of the mouth at Lake St. Clair. Long-term (1986-95 and 2006-12) annual average of GAM-modelled TP load is 364 t/yr without any significant change between two distinct periods with available data of 1986-95 (350 t/yr) and 2006-12 (384 t/yr). Respective annual averages estimated by EGRET are similar (overall: 384 t/yr, 1986-95: 376 t/yr, 2006-12: 395 t/yr) lending support to the GAM estimates that are used to compare with the other stations that do not have as much data as required for EGRET analysis.

McGregor Creek (WQ308102, E007, 1 km upstream of inflow into TR) flows into the Thames River at about 29.7 km upstream of the confluence so that its loading of about 15.3 t/yr (GAM, 2006-12) would also contribute to the loading into Lake St. Clair. In addition, there is direct runoff in the immediate catchment area that is mainly comprised of agriculture and contributes to nutrient load. This load was included to yield the load at the Mouth as computed by combining Kent Bridge monitoring data with pro-rated flows at Thamesville (E003) and McGregor Creek flow station (E007) pro-rated according to the watershed areas (Appendix B).

Assuming that these computations adequately estimate the loading at the Mouth, the results of these three stations can serve to highlight the changes in the Thames River before its inflow into Lake St. Clair and its likely contribution on an annual and seasonal basis for the studied water quality variables (Table 23). At present, these loads and FWCs cannot be verified, because any monitoring directly at the mouth would be compromised by the frequent exchange with St. Clair water.

Table 23. Comparison of GAM-modeled loads within 50 km of the Mouth (2006-2012)

Variable	Station*	Rkm	Load (metric tonnes)			
			Annual	Mar-Apr	May-Sep	Jan-Feb&Oct-Dec
Number of Months:			*12*	*2*	*5*	*5*
TP	Mouth	0	341.6	122.4	45.2	174.0
	McGregor	29.7	15.3	2.9	3.3	9.1
	Kent Bridge	49.7	383.9	129.4	53.5	201.0
DRP	Mouth	0	186.5	84.6	10.6	91.2
	McGregor	29.7	4.8	0.7	1.4	2.7
	Kent Bridge	49.7	97.0	23.1	10.4	63.5
TN	Mouth	0	24,102	8,678	1,959	13,464
	McGregor	29.7	753	200	80	473
	Kent Bridge	49.7	13,256	4,260	2,015	6,981
NO32	Mouth	0	20,978	7,583	1,609	11,786
	McGregor	29.7	617	159	61	398
	Kent Bridge	49.7	11,184	3,620	1,626	5,938
TSS	Mouth	0	112,980	54,052	14,950	43,979
	McGregor	29.7	13,756	2,818	1,211	9,727
	Kent Bridge	49.7	185,249	82,105	34,768	68,376
			Flow (10^6 m^3)			
Flow	Mouth	0	2,029.8	627.4	366.6	1,035.8
	McGregor	29.7	72.7	20.3	9.5	42.9
	Kent Bridge	49.7	1,983.4	616.2	363.6	1,003.6

* Note that loads at the Mouth were computed with monitoring data from Kent Bridge for lack of separate data

On an annual basis, TP load decreased along the last 50 km above the mouth, but the DRP load almost doubled indicating an increased proportion of phosphorus is biologically available phosphate (Table 23). TSS load also decreased substantially which implies that there are fewer particles available to adsorb P and more DRP can accumulate. The decrease of both TP and TSS loads, despite increased flows, is the result of enhanced settling and P retention close to the mouth where the gradient (slope) and water flow is so low that the rivers and tributaries may act as a large lake. Settling of particulate matter is a distinctive feature of freshwater deltas. In comparison, TN and NO32 loads increased proportionally along this stretch.

Increases of the bioavailable, inorganic nutrients DRP and NO32 occur mostly during winter and spring, but not in the May-Sep period. Enhanced values may be a consequence of fertilizer applications in the wet season, through the winter and followed by increased runoff in the spring, while lower values during the growing period may reflect nutrients consumption by phytoplankton and crops (less in runoff).

In comparison, others have estimated Lake Huron's TP load into the Detroit River, upstream of Lake St. Clair, as 419 (321-560) t/yr for 1994-2008 (Dolan and Chapra, 2012) which may be slightly increased at the mouth of Lake St. Clair. The Thames River likely contributes a similar or slightly higher amount. But its contribution is only one in many, because the total TP load of the Detroit River out of St. Clair at the entry to Lake Erie was estimated to be 10 fold at 3,500-4,300 t/yr in 2007 [Bruxer et al., 2011 as cited in (Lake Erie Ecosystem Priority, LEEP, 2014)]. Apparently, no other estimates are available for that area.

6 Climate predictions

According to the UTRCA climate specialist, Mark Shifflet, at time of writing there were no quantitative and detailed reliable hydrologic predictions related to climate for this area available yet. But there is certainty that the frequency and magnitude of storms will increase over time (Intergovernmental Panel on Climate Change, IPCC, "Increasing magnitudes of [global] warming increase the likelihood of severe, pervasive, and irreversible impacts", Summary for Policymakers, p.14, archived 25 June 2014, in IPCC AR5 WG2 A 2014).

Study variables that are correlated with flow are likely to increase, when storm and erratic weather events increase. In a special study that monitored concentration after storm events, maximum concentrations were observed that long-term routine sampling only provided when monitoring for a large number of years (Table 3, Section 2.3.3).

Further, WWTPs may experience more effluent flow, if there is a large amount of CSOs, and higher bypass volumes. This is probably the reason for the five-fold effluent volume in the Greenway plant in June and Aug 2000 (extreme rain events are recorded for 11 Jun and 9 Aug) that led to extremely high effluent loads affecting the long-term mean (Table 5).

Capturing adequate monitoring data will become more difficult because of the erratic nature of the future climate. Increasing temperature and extreme low flow (drought) would increase the probability of internal P loading and the proliferation of cyanobacteria, even though TSS and particulate P may be settling and be retained during these periods.

7 Recommendations

7.1 Monitoring

The consistent observation of higher flow-weighted average concentrations during the spring months underlines the importance of sampling at high flows, especially for the P-compounds and TSS. Contrarily, during summer low-flow periods P concentration can be elevated as well, but NO32 can be reduced. Water quality monitoring with an emphasis on the full range of flows helps define any relationships with flow (Appendix B). Capturing adequate monitoring data will become more difficult because of the erratic nature of the predicted future climate. Adding additional ISCO automated WQ stations (http://www.isco.com/) on the TR, NTR, and STR would capture year round water quality conditions and peak flows and help understand future changes in loadings through the system. Such samplers could also support a study to determine the effect (DO, nutrients, TSS) of pumping stations at one specific site in the lower reach of the TR.

In general, it may be most helpful to increase monitoring frequency at stations that have hydrologic support (EGRET stations in this study with long-term daily flows from EC gauges) rather than stations that do not.

Other specific sites where monitoring is recommended include presently not monitored sites in the headwaters of the NTR and STR and any tributaries with elevated nutrient or sediment concentrations. Such locations include: Medway Cr (214.1km) on the NTR (Table 13); Nissouri Cr (240.3km) into Middle Thames (240.3km) and Middle Thames itself on STR, and Waubuno Cr (222.5km) on the STR (Table 17). Further monitoring is suggested in the Coves (208km), and most of tributaries in the lower Thames that are important contributors because of their large flows: Newbiggen Cr (115.2km), Fleming Cr (89.8km) and McGregor Cr (29.7km) (Table 22). Because Jeannettes Cr (3.5km) is affected by St. Clair flow exchanges it may be less useful to monitor.

The total export to Lake St Clair is difficult to estimate directly because of the frequent exchange of water masses. It is not clear, whether a water quality measuring station closer to the mouth below Jeanette Creek would be useful, perhaps in combination with the installation of continuous thermostats and conductivity probes to determine exchange flows between the lake and river. The recently established stations by EC may try to determine such relationships.

Sediment sampling followed by the analysis of P fractions and organic content along the Thames River (in the old channel) could help determine sediment enrichment and the potential for P release and hypoxia that can lead to internal P loading, as a consequence of past nutrient loading from point (Section 7.2) and non-point sources.

7.2 Waste Water Treatment Plants

In WWTPs nitrogen compounds should be monitored in all WWTPs. Where effluent concentrations of TP and N-compounds are above those of the receiving streams, treatment should be enhanced, particularly in plants with proportionally large effluent volumes. Bypasses, especially primary bypass (e.g., Vauxhall WWTP) and overflow from combined sewers (CSO) should be minimized.

Data availability varies for WWTPs across the Thames watershed (e.g., City of London has longest data record) and are consistently available since 2000. Past WWTP loads may have affected the river and its impoundments. Extending the long-term trend in effluent concentration backwards to before 2000, there may have been a larger nutrient input that is now accumulated as a legacy load in the bottom sediment of the slower moving sections (e.g., lower Thames River) and impoundments (e.g., Fanshawe Lake on the NTR). This is reflected in relatively high sediment TP concentration and may explain the incidence of internal P loading and the elevated summer and fall TP concentrations and occasional cyanobacterial blooms. The extent could be determined by targeted analysis of bottom sediment downstream of large WWTPs.

7.3 Urban

Urban non-point sources contribute nutrients and sediment to the Thames River with highest loads during rain events. With the predicted increase in frequency and intensity of storms, urban best management practices (BMPs) should be implemented to reduce runoff volume and nutrient and sediment load. Some recommended practices include *Low Impact Development*, erosion control, minimization of fertilizer applications, and general pollution prevention (UTRCA http://thamesriver.on.ca/watershed-health/watershed-report-cards/).

7.4 Agriculture

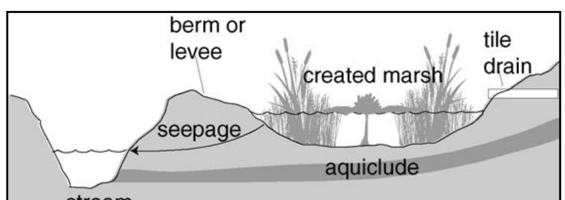
The Thames River watershed is dominated by highly productive agricultural land. With the predicted increase in intensity and frequency of runoff events agricultural best management practices are recommended that reduce nutrient runoff and soil erosion. Some recommended BMPs include soil conservation practices and efficient fertilizer application in crop production (UTRCA Watershed Report Cards). Increased intensity of runoff events has necessitated the need to develop new or modify traditional BMP's (e.g., modified erosion control structures in farm fields; pers. comm. C. Merkley, UTRCA)

Agricultural best management practices should be targeted for the time of year of most impact, which is the late winter and spring runoff period. Such recommendations are the results of many studies including other streams in South-western Ontario, the Mississippi–Ohio–Missouri Valley, and other watersheds around Lake Erie (Lake Erie Ecosystem Priority, LEEP, 2014; Ministry of Environment, 2012; Mitsch and Day, 2006).

Decreasing agricultural effects by implementing natural channel redesign of drainage ditches, artificial wetlands, increased reforestation and naturalization of marginal lands as suggested in previous UTRCA reports (UTRCA 2012) are still recommended. Five principal restoration measures are discussed in a stream restoration manual (Vought and Lacoursière, 2010): re-creation of buffer-strips, alteration of tile drainage, in-channel interventions, creation of riparian wetlands/ponds, and "daylightening" (uncovering and exposing of underground streams).

The ability of natural and constructed wetlands to remove nutrients and organic loads from farm runoff has been well demonstrated and wetland applications in head water areas as well as low-lying areas are especially effective (Mitsch and Day, 2006):

Head waters of the tributaries and the NTR and STR branches tend to have elevated nutrient and TSS concentration (Section 4.3, and 5). Wetlands are especially effective if they are located in the headwaters of small watersheds and downstream from runoff sources.



From: Mitsch and Day 2006

Low-lying areas in the Thames River watershed have varying amount of tile drainage. Studies describe *controlled tile drainage with subsurface irrigation* as an effective way to minimize nutrient export in some farming systems without compromising agricultural yield (Drury et al., 2009; Tan and Zhang, 2011).

However, the simple conversion of agricultural fields into marshland is not without controversy because of possible nutrient release from the sediments of formerly enriched areas (Steinman and

Ogdahl, 2011). Further, there are obvious sociological and economic reasons that would prevent such conversion, which usually apply to marginal lands only.

7.5 Restoration of impounded areas

In some areas, modifying or removing existing water control structures may be beneficial, as suggested by MNR legislation (Lakes and Rivers Improvement Act, http://www.e-laws.gov.on.ca/html/statutes/english/elaws_statutes_90l03_e.htm). However, many ponds and reservoirs have numerous benefits, and their multi usage includes recreation, landscape features and wild life habitat besides flood control. In addition, impounded areas retain and accumulate pollutants over time, a benefit that becomes reversed under certain conditions so that aged impoundments can become a source of nutrients. In many cases the elimination of reservoirs in the Thames River watershed would not be feasible and may adversely affect the general highlights of the area.

In such cases, the restoration of impoundment is suggested, as for Fullerton Pond (Nürnberg and LaZerte, 2006), the Cove ponds (Nürnberg, 2007a) and other off-river slow moving sections. Potential treatment includes chemical sediment capping, carp management, invasive plant management, and flow management. Restoration of the larger impounded sections, especially of reservoirs that do not stratify such as Lake Mitchell, Lake Victoria, and Lake St. Marys would also be useful and includes best management practices and further elimination of point sources. Detailed suggestions are presented in previous reports (Nürnberg and LaZerte 2005, 2006; Nürnberg, 2007b) and restoration manuals (McComas, 2003; Smayda and Packard, 1994).

8 Summary and Conclusions

(2) Land use of the whole Thames River catchment basin (5,692 km^2):

Agriculture, 80%, urban, 7.8%, area of deciduous trees, 5.1% and wetlands, 4.6%. On average 59% of the agricultural area is tile-drained.

(3) Hydrology

a. There are no long-term patterns or trends in river flows discernable. This means that any such trends in the water quality variables cannot be explained by hydrology but indicate changes in loads and concentrations.

b. Seasonal flow patterns: Flows are elevated in the spring, followed by a decrease to a late summer minimum, then upward trend over winter to spring. This means that loads would have a similar, hydrologic induced pattern and therefore, flow-weighted concentrations are best for seasonal and annual comparison and the detection of sources or losses.

c. These patterns (a. and b.) were most pronounced in the Thames River below the Fork and least in the South Thames River.

(4) Internal phosphorus load

a. Primarily in stagnant water, such as impoundments (eutrophic reservoirs) and slow moving sections of the Lower Thames River and its tributaries, and possibly downstream of WWTPs.

b. Seasonality: highest in the summer, possible also under ice.

c. Because the process constitutes the geo-chemical release of phosphate (DRP), internal loading contributes a more biologically available phosphorus than most other external sources, similar to that of fertilizer and WWTP effluent.

(5) Reservoirs and other impounded areas

a. Retain (diminish) particles, including TP and TSS over time

b. Can create inorganic P from internal P loading during warm period

(6) Waste water treatment plants

a. High nutrients but low TSS export

b. A high proportion (on average 30 to 50%) of TP is DRP and therefore biologically available (differs with treatment process, no DRP data were analyzed)

c. No consistent N data are available across the Thames River watershed. However, N-concentration can be high.

d. Especially influential at low river flows during the dry summer period

e. Evidence for much higher nutrient loads in the past, perhaps accumulated in downstream sediments.

(7) Main-stem trends

a. Temporal trends show statistically significant decreases in FWC-TP from 1986 to 2012 along the TR and STR, possibly also the NTR. Long-term changes in inorganic nutrients (NO32, DRP) are ambiguous because decreases may stem from biogenic influences (biological uptake, etc.). TN consists mainly of NO32 (median of 6453 samples is 84%). There are no annual trends in FWC-TSS.

b. Seasonal trends of flow-weighted average concentrations follow the water flow volume to various degrees. Flow dependencies are most pronounced for EGRET modeled results of FWC-TP, DRP and TSS. FWC-nitrogen variables exhibit a pattern that is occasionally flow dependent with steep decreases from spring over summer to the fall, followed by increases over the winter for all three models results.

c. GAM and EGRET flow-weighted average concentrations reveal significant spatial trends of decreasing DRP, TN and NO32 from the headwater stations of the STR and the NTR to the Forks, but this pattern is not significant for FWC-TP. FWC-TP and FWC-NO3 decrease in the lower Thames River, while FWC-DRP and FWC-TN remain relatively constant. FWC-TSS significantly decreases in the NTR, but increases in the TR towards the mouth; there is no trend in the STR.

d. Loads are highly dependent on flows. They increase from the headwaters towards the Fork, where they more than double, and further towards the mouth. Loads also follow the seasonal pattern of flows so that the highest loads occur during wet periods in the winter and spring. For example, 66% of the annual average TP load at Byron, just below the confluent of NTR and STR, occur during the 5 month period of Dec-Apr.

(8) Detailed trends along the Thames River

Land use, impounding, tributaries, WWTPs, and water flow control the concentrations and loads of the study variables to various extent (Table 10, Table 14, and Table 18) and their effect on the Thames River system is evaluated in Table 13, Table 17, and Table 22.

(9) Export into Lake St. Clair

a. Water levels between Lake St, Clair and the Thames River are close and permit exchange flows, so that loads are hard to separate. Export was computed from known and modelled contributions from various tributaries with the gauged and monitored Thames River station closest to the mouth.

b. Estimated annual export (t, metric tonnes): TP, 342 t/yr; DRP, 187 t/yr; TN, 24.1 10^3 t/yr; NO32, 21.0 10^3 t/yr; TSS, 113 10^3 yr for an annual flow of 2,030 10^6 m^3. Table 12 presents estimates for further seasons and neighboring stations.

(10) Climate change predictions

a. The increase in frequency and magnitude of storms is certain. Characteristics that are especially dependent on flow are likely to increase, i.e., TP, DRP and TSS.

b. Increasing temperature and extreme low flow (drought) increases the probability of internal P loading and the proliferation of cyanobacteria.

c. Capturing adequate monitoring data will become more difficult because of the erratic nature of the future climate.

(11) Recommendations

a. Monitoring along the river: More intense monitoring for extreme (low and high) flow conditions, especially where flow gauges are available. Extensive monitoring of bottom sediments for P-fractions and organic content in the Thames River (deep) channel to determine their potential of internal P loading by increased phosphorus release and hypoxia, especially in the vicinity of past and present point and non-point sources.

b. Monitoring load into Lake St Clair: Create a water quality measuring station closer to the mouth below Jeanette Creek. Install continuous thermostats to determine exchange flows between the lake and river. (Perhaps already attempted by EC.)

Determination of the effect of pumping stations on DO, nutrients, and TSS for at least one specific site in the lower reach of the TR.

Installation of ISCO automated WQ stations especially in combination with continuous flow measurement to capture water samples year round and during peak flows on the main Thames River stations.

c. More consistent surveillance of WWTP effluent including nitrate loads. Diminishment of bypass events and elimination of CSOs.

d. Respective spatial variation: Phosphorus loads are cumulative and contributed across the watershed with similar annual loads from NTR, STR, and about 1.5 times of those loads from TR. Implement actions to reduce nutrients in each of these 3 branches of the Thames. Where adequate monitoring exists to inform targeting, prioritize actions to subwatersheds with highest unit area TP loads.

e. Respective temporal variation: Implement actions which minimize nutrients in runoff when largest loadings occur in winter and spring high flows. Investigate causes of elevated flow-weighted concentrations throughout the year and implement actions for their reduction.

f. Non-point sources contribute a large portion of the phosphorus and sediment load annually to the Thames. Implement non-point source actions to reduce nutrient loads and concentrations across the watershed.

g. Internal loading from bottom sediments, especially in slow moving sections and impoundments contribute to phosphorus concentrations and loads. Best practices should also be targeted to larger impounded sections of the Thames to minimize internal loading over time.

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