Trent River Pulsed Discharge Study

Barlak, R.R., Deniseger H. J., Ewen, S., Henderson, G., Epps, D. and Obee, N.

Environmental Protection Division
Ministry of Environment
2080-A Labieux Rd
Nanaimo, BC V9T 6J9

November 2011

Executive Summary

The lower reaches of the Trent River are exposed to very high phosphorous loading from the Village of Cumberland (VOC) Sewage Treatment Plant. As a result, in the summer months, problematic levels of algal biomass can accumulate in the river. In an attempt to reduce these impacts, the British Columbia Ministry of Environment and the VOC joined in partnership to conduct an experimental pulsed discharge study on the Trent. During the summer of 2005, pulses were scheduled to potentially limit discharge presence in the Trent River to evening/night. Theoretically algal uptake of phosphorous would then be reduced and algal biomass in the river would decrease. At the same time, collection of nutrient data downstream was intended to give a more accurate estimate of travel time in the river. The study was in addition to the Liquid Waste Management planning process being undertaken by the VOC.

Due to underestimations in late summer travel time in the system and biological interactions of nutrients, the 2005 season was largely inconclusive in regards to the potential success of the project or travel times. Data suggested success of the pulse was primarily dependent on the travel time in Maple Lake Creek. To more accurately determine travel time as flows change throughout the summer, dye testing was implemented in the summer of 2006 through an additional partnership with the University of British Columbia. Low recovery rate of the dye suggested that a large amount of hyporheic exchange and interaction with groundwater occurs in the system. Early summer travel time in Maple Lake Creek was found to be greater than 5 days, largely due to the presence of at least one large beaver dam in the system. This indicated that, before even passing through the Trent River, the pulse dispersion was far greater than the 12 hours required to limit the pulse to evening/night. Dye testing and intensive sampling was thus terminated early in the 2006 season; water quality data collection continued by VOC in order to obtain a full data set for use in the Liquid Waste Management planning process.

Introduction

The lower reaches of the Trent River on central Vancouver Island are subject to very high phosphorous input from the Village of Cumberland (VOC) Sewage Treatment Plant (STP). A combination of this nutrient loading, river geometry and summer low flow results in excessive algal biomass accumulation in the Trent River during the summer months (Ministry of Environment (MOE), 2005 unpublished data).

Algal biomass accumulation in streams and rivers can negatively influence many stream characteristics. Odours, taste and visually unacceptable algae levels reduce the aesthetic appeal of a waterbody and impact recreational use such as swimming and fishing (Nordin, 1985). Reduced flow through gravel due to algal biomass can result in inadequate oxygen replenishment, impacting spawning areas and insects (Nordin, 1985). Aquatic insect numbers can be reduced, which in turn affects rearing salmonids. Fish movement may also be restricted in the water body.

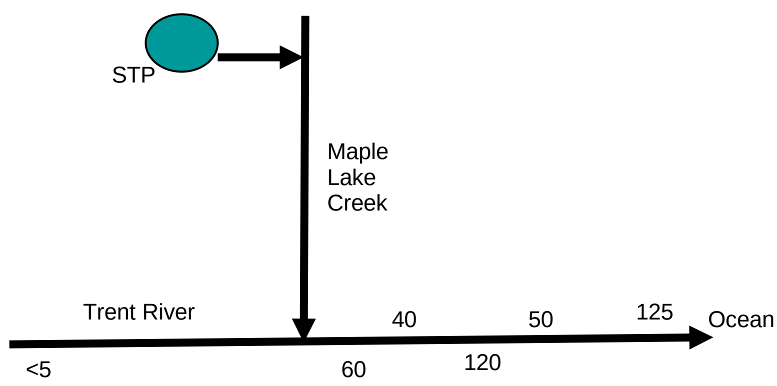
Algal growth is governed by site specific factors such as light (varies with streamside vegetation, solar arc and aspect), temperature, substrate, current, depth and season, as well as nutrient availability (Nordin, 1985). Phosphorous is normally the limiting nutrient to algal growth (Dodds *et al*. 2002; Mainstone & Parr, 2002; Elser *et al*. 1990; Nordin, 1985) and an exponential relationship exists between phosphorous and algal biomass (Bothwell, 1988; Bothwell *et al*., 1992; Nordin, 1985). Bothwell (1988) found that temperature was the major control variable that most affected algal growth in water with an abundance of phosphorus. Thus, in warmer waters there is also more opportunity for algal growth.

Coastal streams in British Columbia (BC) are typically nutrient starved and algae have evolved to be incredibly efficient at utilizing any amount of phosphorous. In addition, these streams are more susceptible to algal growth because of their substrate, gradient and usually more open stream channels. Phosphorous addition at low levels can be of benefit to these streams. However, phosphorous loading can quickly produce high and problematic amounts of algal biomass if not carefully managed (Nordin, 1985).

Due to the many site specific factors interacting with nutrient availability to determine the amount of algae in a given area, it is difficult to establish BC Water Quality Guidelines for phosphorous that are applicable province-wide. To address this issue, provincial guidelines were developed for algal quantity rather than phosphorous. Algal quantity is measured as chlorophyll α (chl α). The BC Water Quality recreation and aesthetics Guideline for chl α is 50 mg/m^2, and the protection of aquatic life Guideline for chl α is100 mg/m^2. In addition, the West Coast Region of MOE is working on developing interim objectives for total phosphorous on Vancouver Island based on Vancouver Island specific data. In draft form these objectives specify a maximum total phosphorous of 7 ug/L and a May – September average of 5 ug/L.

During the summer the Trent River can regularly have chl α levels of about 160 mg/m^2 (MOE, 2005 unpublished data), exceeding water quality Guidelines for both aesthetics and aquatic life. Normally, the Cumberland STP discharges into Maple Lake Creek (MLC) on a continuous basis. The creek enters the Trent approximately 4 km downstream of the discharge point. During the summer there is low dilution downstream of the confluence of MLC and the Trent. As such, in the summer, the area immediately downstream of MLC is subject to excessive algal growth where light is sufficient (Fig 1). From the confluence of MLC and the Trent, it is nearly 5 km to the Strait of Georgia. Much of the initial 2-3 km of the Trent downstream of MLC is steep sided, so light availability is limited. This stretch of river effectively acts as a conduit carrying the phosphorous downstream to the lower reaches of the Trent. This last 1-2 km is where the worst of the impacts occur.

Figure 1 – Diagram of the flowpath of discharge from the Cumberland STP through MLC and the Trent River, showing typical late summer/early fall chl α levels (mg/m^2) (MOE, 2005 unpublished data).



As shown in Figure 2, the initial increases in phosphorus loadings to coastal streams such as the Trent significantly increase algal biomass. Continued increases in phosphorus soon have minimal effect as other factors such as current and substrate stability limit maximum algal biomass. In the Trent River, phosphorus loadings saturate this dose-response relationship. Due to this saturation, initial decreases in phosphorus loadings would have little benefit.

Uptake of phosphorous by algae occurs mainly during the day when sufficient light is available. However, algae have evolved to be efficient phosphorous scavengers, and will store excess phosphorous even at night (Wetzel, 2001). The proximity of the MLC/Trent confluence to marine waters provides a unique set of circumstances, i.e. that travel time of discharge water from the confluence to marine waters is hours, not days. Little is known of the travel time in MLC. This study was initiated on the theory that, if the discharge could flow through the entire system at night, minimal uptake of phosphorous by algae would occur. After sunrise the algae would begin to photosynthesize utilizing the stored phosphorous. As algae are far less efficient at nitrogen utilization and no nutrient rich discharge would be flowing through the system during the day, the nitrogen would quickly take over as the limiting nutrient. As a result, excess algal biomass would not be supported. This would potentially reduce algal growth, in turn reducing impacts of algal growth on these reaches of the Trent.

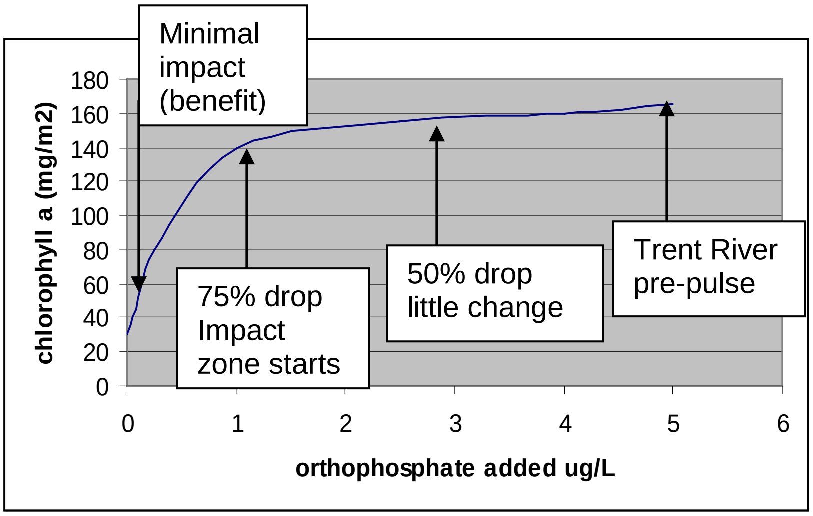


Figure 2 – Predicted impacts of decreased phosphorous loading on the Trent River based on data from Bothwell *et al.* (1992).

Maximum daily temperature variation in a fourth order stream like the Trent River can be as high as 10°C (Vannote & Sweeney, 1980; Wilcock *et al*., 1998), with lowest temperatures occurring when air temperatures are lowered (i.e. night and in the early morning) (Wetzel, 2001). Thus, potential exists for further minimizing of algal growth by coinciding the pulsed discharge with the hours of lowest temperature. Due to the lag time between change in sunlight exposure and change in temperature, the window of maximum potential decrease may be limited to only a few hours.

In an attempt to reduce the impacts of phosphorous loading on the Trent River, the BC MOE and the VOC joined in partnership to conduct an experimental pulsed discharge study on the Trent River. This study was in addition to the Liquid Waste Management Planning process being undertaken by the Village of Cumberland. At the time of the study, the Village was investigating some long-term plans to further process their sewage in a series of man-made wetlands. The theory behind this was that the plants associated with the wetlands would take up the excess phosphorus before it travels downstream to the Trent River. It was anticipated that, between the pulse study and future wetland treatment, the issue of high nutrient loading would be addressed. To the best of our knowledge, such a pulsed discharge study had not previously been conducted. To determine the plausibility of this approach, the travel time of the sewage effluent also needed to be determined.

Pulses were experimental in 2005, while nutrient data was collected to get a better estimate of travel time in the system. Inconclusive results from 2005 nutrient data led to follow up dye testing in 2006 as a means of accurately determining travel time of the sewage effluent. The determination of these values was the focus of the 2006 portion of the study. A fluorescent dye, Rhodamine WT (RWT), was used in conjunction with in-stream fluorometers. This method allowed determination of discharge rates at the time of sampling and recovery rates of the RWT.

Methods

Sample area

The study area followed the route of discharge of treated sewage from Village of Cumberland STP, through Maple Lake Creek and into the lower 5 km of the Trent River (Fig 3).

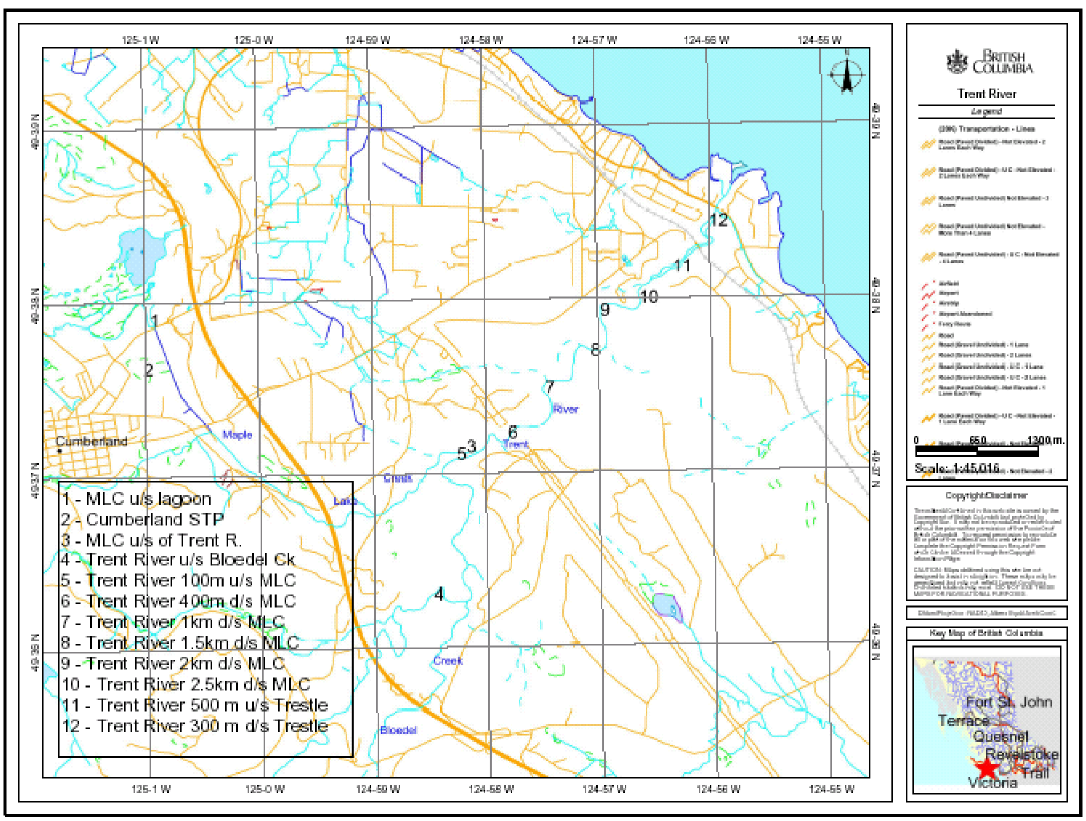


Figure 3 — Sample sites as on Maple Lake Creek, the Trent River and Bloedel Creek.

Time of travel and pulsed discharge 2005

To estimate travel time of the sewage discharge through MLC to the Trent River, the Village of Cumberland employed an automated sampler at MLC u/s Trent from June 28- August 8, 2005. Prior to July 26, 2005 (12:00 pm), the sample interval for this sampler was 6 hours; after and including this date, the sample interval was 3 hours. To obtain information on travel time through the upper 400 m of the Trent River after the confluence, an additional automated sampler was set up by MOE at 400 m d/s MLC from July 26-29th. This sampler took samples every 3 hours over a 72 hour period. Samples were transferred to 250 ml sample bottles for transport to the lab. They were then analyzed for total phosphorus at Maxxam Analytics Ltd (MOE samples) and North Island Labs (VOC samples). From July 14-25, 2005 orthophosphate was included in the VOC's analyses. This was done to obtain additional information on the ratio between orthophosphate and total phosphate.

Based on automated sampler data prior to the start of the pulse, travel time from the point of discharge to the confluence of MLC and the Trent was estimated to be 36-48 hours. Estimates placed travel time from the confluence to Georgia Strait to be less than one day and likely only a number of hours. From July 15 to September 15, treated sewage was scheduled to be discharged from the Cumberland STP to MLC in short term pulses, attempting to limit the sewage presence in the Trent River to sunset to sunrise. Discharge control was limited to the working hours of STP staff, i.e. 08:00-16:00 hours.

As per the experimental nature of this phase of the project, the first pulses were varied in amount discharged, length of discharge and intervals between discharges. These early pulses clarified that variation in flow in MLC made the pulse timing much more complicated than originally perceived. As a result, the remainder of the pulse was unified and viewed as background data vital to the future of a successfully timed pulse. From August 4 to September 15, pulses were consistently 24 hours in length and spaced three days apart.

Velocity data was collected at each sampling site throughout the sampling season for comparison of water velocity at each site. No stream width measurements were obtained, thus flow was not determined. In the absence of flow data for the Trent, real-time flow data for the neighbouring Tsolum River was used to model flow on the Trent and to determine travel time. This assumed that mean velocity was related to flow as a power model.

Time of travel and pulsed discharge 2006

2005 conclusions influencing 2006 methodology

The data collected utilizing automated samplers deployed at MLC and at the Trent River 400 m downstream of MLC were of limited use in determining the travel time of the sewage effluent through MLC into the Trent. Natural biological processes, lack of gradient and pooling of water in MLC hampered the use of phosphorus to predict sewage flow through MLC. To gain a better understanding of this issue, follow up dye testing was performed in 2006.

Dye testing and flow measurements 2006

As per the above recommendations, in 2006, travel time was determined using dye testing and flow data. The University of British Columbia (UBC) Department of Forest Resources Management Water Tracer Laboratory led the dye testing portion of this study, providing equipment, methodologies and field direction.

Sampling associated with dye testing was conducted twice in the summer of 2006. The first sample date was intended to capture the moderate to high flows and was undertaken from May 31 to June 2. The second trial intended to capture low to moderate flows, and was done from June 21 to June 27.

The amount of Rhodamine WT (RWT) to be injected was determined using

$M_{trac} = 1.7893(QC_pL_m)^{0.692}$,

where Q was assumed to be the average value of historic flows. Then, a standard solution to be used for calibration was created as follows:

a) 1.0162 g RWT added to 1 L of water (solution 1) [RWT] = 1 ppt

b) 5 mL of solution 1 added to 1 L of water (solution 2) [RWT] = 5 ppm

Final calibration of the Turner Designs Cyclops-7 (Model No: 2100-000) fluorometer was done in the field. Solutions were made, and then fluorescence was measured in a small, brown Nalgene bottle.

c) 1 mL of solution 2 added to 500 mL of stream water [RWT] = 10 ppb

d) 0.5 mL of solution 2 added to 500 mL of stream water [RWT] = 5 ppb

The discharge route was divided into two main sections for the dye testing:

1) STP discharge to MLC above the confluence with the Trent and

2) MLC/Trent confluence to the railway Trestle Bridge about 5.0 km downstream of MLC.

At the bottom of each section, fluorometric probes were installed with a Campbell Scientific CR1000 data logger (see Table 1 for injection times, locations and amounts of RWT). This way, river fluorescence data prior to die injection (“background”) could be obtained. Then, at the top of each section, the RWT was injected. One exception to this was on the second injection in MLC. Expecting a very long travel time, injection then was done as early as possible to allow more time for the dye to move downstream in hopes to actually see a breakthrough.

Probes were removed when a data check showed all dye to have passed through the system, or when no dye had passed through the system after 2 and 5 days (May 31 and June 21 sample periods respectively).

Table 1: Summary of injection and fluorometric probe location information for all trials.

Date	Time	Location	Amount (g) RWT
May 31, 2006	21:57	Trent – 200 m d/s of MLC confluence	696.41
June 1, 2006	10:15	MLC – sewage outflow	313.14
June 21, 2006	9:15	MLC – sewage outflow	107.4689
June 21, 2006	14:08	Trent – 200 m d/s of MLC confluence	647.6536

Fluorometric data analysis consisted of converting field fluorescence values into values of parts per billion (ppb) using a regression of the calibration values collected. These were then converted into g/m^3. Next, a convective dispersion model (CDM):

> $g/m^2/s = (C*x)/(\sqrt{(\pi*E*t^3)})*e^{-(x-Ut)^{\wedge}2/(4Et)}$
> ($C = g/m^3/s$, x = distance downstream (m), $E = m^2/s$, t = time since "spill" (s), U = velocity (m/s); C, U and E were optimized in Excel using an optimizing function)

was used to fit a curve to the one represented by the data collected in the field. It was optimized in Excel by minimizing the mean squared error. Recovery rate of the RWT in the Trent River was also calculated.

Flow data were collected according to Resource Inventory Commitee standards (MOE, 1998) using a Swoffer 2100 Current Velocity Meter at the old Water Survey of Canada Station (Environment Canada 2006) in the Trent River just upstream of the Old Island Highway bridge. On the second sampling date, flow data was also collected at Trent 200 m d/s MLC, Trent 1.5km d/s MLC, Trent 2.0 km d/s MLC, and Trent about 100 m u/s Trestle (Table 2). This was done to determine losing and gaining trends in the Trent system.

Assessment and Monitoring

Historical Data

Data gathered by MOE in MLC and the Trent from 2002-2004 were compiled and summarized in Appendix 1 for purposes of comparison and reference.

Sites

In 2005, thirteen monitoring sites (Table 2, Fig 3) representing STP influenced and non-influenced areas were determined in early June. Initial background data were collected by the MOE from the monitoring sites on June 22 and July 5/6 prior to the start of the pulsed discharge. After this, data was collected approximately every three weeks (July 26, Aug 23/24 and Sept 13/15) until the end of the pulsed discharge period (Sept 15) (Table 3). Twelve of the monitoring sites were sampled for water chemistry parameters, nine sites were sampled for chl α and four sites were sampled for algal taxonomy (Table 2) on each sample date. All samples were shipped on ice in coolers for overnight transport to the applicable laboratory (below). Three photos were taken at each site (upstream, actual site, downstream).

In 2006, due to budget constraints, the number of sample sites for water chemistry and chl α were reduced (Table 2). Sites to remove were chosen based on data similarities, removing only sites that had very similar results to other sites. Samples were collected May 15, June 5, June 27, July 31, August 21 and September 10, 2006.

Table 2 – Summary of sample sites in the Trent River Pulsed Discharge study. "05" indicates sampled in 2005 only, while "05/06" indicates sampled in both 2005/2006.

Name	**EMS Site**	**Chl α**	**Auto-mated sampler**	**TSS and BOD**	**Water Quality**
MLC @ Cumberland Rd	0140120				05/06
Cumberland STP	E100753			05/06	05/06
MLC u/s of Trent R.	0140124		05		05/06
Trent River u/s Bloedel Ck	E259743	05/06			05/06
Trent River 100m u/s MLC	0127581	05/06			05/06
Trent River 400m d/s MLC	E227350	05/06	05		05/06
Trent River 1km d/s MLC	E259739	05			05
Trent River 1.5km d/s MLC	E259740	05/06			05/06
Trent River 2km d/s MLC	E259741	05			05
Trent River 2.5km d/s MLC	E259742	05			05
Trent River 500 m u/s Trestle	E259744	05/06			05/06
Trent River 300 m d/s Trestle	E259745	05			05

Water Quality

Samples for nutrient analysis were taken using a clean 250 ml plastic sample bottle per site. These samples were analysed for total phosphorus, orthophosphate, ammonia and nitrate+nitrite. A 1L clean, plastic sample bottle was used for general chemistry sample collection, and the samples were analysed for pH, turbidity and true colour. All samples were analysed according to the Provincial Laboratory Standards Manual (MOE, 2005) at Maxxam Analytics Ltd.

Water for bacteriological analysis was sampled using one 250 ml sterilized plastic sample bottle per site. JR Labs (2005) and Cantest Labs (2006) analysed the samples for fecal coliforms and *Escherichia coli* as per Provincial laboratory standards.

Chlorophyll α

Algal samples for chl α analysis were collected from both natural and artificial substrates (Table 2). Natural substrates were sampled for standing crop as measured by chlorophyll α. With artificial substrates however, it should be recognized that the initial exposure period should not be used for standing crop chlorophyll α comparisons. Rather, the initial exposure period represents accrual or the rate of accumulation of algal biomass. Natural substrates do pose limitations in that natural variables such as substrate stability and disturbance are difficult to control. Natural substrates were used for comparison with historical data and because the provincial Guidelines for chlorophyll α are based on natural substrates. Artificial substrates provide more control over experimental variables such as duration of exposure and consistent substrate type, but can be damaged by vandals or storm events. All samples were sent to Maxxam Analytics Ltd. for analysis according to Provincial laboratory standards.

Natural substrate: Three (28.26 cm^2) 6 cm diameter circles of algae were scraped off three individual river bed rocks using a toothbrush, and transferred into three individual tissue cups. The algae were then filtered using a chl α pump, and the filters placed in Petri dishes.

Artificial substrates: At each site, on June 6/7, 2005 and May 15, 2006 one Styrofoam block (20 cm x 40 cm) was mounted on the same size cement block and placed in the river bed. Mid-May is the earliest sampling can safely occur due to high river levels in spring. On each sampling day, three 9.61 cm^2 (3.5 cm diameter) cores were taken out of each Styrofoam block and placed in a tissue cup for shipment to the lab.

In 2005, at the Trent 400 m d/s MLC site, the original block was vandalized within the first week of being set out. In order to predict the missing values, duplicate blocks were set out at this site and three others throughout the system, two weeks after the original blocks were placed (Table 3). Calculating the average log (ratio) of the difference between the original and duplicate block chl α resulted in a value of 0.21. This indicated that the original blocks had chl α levels on average about 21% larger than the duplicate blocks. Thus, the missing chl α value for the vandalized original blocks at Trent 400 m d/s MLC could be imputed from the duplicate block values. Also, on July 26/27, 2005, half of the original block was removed at all sites and replaced with a new half block. These new half-blocks were intended to collect data based only on the pulsed discharge.

In August 2006, the Trent u/s Bloedel block was vandalized and found removed from the water. Thus, no Aug 21, 2006 sample was taken from this block. The block was returned to the water, though the September sample may have been compromised.

Table 3 – Summary of 2005 chl α sampling dates and activities in the Trent River Pulsed Discharge study.

DATE	June 6/7	June 22	July 5/6	July 15	July 26/27	Aug 23/24	Sept 13/15	Sept 15
ACTIVITY	Set up artificial blocks	Partial sample set	Sample set 1 – pre-pulse	Start of pulse	Sample set 2 - during pulse	Sample set 3 – during pulse	Sample set 4 – during pulse	Pulse stop
			Set out duplicate blocks		Set out half-blocks		Remove blocks	

Data analysis

Where <15% of values were less than the minimum detection limit (MDL), values were replaced with half the MDL for that parameter. Regression analyses were performed in Statistica to determine the ability of predicting values of one parameter from values of another. Where applicable parameter values were compared to BC Water Quality Guidelines (BCME, 2006), referred to from this point on as "the Guidelines".

Results

2002- 2004

Historical water chemistry and algal biomass data are included in Appendix 1.

2005 - 2006

Water Chemistry

Of the 2005 water chemistry parameters considered in this study, all but ammonia in 2006 and pH showed very low or below MDL levels at Trent River sites upstream of the confluence of MLC. Below the confluence of MLC, all but pH and ammonia showed a clear decreasing trend with distance from the confluence. With the exception of nitrate+nitrite in both years, and true colour in 2006, there was also a clear decreasing trend after late-June as the sampling period progressed. For all parameters but nitrate+nitrite, levels similar to background levels were reached at 300 m d/s of the trestle in 2005. In 2006, the same was true at 500 m u/s trestle.

pH

Mean pH in 2005 and 2006 were very similar [7.6 (SE 0.04), and 7.7 (SE 0.05), respectively], though slightly lower than the 2002-2004 historical data [8.0 (SE 0.05), Appendix 1]. There was a trend of increasing pH as the sampling period progressed in both 2005 and 2006. Late summer values reached values similar to those from the singular late summer values from previous years.

True colour
The Trent River upstream of MLC was generally at or below the MDL of 5 TCU. However, MLC is generally characterized by a dark brown tea colour as high as 80 TCU in June and early July. The sources of colour in MLC are likely the large wetlands and the sewage discharge. Downstream of MLC, the Trent also exhibited an obvious brown colour, reflective of the influence of MLC. True colour decreased with increasing distance from the confluence with MLC, and again reached values of less than the MDL at 500 m u/s trestle in all sample years (Fig 4 and Appendix 1). The earlier sampling period start date in 2006 allowed observation of minimum values as far upstream as 400 m d/s MLC in May and early June. All values in the Trent decreased as the sampling period progressed. Late summer 2005 and 2006 values tended to be lower than those from singular late summer values from previous years (Appendix 1).

True colour cannot be compared to water quality Guidelines (15 TCU) as samples were not collected to obtain a 30-day average. MLC @ Cumberland Road (u/s of the Cumberland Lagoon) had true colour greater than 15 TCU on all sample dates in 2005 and on the three earliest of the six sample dates in 2006. MLC u/s Trent had true colour greater than 50 TCU on all 2005 and 2006 sample dates. In early July 2005, true colour was greater than 15 TCU at all sample sites downstream of MLC. With the exception of Trent 1km d/s MLC in late July 2005, all other sample dates and sites in the Trent in 2005 and 2006 were below or equal to 15 TCU.

Turbidity
Turbidity in the Trent River upstream of MLC was less than 1 NTU in both 2005 and 2006. In MLC @ Cumberland Rd, upstream of the sewage discharge, turbidity ranged from 31.6 to 85.2 in 2005 and 35.8-135 in 2006 (Fig 5), much higher than at any other site. This is the only site at which recreation and aesthetics Guidelines for turbidity (50 NTU) were exceeded in 2005 and 2006. By the time MLC reached the Trent River, turbidity values were about an order of magnitude lower (max 4 NTU in 2006). Nevertheless, a slight increase in turbidity in the Trent River downstream of MLC (compared to upstream turbidity levels) illustrated the influence of MLC. Values decreased with increasing distance downstream from MLC and as the sampling season progressed. In 2005 and 2006 late summer values tended to be similar to those from singular late summer values from previous years (Appendix 1). One exception to this was in MLC, where turbidity values were slightly higher than previous years.

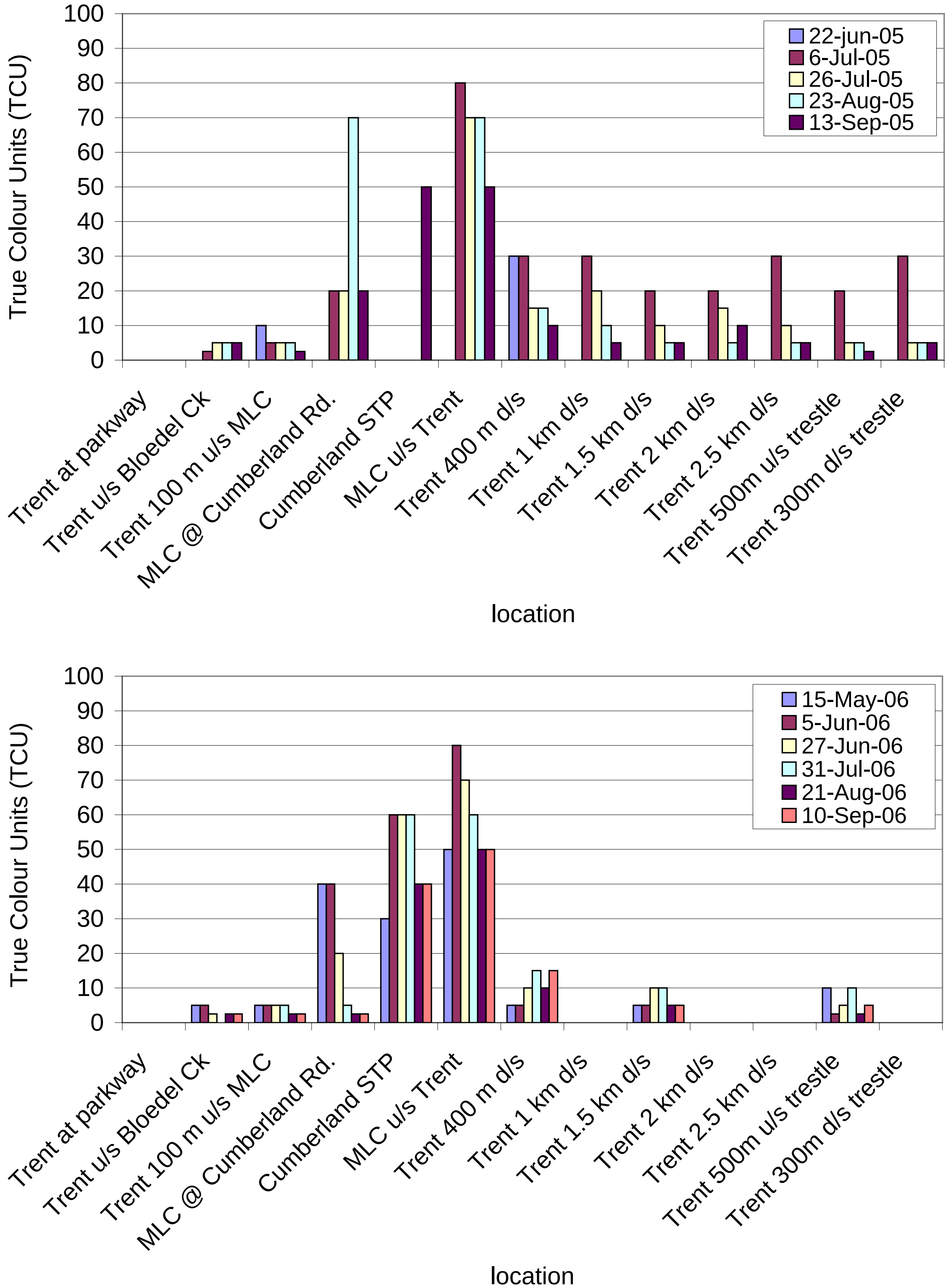
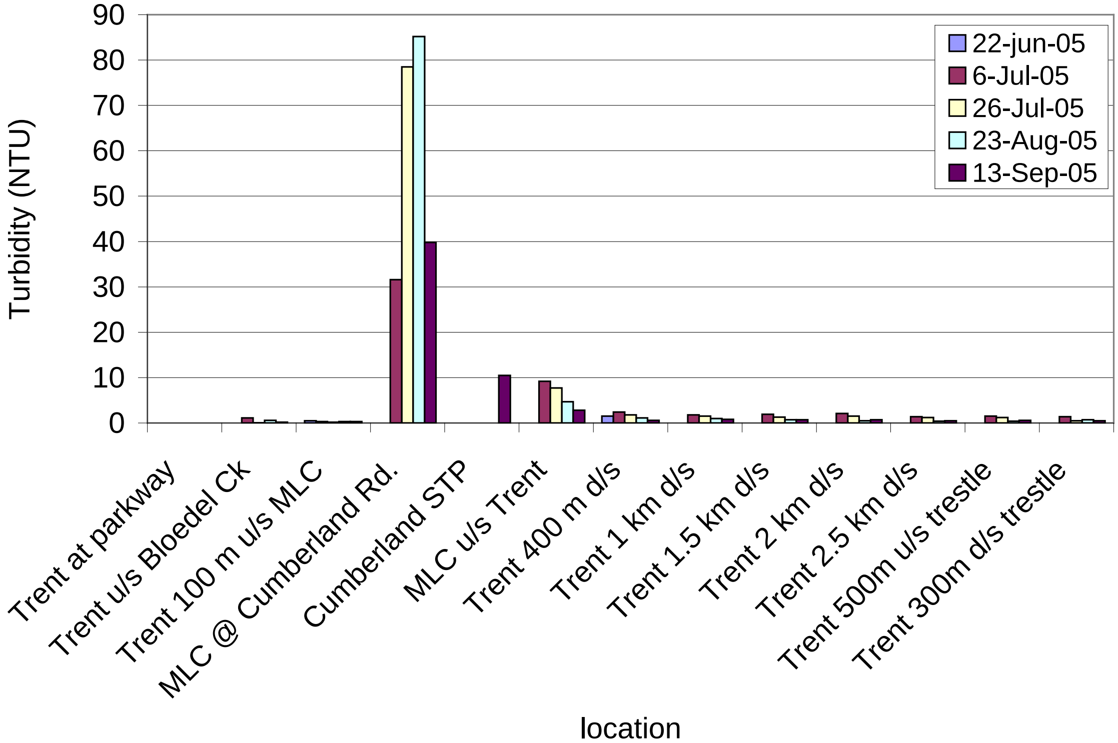


Fig 4 – True colour over four 2005 and six 2006 sample dates in the Trent River and MLC.



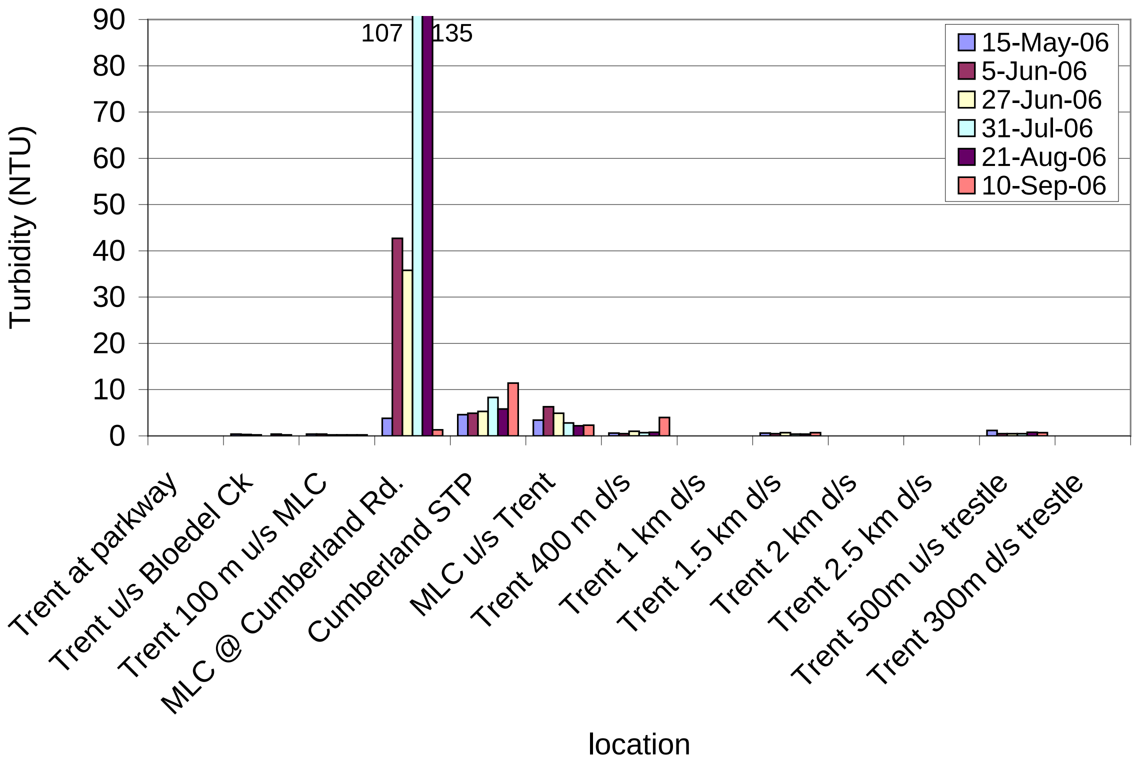


Fig 5 – Turbidity over four 2005 and six 2006 sample dates in the Trent River and MLC.

Ammonia

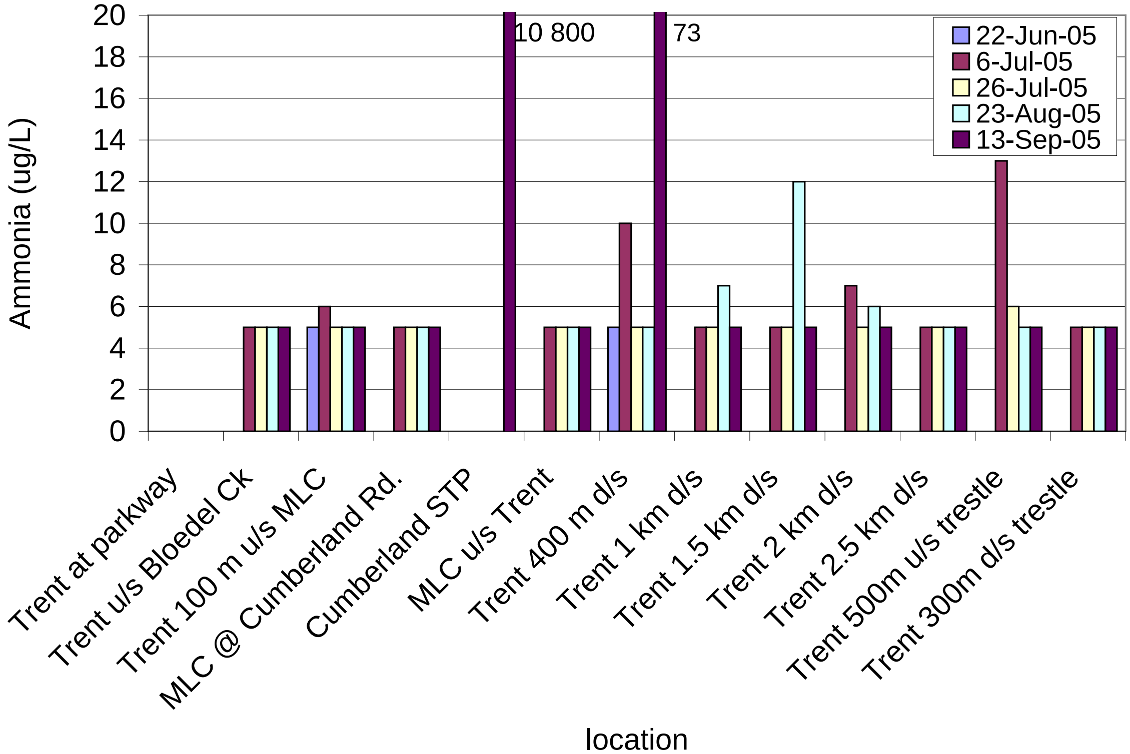
Ammonia levels in the Cumberland sewage discharge from all sampling years ranged from 6600 to 18 000 ug/L (Fig 6 and Appendix 1), indicative of incomplete nitrification in the sewage lagoons. This was confirmed by the relatively low nitrate/nitrite levels in the discharge (see next section). In MLC, as the sewage travels downstream to the confluence with the Trent, the ammonia is gradually converted to nitrate. As a result, in MLC just upstream of the Trent, ammonia was less than or near the detection limit of 5 ug/L in all samples in 2005, and did not exceed 18 ug/L in 2006. In the Trent River upstream and downstream of MLC, ammonia was below or near the detection limit in all samples, not exceeding 14 ug/L in 2005, and 22 ug/L in 2006. As a result, ammonia levels throughout the Trent River met the receiving environment maximum guidelines for the protection of aquatic life in all samples.

Nitrate+nitrite

The Cumberland sewage discharge has been shown to be a significant source of inorganic nitrogen. As discussed above, incomplete nitrification in the lagoons results in elevated ammonia levels in the final discharge. The ammonia is gradually converted to nitrate in MLC in the presence of oxygen.

Levels of nitrate+nitrite in all years of sampling ranged from 4 to 391 ug/L in the Cumberland STP discharge (Fig 7 and Appendix 1). In the Trent River upstream of MLC, concentrations ranged from 6 to 319 ug/L. However, levels ranged from 12 to 1480 ug/L in the Trent 400 m downstream of MLC, reflecting the ammonia concentrations in the STP discharge and the lack of dilution in the Trent River. There was a slight decreasing concentration gradient over the initial 2 km in the Trent, with further decreases in the lower reaches of the Trent River.

Nitrate+nitrite levels remained elevated to the furthest downstream sampling site, 300 m d/s trestle. In 2005, early July values were slightly lower than most other values, with highest values in most cases occurring in late July, and then decreasing with the progression of the sampling period. In 2006, levels increased gradually over the sampling season until the maximum values were reached at all sites in early September. Nitrate+nitrite 2005 and 2006 levels were lower than 2002-2004 levels (Appendix 1). All MLC and Trent River values were lower than the maximum nitrate aquatic life (32 800 ug/L) as well as the recreation and aesthetics (10 000 ug/L) Guidelines.



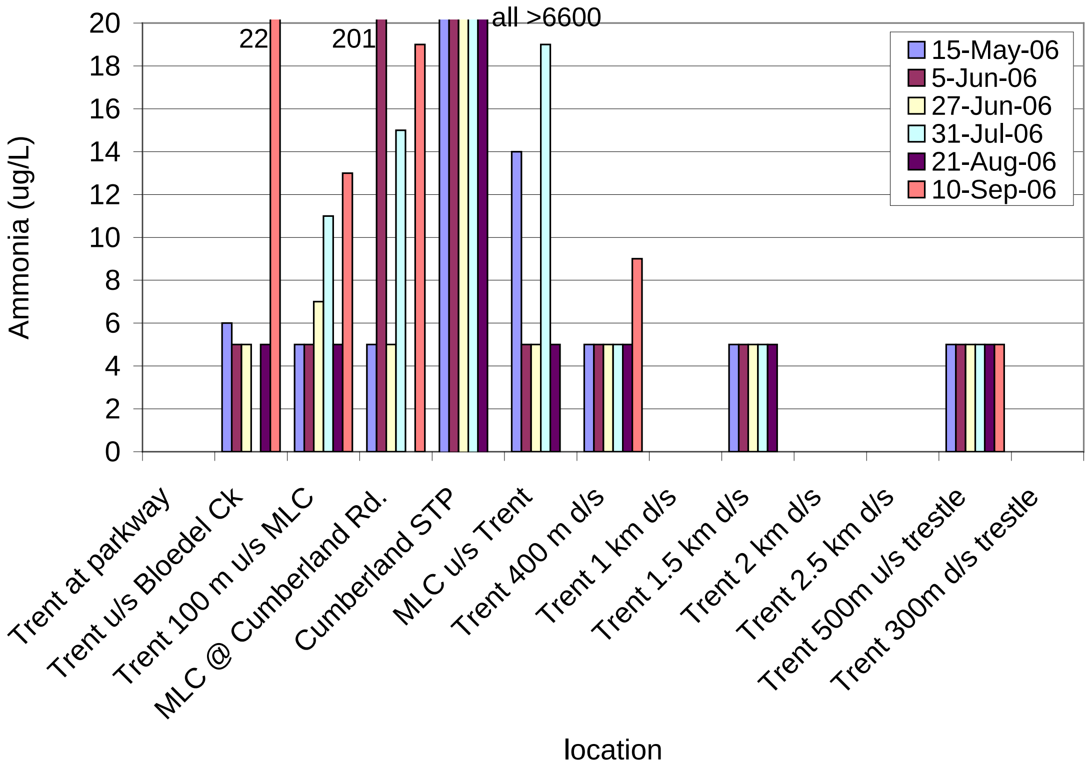
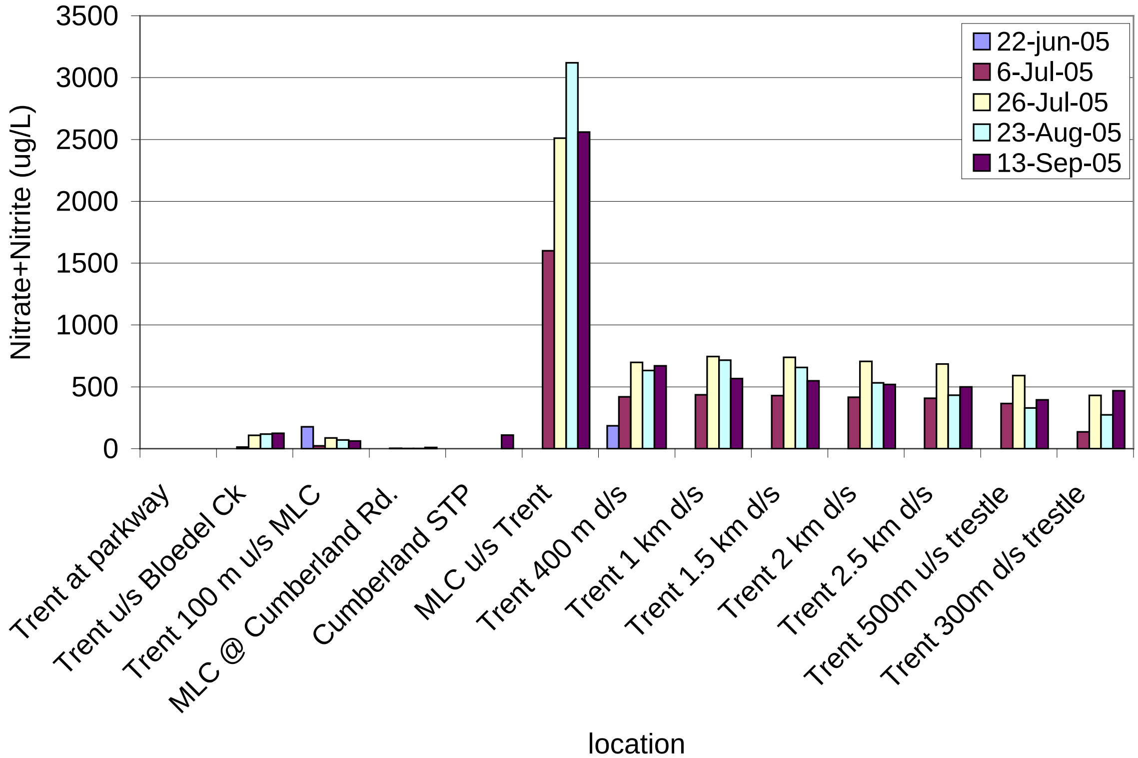


Fig 6 – Ammonia over four 2005 and six 2006 sample dates in the Trent River and MLC.



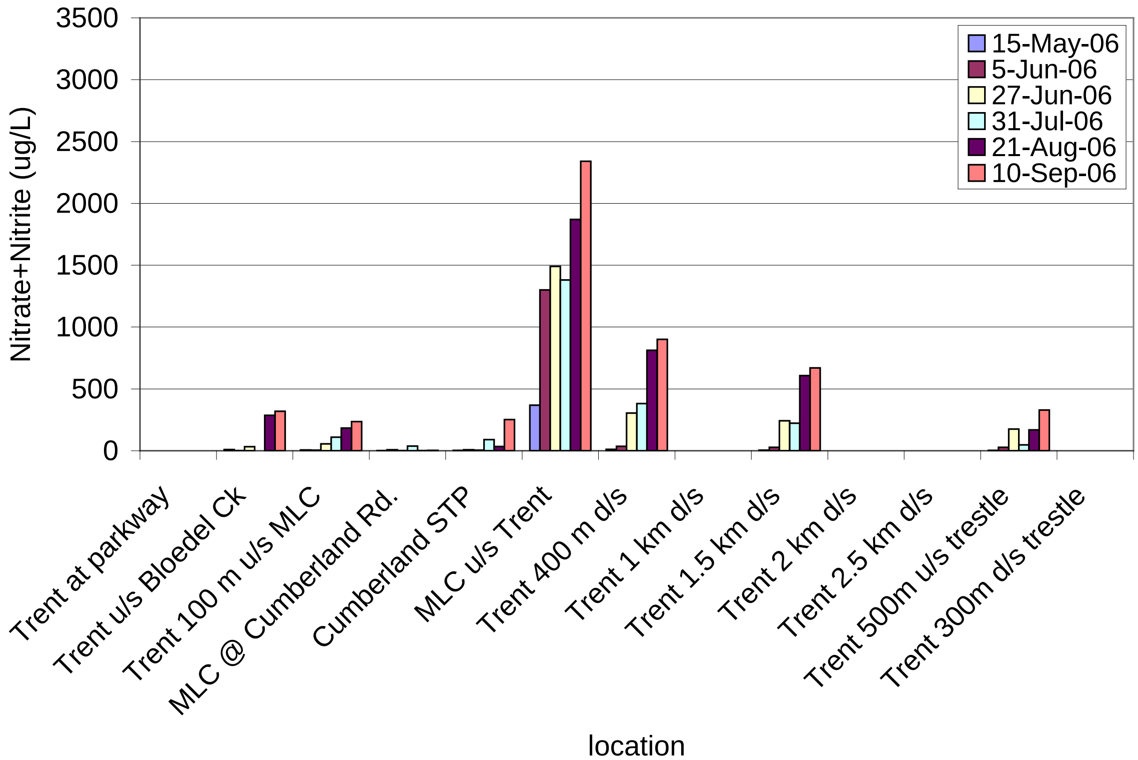


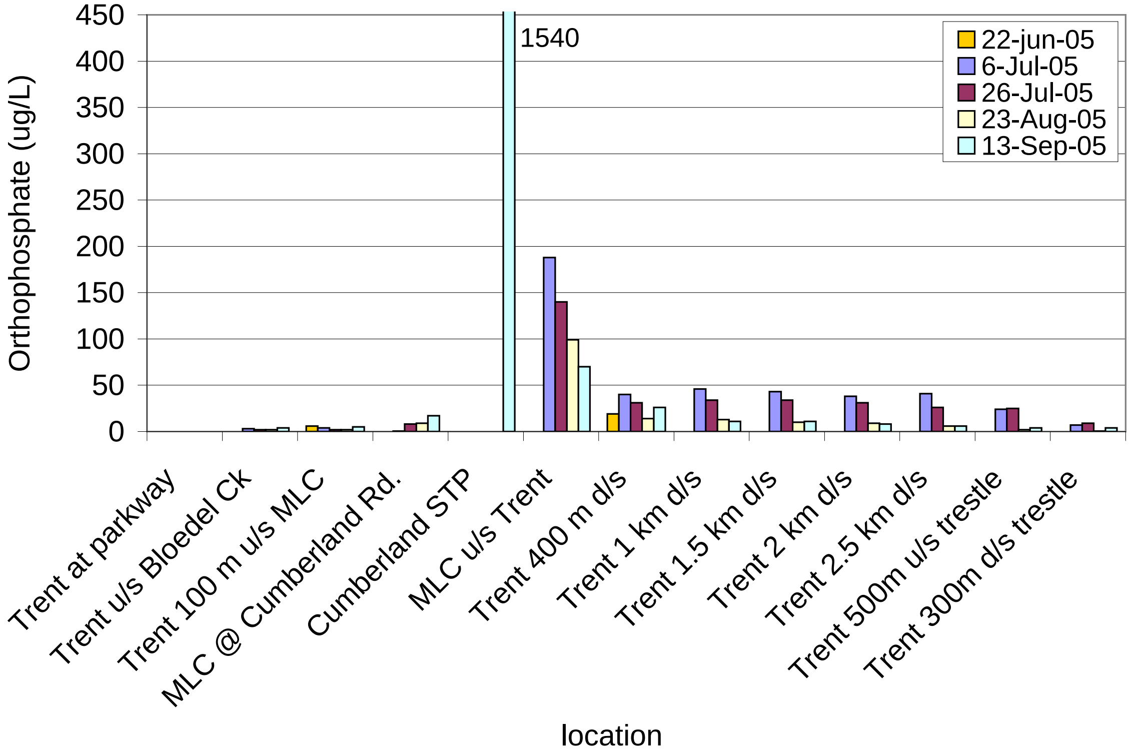
Fig 7 – Nitrate + nitrite over four 2005 and six 2006 sample dates in the Trent River and MLC.

Orthophosphate
Phosphorus is the limiting nutrient to algal growth in the Trent River, and orthophosphate is the biologically available form of phosphorus. In the Trent River upstream of MLC orthophosphate ranged from less than the detection limit of 1 ug/L to 6 ug/L in all years of sampling (Fig 8, Appendix 1). In contrast, the Cumberland STP peaked at 2480 ug/L in 2006. As a result, MLC just upstream of the Trent was also elevated, ranging from 29 to 188 ug/L in 2005 and 2006. In the Trent River 400 m downstream of MLC, levels ranged from 2 to 40 ug/L during this same period. Levels of orthophosphate decreased only slightly through the initial 2 km below MLC. Concentrations approached background at the two most downstream sites (500 m u/s and 300 d/s of the trestle). Generally, orthophosphate showed a seasonal trend of increasing until maximum levels were reached in late June/early July, then gradually decreasing through the rest of the summer. Orthophosphate also tended to decrease over the years of sampling from 2002-2006.

Total phosphorous
Total phosphorous showed very similar trends to that of orthophosphate by season and when compared to previous years' data (Fig 9 and Appendix 1). In the Trent River upstream of MLC, total phosphorus ranged from less than the detection limit of 2 ug/L to 9 ug/L. The maximum value at the Cumberland STP for all years was 2740 ug/L in 2006. MLC just upstream of the Trent was also elevated, ranging from 54 to 311 ug/L in 2005 and 2006. In the Trent 400 m downstream of MLC, levels ranged from 6 – 80 ug/L. Levels of total phosphorus decreased only slightly through the initial 2 km below MLC. The maximum Vancouver Island draft interim phosphorous objective of 7 ug/L for Vancouver Island streams was exceeded in all but three (2006) and seven (2006)Trent River samples downstream of the MLC confluence. Concentrations only approached background at the two most downstream sites in May, August and September.

Microbiological indicators

E. coli and fecal coliforms
E. coli and fecal coliforms showed no clear trends throughout the sampling season or with distance from the confluence. Also, little to no dilution influence was observed (Fig 10, Fig 11). All peak *E.coli* values downstream were higher than *E.coli* levels in MLC. The maximum *E. coli* values were observed 1.5 km downstream of MLC (190 CFU/ml in late July 2005 and in late June 2006), three peaks were observed in the lower reaches of the Trent (83-96 CFU/ml) in early July 2005, and another peak at 400 m d/s MLC in late July and August 2006. Similar observations were apparent in the fecal coliform data, with two and four sites showing high peaks in early and late July 2005, respectively. Also, 1.5 km d/s MLC showed a peak in late June 2006. Bacteriological samples were not collected frequently enough to compare to the primary contact recreation Guidelines, though the Guidelines has been shown on the figures to demonstrate that some of the values are notably high.



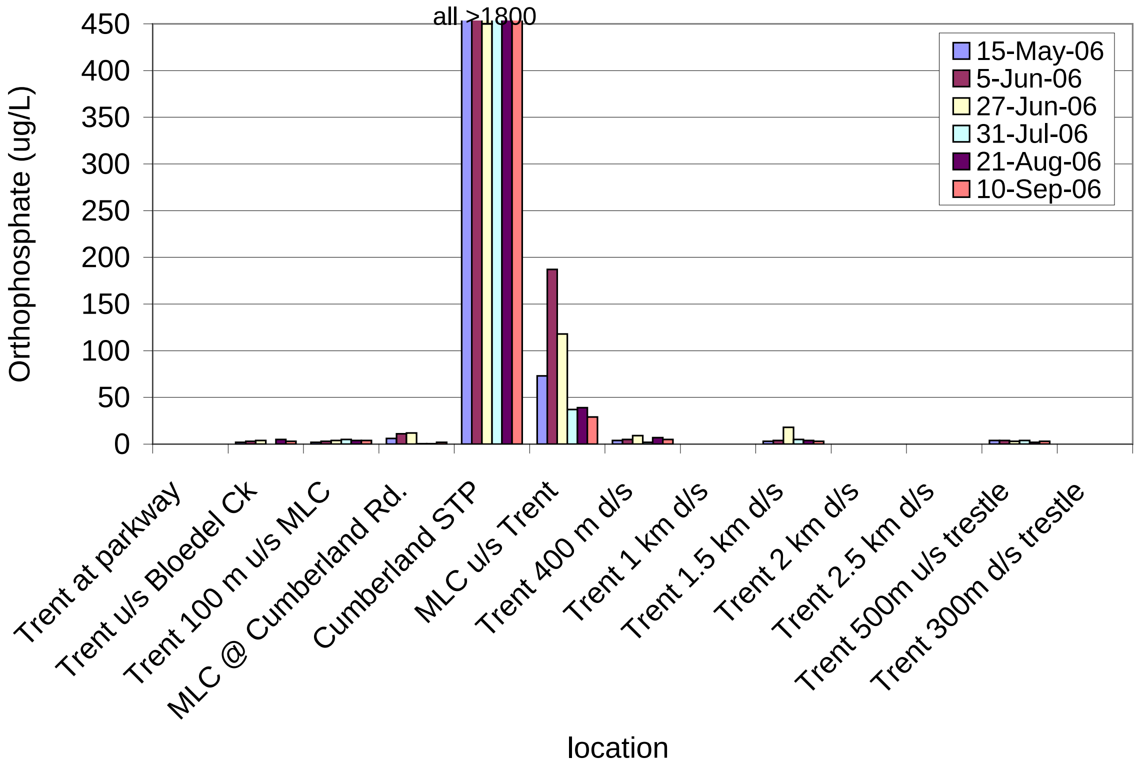
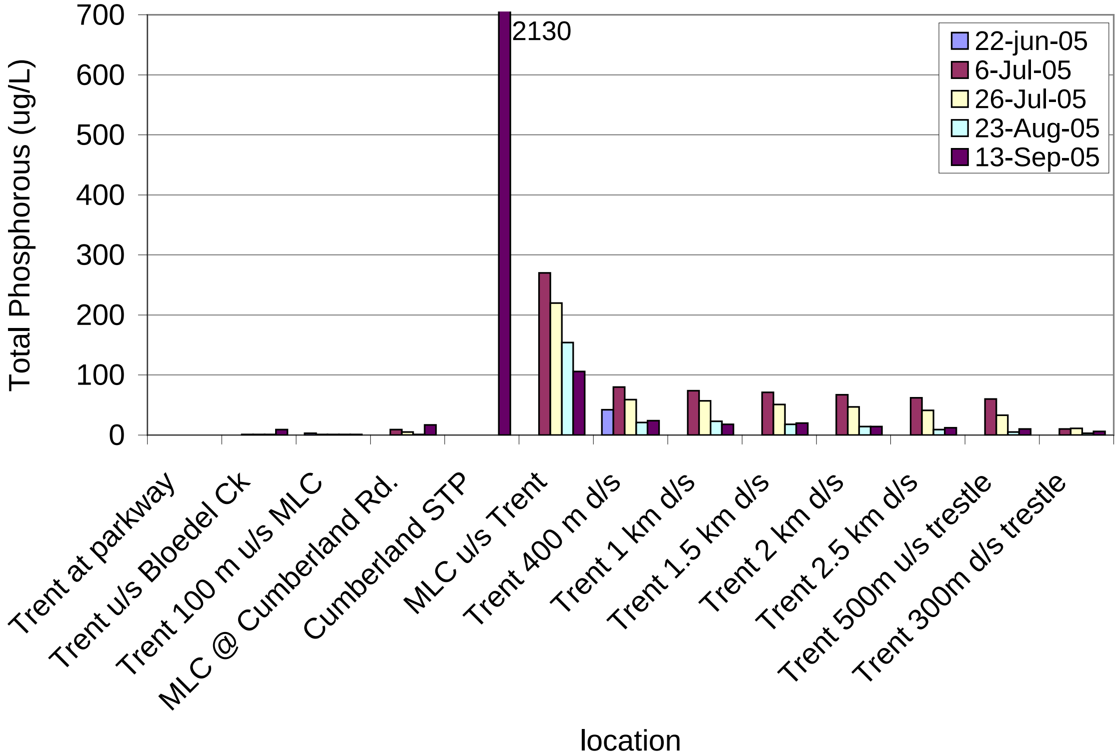


Fig 8 – Orthophosphate over five 2005 and six 2006 sample dates in the Trent River and MLC.



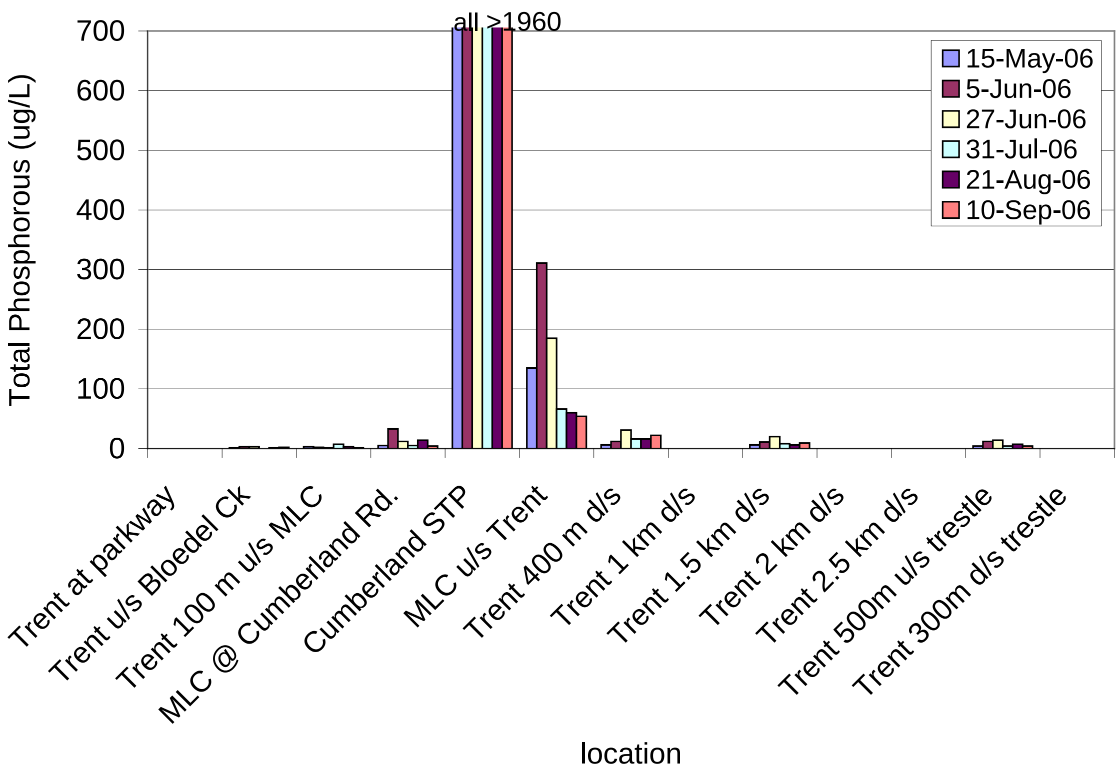


Fig 9 – Total phosphorous over four 2005 and six 2006 sample dates in the Trent River and MLC.

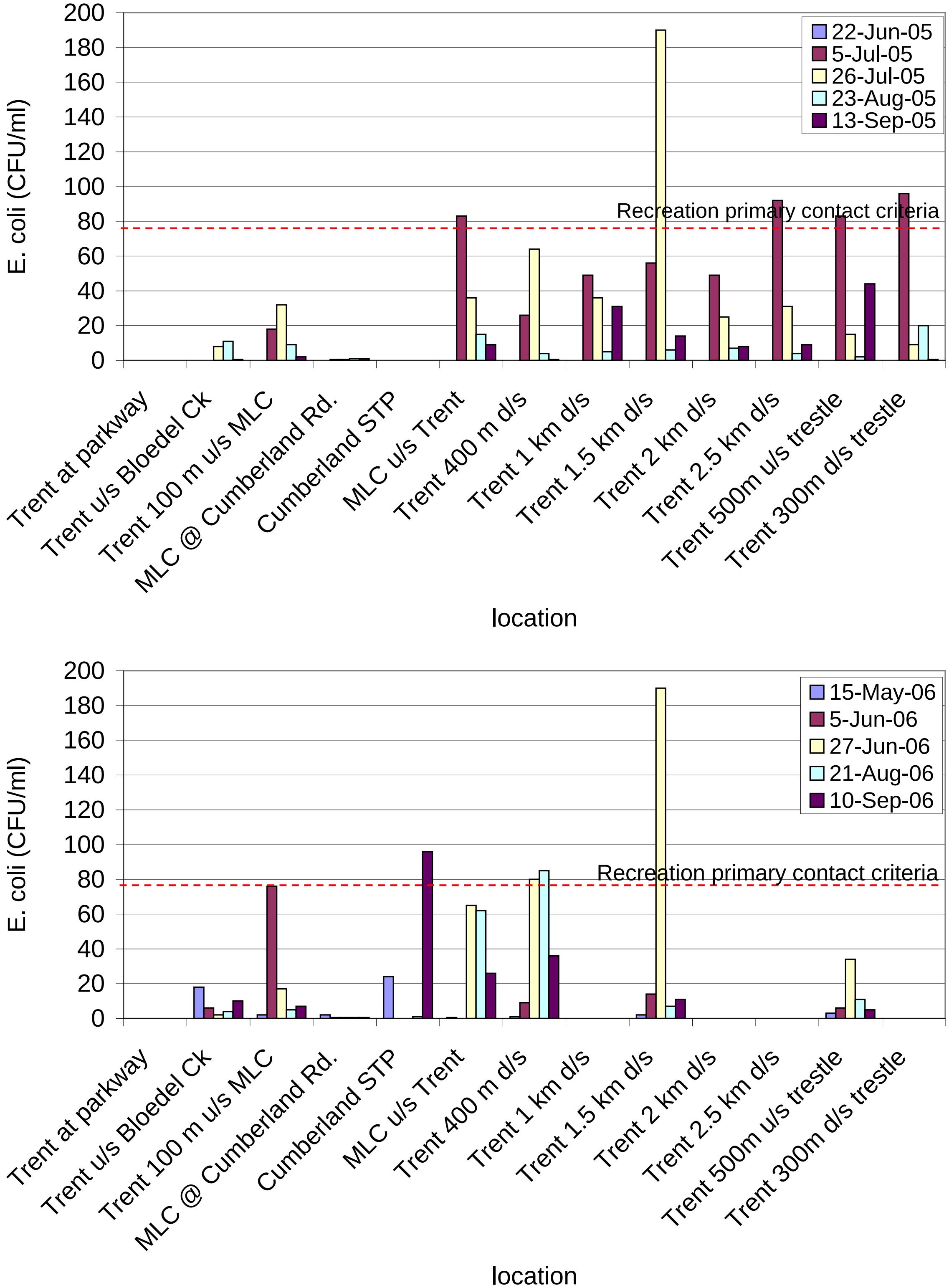
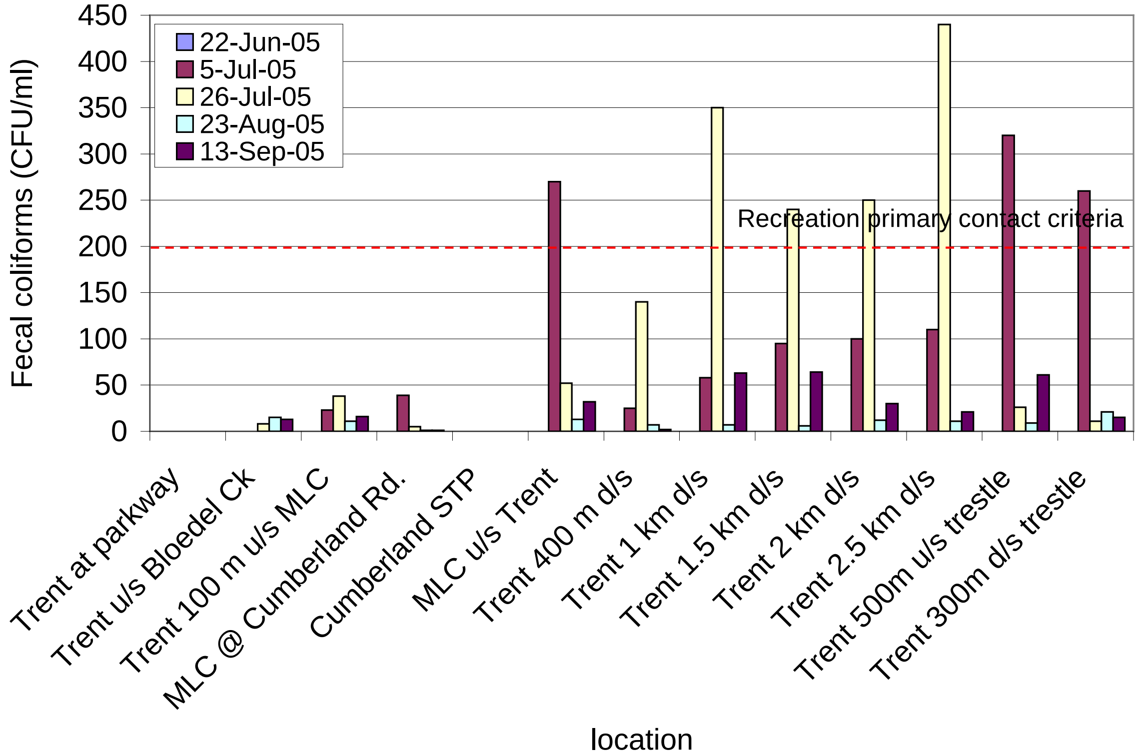


Fig 10 – *E. coli* over four 2005 and six 2006 sample dates in the Trent River and MLC.



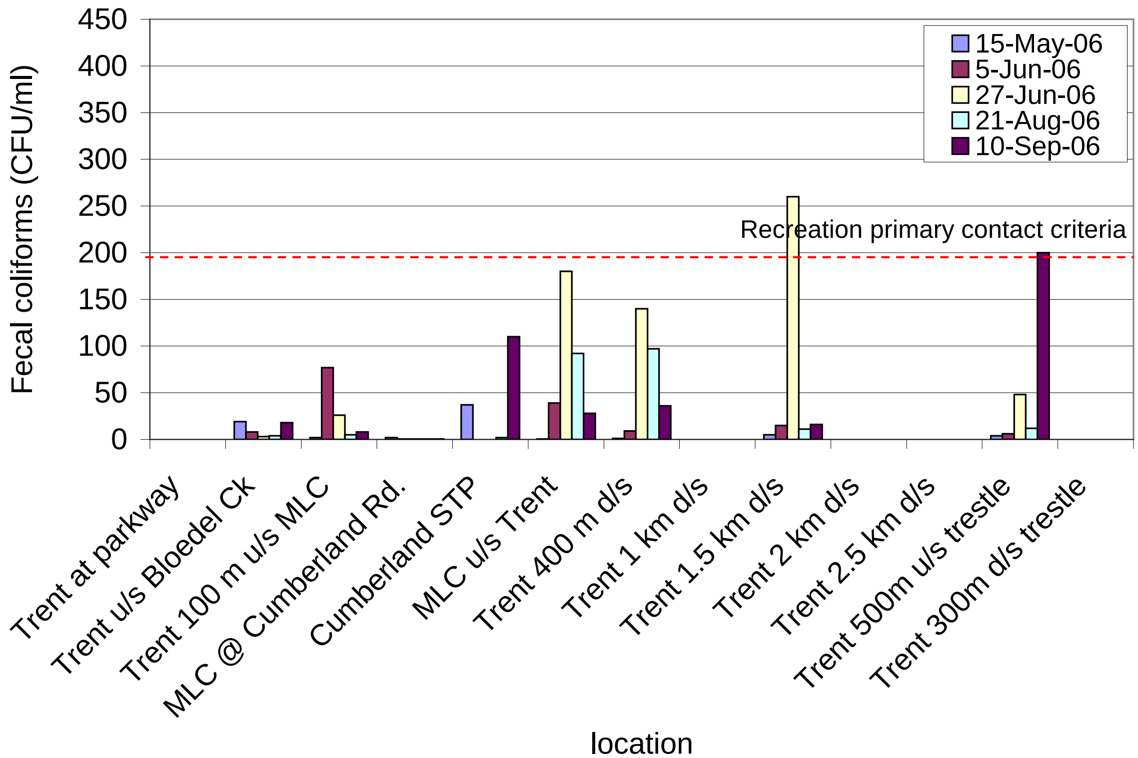


Fig 11 – Fecal coliforms over four 2005 and six 2006 sample dates in the Trent River and MLC.

Algal biomass

Chlorophyll α

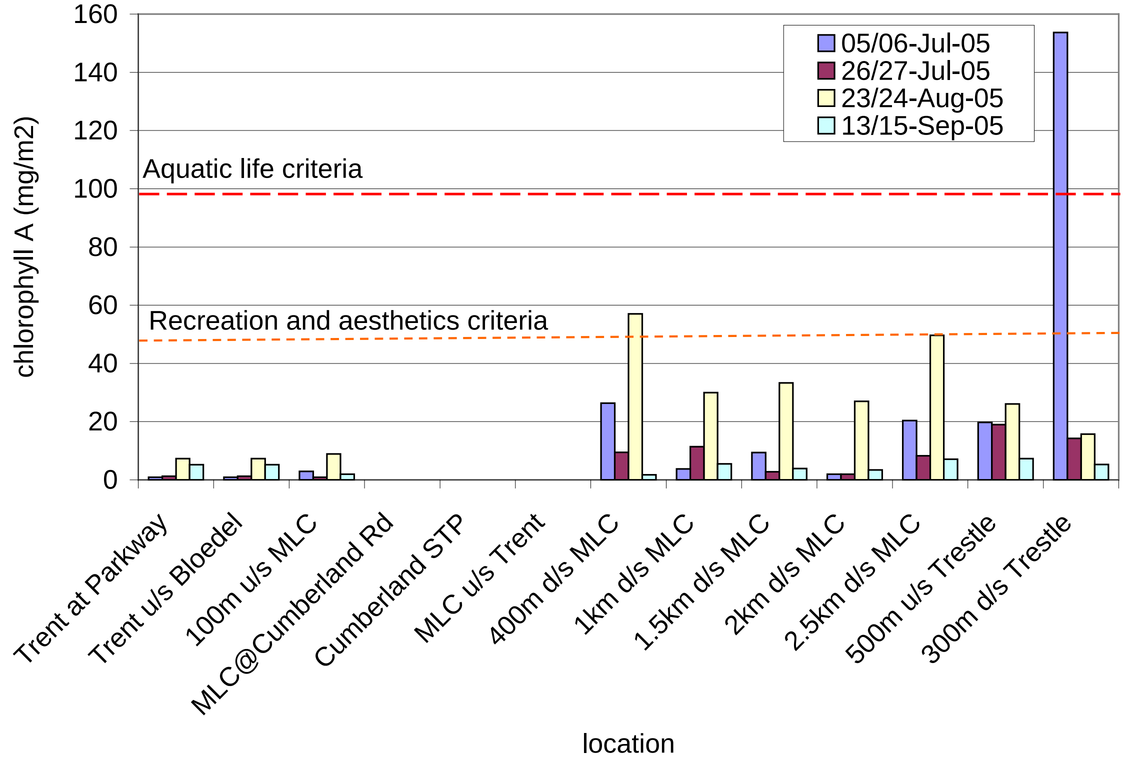
Natural substrates

Natural substrate chl α data reflect standing crop. Late summer 2005 values (Fig 12) were comparable to the late summer 2003 data (Appendix 1), while 2006 values at the downstream sites were slightly lower than all previous years. While 2005 data showed chl α peaks in July and August, 2006 data showed a gradual increase throughout the summer with highest values in September. At the Trent River sites upstream of MLC, chl α ranged from 0.6 to 14.9 mg/m^2 (Fig 12), generally about an order of magnitude less than at the Trent River 400 m d/s of MLC. Overall, natural substrate chl α exceeded recreation and aesthetics Guideline only twice: in August 2005 at 400 m d/s MLC (57 mg/m^2), and in early July 2005 at 300 m d/s of the trestle (154 mg/m^2), which also exceeded the aquatic life guideline. Chl α values from 1km d/s to 2km d/s MLC tended to be lower than at the other downstream sample sites. In early July and September 2005, values in these areas approached upstream levels. Village of Cumberland chl α natural values were similar to but generally less than MOE values on similar dates, and no single value exceeded 18 mg/m^2 (Appendix 1).

Artificial substrates

In 2005, the initial samples taken from the artificial substrates after a 4 week exposure reflect accrual rather than standing crop. The second sample set taken after a 7 week exposure better reflects standing crop and a stable algal biomass. Original artificial substrate (installed pre-pulse) chl α were consistently higher than natural substrate chl α (Fig 13). While September 2005 artificial substrate values at downstream sites were lower than upstream (background), other dates sampled generally showed values much higher than background levels. Of the seven downstream sites in 2005, three exceeded the recreation and aesthetics Guideline in early July, six in late July, and all exceeded in August. Of the three downstream sites in 2006, one exceeded these Guidelines in June, and all exceeded in July-September. The aquatic life Guideline was exceeded by one site in early July 2005 and four sites in August 2005, but was not exceeded in the 2006 data. As with the natural substrate data, the maximum value of 151 mg/m^2 occurred in August 2005 at 400 m d/s MLC. Unlike the natural substrate data, no sites consistently showed comparatively low chl α levels.

Half block values (installed during pulse) are not presented as a bar graph, as they were so similar to original block data. A regression analysis of data from both types of blocks was used to see if one could predict the other. A successful pulse should have resulted in a lack of relationship between the two. The analysis showed a significant relationship ($p = <0.05$; $y = -1.91 + 0.88*x$) between the two (Fig 14), with an r^2 close to 1 ($r^2 = 0.93$).



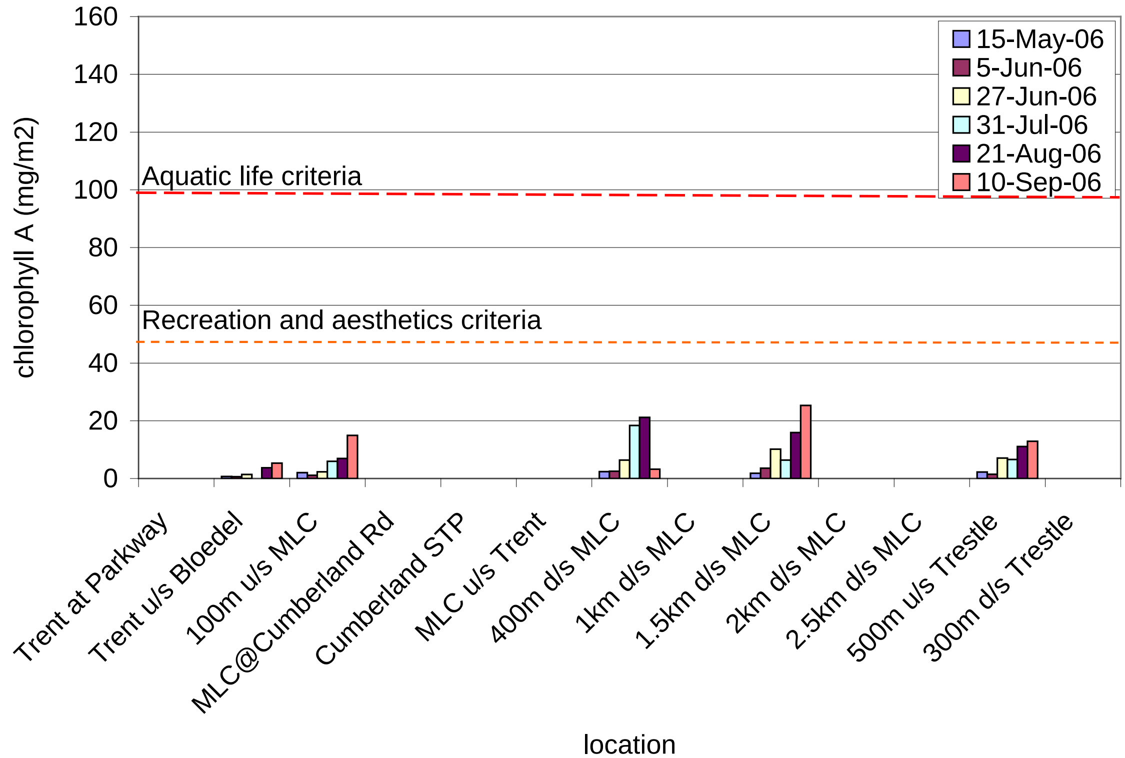
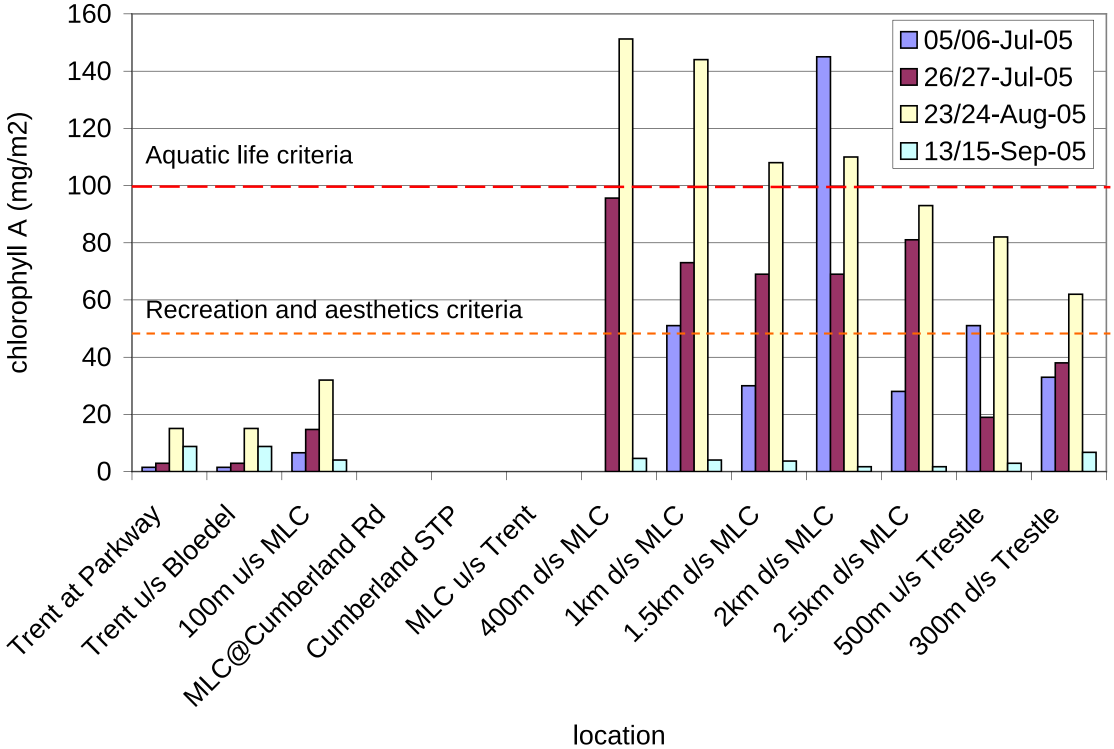


Fig 12 – Chl α (natural substrate) over four 2005 and six 2006 sample dates in the Trent River.



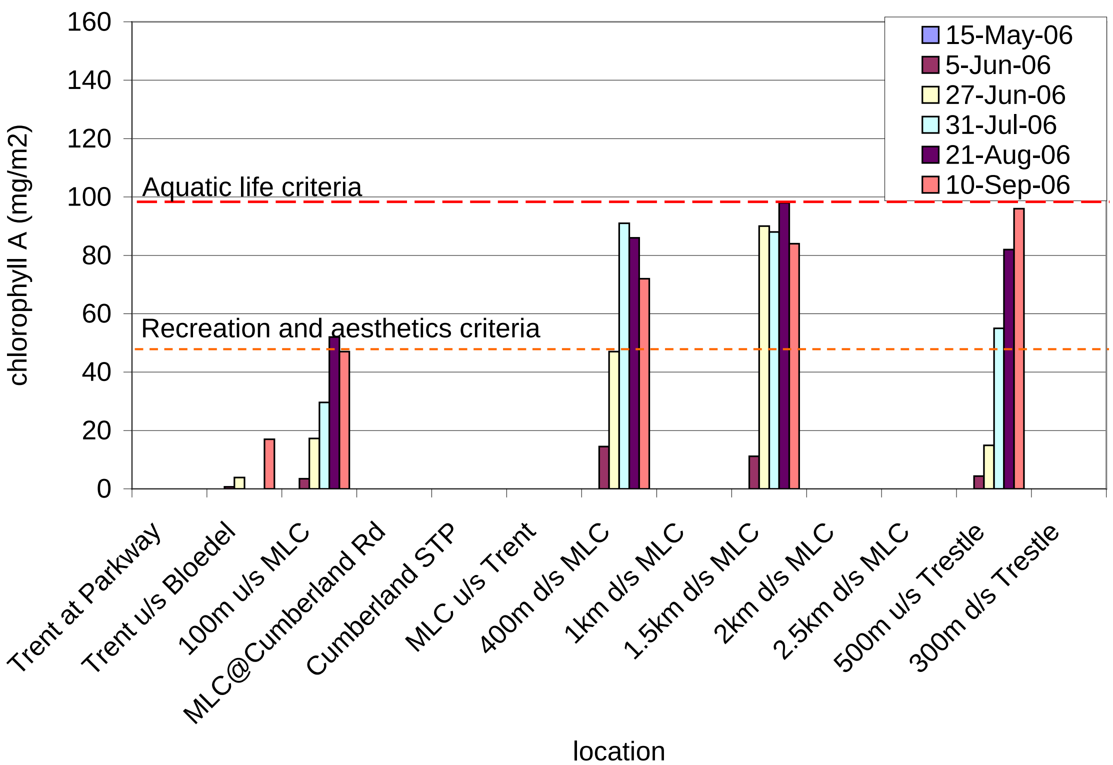


Fig 13 – Chl α (original artificial substrate) over four 2005 and six 2006 sample dates in the Trent River.

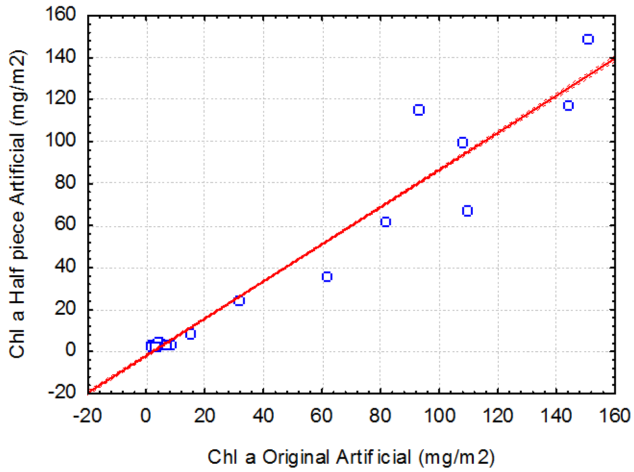


Fig 14 – Regression of chl α original artificial with chl α half-piece artificial substrate (r^2=0.93, p =<0.05, y=-1.91+0.88*x). Dashed lines show a 95% confidence interval.

Water velocity and flow model 2005

Existing velocity data did not include late July data (Fig 15). Data show that, at most sites, lowest water velocities occurred in August and ranged from 4 -25 cm/s). Seasonal values ranged from 4-40 cm/s. The maximum velocity in early July (40 cm/s) and August (25 cm/s) occurred at Trent 1 km d/s MLC.

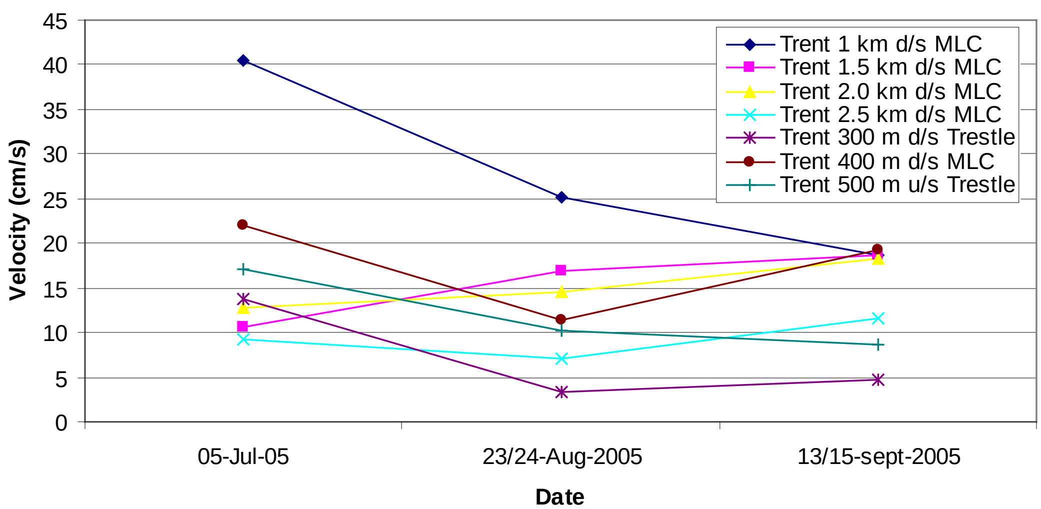


Fig 15 – Velocity data by date and site along the Trent River downstream of MLC.

Results of the flow model indicated a strong location effect on velocity (DF=6, F-ratio=4.96, p<0.05). However, despite a large variation (650-1350 L/s), flow had no appreciable effect on velocity (DF=1, F-ratio=1.24, p=0.27). Mean velocities from the antilog of the values ranged from 6-26 cm/s. These imply that travel time over the 5km from the confluence to the Strait of Georgia would range from approximately 5-23 hours, dependent on location.

Pulsed discharge and automated samplers

The pulsed discharge period started with two 50% max flow discharges 72 hours in length. They were followed immediately by one full strength,12-hour pulse, and all were spaced one day apart. With another day's spacing came a 50% max flow 24-hour pulse, after which there was a week of no discharge. The remainder of the summer (August 4 – September 15) consisted of eleven 24 hours pulses spaced three days apart (Fig 16 and 17). There was a noted minor leakage around the edges of the shutoff valve.

Village of Cumberland total phosphate data ranged from 70-467 ug/L total phosphate at the MLC outlet. Peaks in total phosphate levels tended to occur slightly after the beginning of a discharge release. The onsets of these peaks were gradual, and lag time from start of discharge until the full peak was visible appeared to vary from approximately 18-72 hours over the entire sampling period. Some non-pulse "valley" phosphorous levels were higher than "peak" pulse related values. Two anomalous data spikes were noted during a period in which no discharge was released. For a six day period from the July 14-20, 2005, the Village of Cumberland automated sampler samples were mistakenly analyzed for dissolved phosphate rather than total phosphate; thus, for this period dissolved phosphate data are included in place of total phosphate data in Fig 16. With only six days of both orthophosphate and total phosphorous data, it was difficult to determine trends of orthophosphate beyond the period sampled. However, in this period the ratio of orthophosphate:total phosphorous varied from 0.45 -1.3, with a mean of 0.63.

MOE's automatic sampler data from 400 m d/s MLC, though limited to only three days (26-29 July) and two releases from the STP, showed similar peaks as noted in the MLC outlet data for the same dates. The peaks occurred 3-12 hours after the peak noted by the automated sampler in MLC.

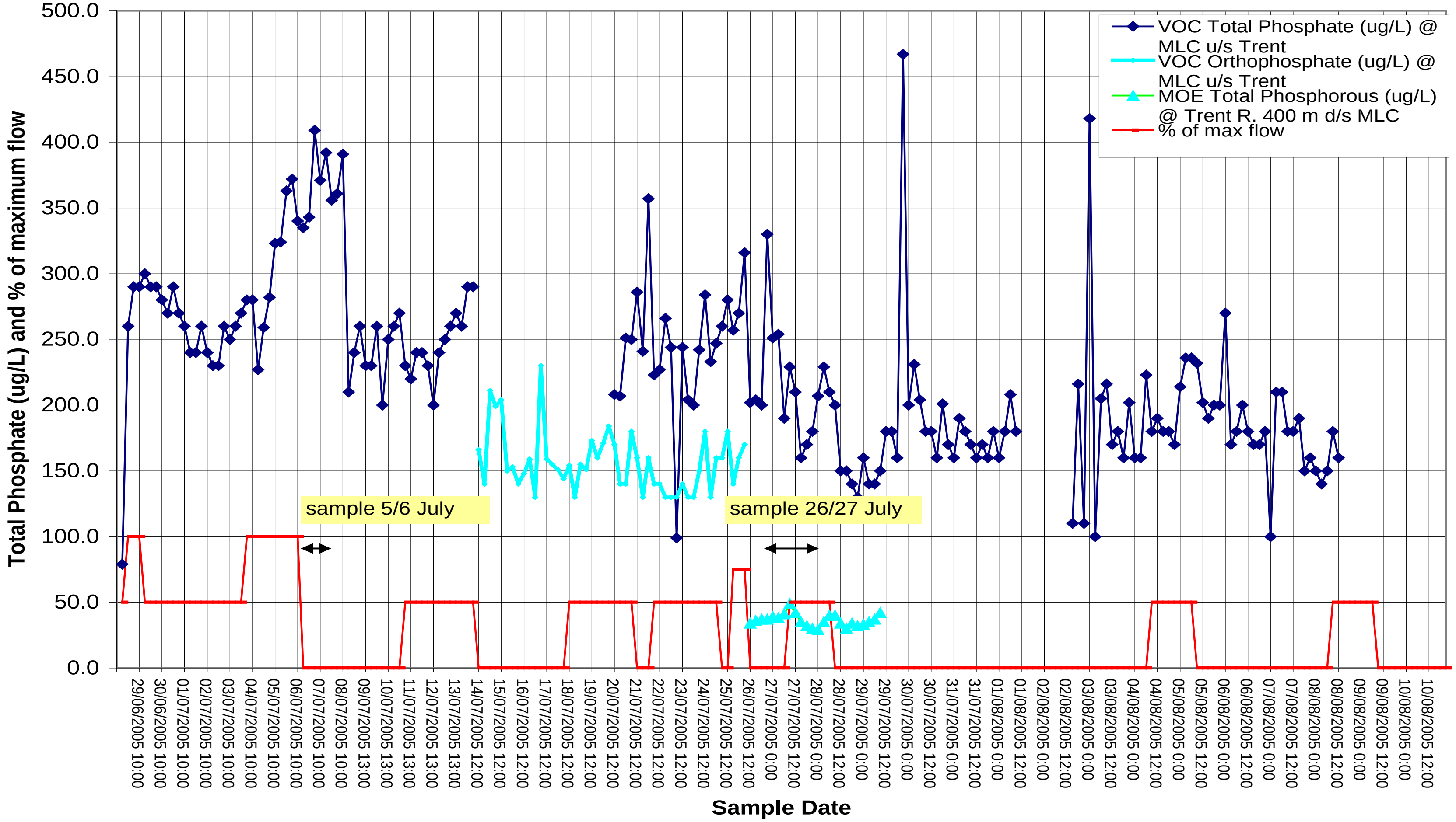


Fig 16 – Pulsed discharge and automated sampler data from June 29 to August 10, 2005. Pulsed discharge periods range from 12-72 hours. VOC=Village of Cumberland, MOE=Ministry of Environment. MOE sample dates are labelled as such and indicated by inclusive arrows. Note: Before “26/07/2005 12:00:00 PM”, the sample interval is 6 hours and each vertical line represents 24 hours; after and including “26/07/2005 12:00:00 PM” the sample interval is 3 hours and each vertical line represents 12 hours.

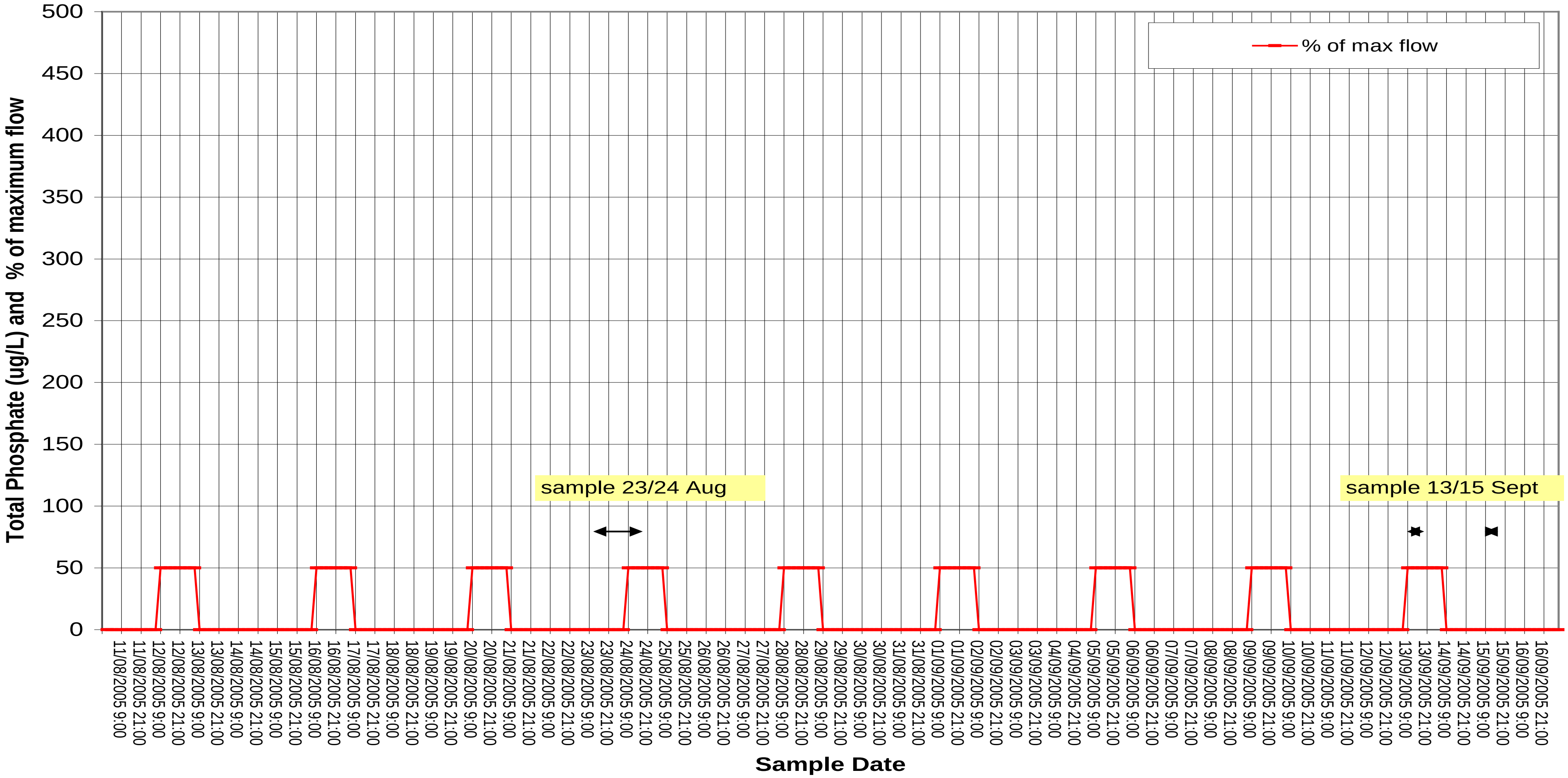


Fig 17 – Pulsed discharge data from August 11 to September 16, 2005. Each pulsed discharge period was approximately 24 hours. No automated sampler data was taken over this time interval. MOE sample dates are labelled as such and indicated by inclusive arrows.

Dye testing 2006

For both tests, clear breakthrough curves (when the fluorescence from the dye is first visible on a plot as an upward curve) on the Trent River were observed. The first trial on May 31, 2006 produced a curve that returned to normal stream conditions within 5 hours (Figure 18). The second trial on June 21 produced a curve that appeared to return to normal after approximately 32 hours (Figure 19). The second trial was more difficult to interpret because probes were switched during the tail portion of the curve. At this point, the fluorescence values increased dramatically again, and took a significant amount of time to fall back down. In fact, the values appeared to still be dropping slightly when the probe was removed. Testing of the probes showed that they produced readings with approximately the same values for samples in the lab. Thus it was assumed that this second peak was a result of a contamination of the area while changing probes. RWT may have been on the sampler's hands or clothes while changing the probes, causing a drastic change in the localized concentration of RWT.

There were no visible breakthrough curves on MLC after 25 and 147.5 hours on June 1st and June 21st, respectively (Figures 20 and 21).

Although a breakthrough curve was observed on both occasions for the Trent River, the recovery rates were quite poor. On the May 31st trial, 3.3% of the injected mass of RWT was recovered. On the June 21st trial, 1.9% of the mass injected was recovered, including the mass represented by the spike following the probe switch. When the model was used, it was difficult to get the peak of the curve as low as the actual data. In order to get the peaks of each curve to line up, it was necessary to use an estimated discharge amount (Q) that was slightly higher than the discharge measured further downstream, to decrease the value of C (injection/Q = g/(m^3/s)). This was most likely required because the tails of the actual data represented a high proportion of the recovered mass, and the model used was not complex enough to show an asymmetrical curve. The results of the optimized models are in Table 4.

The discharge measured at the old Water Survey of Canada station (old highway bridge) on June 1st was 2.4 m^3/s. The June 21st measurements are summarized in Table 5. The values shown may not accurately represent real flows because of the accuracy of the flow meter in such low-volume streams. However, the trend shown in Figure 22 makes it quite clear that the first half of the stream is "losing" while the second half is "gaining".

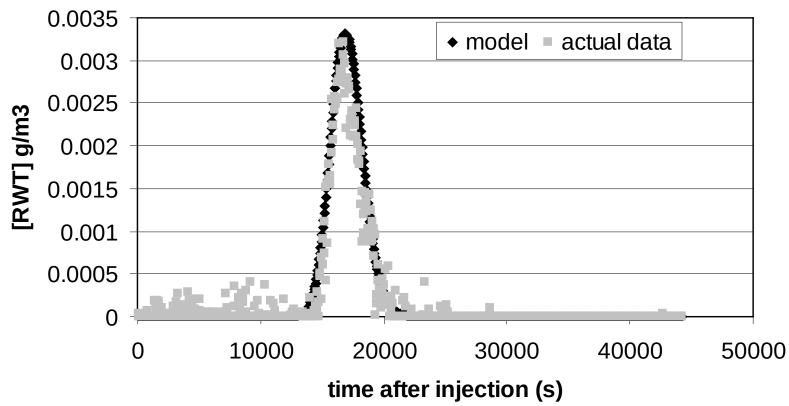


Fig 18 - Actual and modeled dye testing breakthrough curves from Trent River on June 1, 2006.

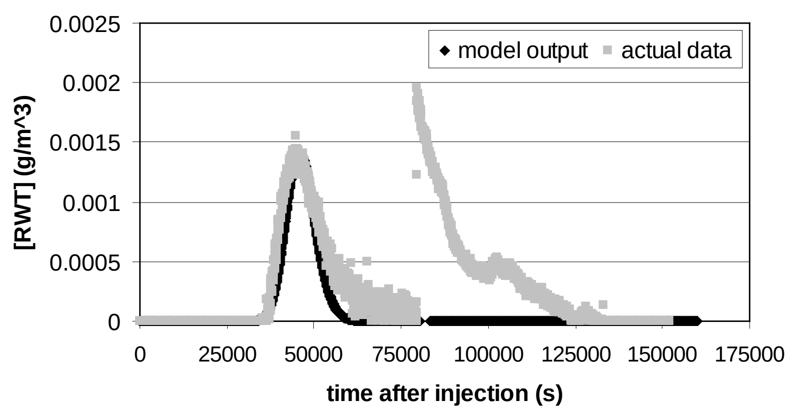


Fig 19 - Actual and modeled dye testing breakthrough curves from Trent River on June 22, 2006.

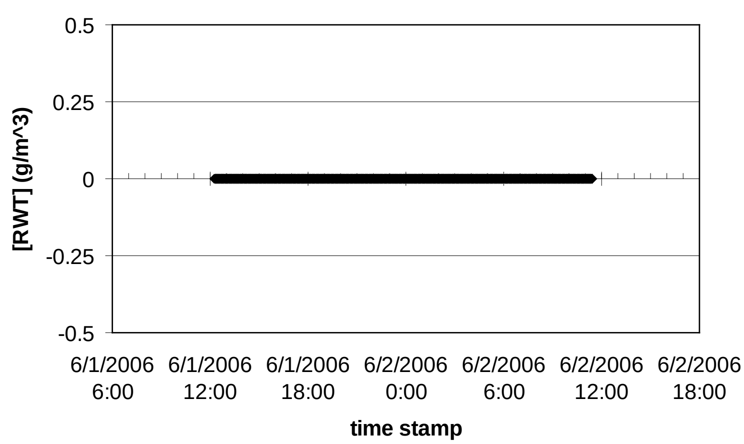


Fig 20 – Converted dye testing field results from MLC on June 1, 2006.

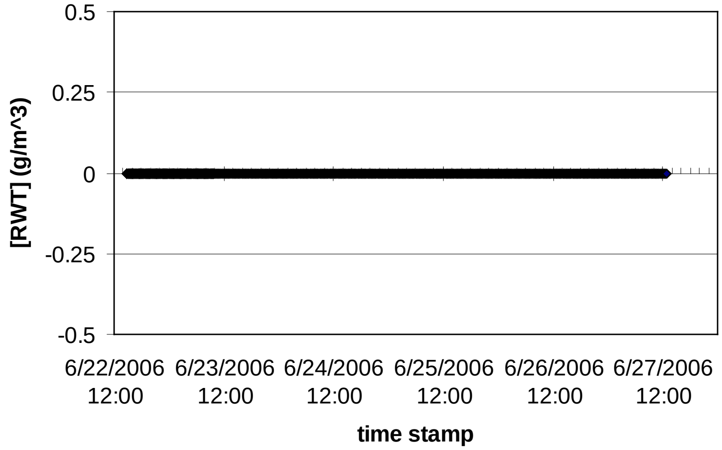


Fig 21 - Converted dye testing field results from MLC on June 22, 2006.

Table 4 - results from the CDM for Trent River

Date	Recovered Mass (g)	Recovery Rate (%)	Model Q (m^3/s)	Velocity (v) (m/s)	Dispersion (D) (m^2/s)
June 1, 2006	22.87	3.28	4.3	0.264	3.4
June 22, 2006	5.90	0.91	0.8	0.097	1.8

Table 5 - Summary of June 21, 2006 discharge results for Trent River.

Location	distance from MLC (km)	Discharge (m^3/s)
200 m d/s MLC	0.2	0.22386
1 km d/s	1	0.2413575
1.5 km d/s	1.5	0.14292
2 km d/s	2	0.122775
old highway bridge	5	0.25878

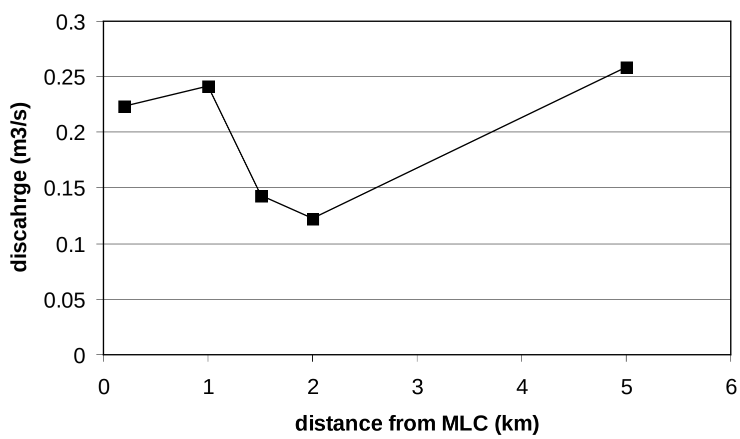


Figure 22 - Discharge measurements showing losing and gaining trends in Trent River.

Discussion

As a result of the thorough 2005 sampling conducted in the Trent, a substantial data set was produced that has provided a clearer understanding of the Trent River system downstream of MLC. Unfortunately, due to the underestimated complexity of the system and lack of adequate travel time data, it was only possible to obtain a rough estimate of travel time in the Trent. In 2005, without accurate travel time data, it was not possible to optimally implement the pulse or to determine if a pulsed discharge could be successful in reducing algal growth in the river. In 2006, dye testing provided some general information about travel time and temporal spread of an injected substance during low to moderate flows on the Trent River. This more accurate data quickly confirmed that travel time in MLC was too long for a successful pulsed discharge to be implemented.

Water chemistry and chlorophyll α

For the most part, trends and ranges of observed nutrient and bacteriological levels in both 2005 and 2006 were as expected based on previous data collection and the influence of the STP. In early summer, the Trent provides somewhat greater dilution of MLC and the influence of the STP. By mid summer, as the Trent River's flow continues to decrease, there is only minimal dilution of MLC (typically less than 5:1). Higher pH as the season progressed can be attributed to less precipitation in the summer (Wetzel 2001). Levels of orthophosphate, total phosphorus and nitrate+nitrite decrease only slightly through the initial 2.5 km of the river upstream of the trestle. It is only when the river opens up to more light that natural biological uptake of these nutrients becomes a significant factor. Thus, they approach background levels only in the most downstream sites in the Trent River. Turbidity levels tended to decrease through the summer months in MLC. The dominant source of turbidity appears to be MLC upstream of the STP. As flows decrease, the influence of MLC upstream of the STP becomes insignificant, i.e. it is virtually dry by mid summer. The source of true colour, an indicator of organics, appears to be both the STP and MLC. Given the lack of gradient, significant wetlands and aquatic vegetation in MLC downstream of the STP, this is not surprising.

Some data were unexpectedly high for varying reasons. Though still relatively low, the 22 ug/L ammonia value at Trent u/s Bloedel was anomalous, and probably a result of a chance capture of wildlife influences in the area. Exceedences of the TSS Guidelines at MLC@Cumberland Rd may be due to sediment disturbances during sampling. Very low water levels and presence of iron hydroxide precipitate at this location make it difficult to sample without disturbing the sediment (Anderson pers. comm., 2006). Two anomalous phosphorous peaks in the Village of Cumberland's automated sampler data were observed when no discharge was released. In the summer months, there is very little flow in the downstream reaches of MLC (Anonymous, 2003). Peaks in phosphorous during periods of no STP discharge were expected to be attributed to rainfall events; however, Environment Canada historical weather data indicates no rainfall in the area 72 hours up to and including the dates in question. It may be possible that these peaks were just chance excess nutrients in one particular sample. It is recommended that upon such unexpected observations, it be requested that the lab re-analyse the sample for verification.

Notably, nitrate+nitrite, as well as *E. coli* and fecal coliforms, did not reach background levels in the lower part of the Trent in 2005. Nitrate+nitrite levels in the discharge were relatively high in 2005, thus much more dilution and biological uptake was likely required to reach background levels. In 2006 levels were much lower and background levels were almost reached downstream. The observation of higher *E. coli* and fecal coliform levels downstream is not as expected when considering the clear dilution trends of all other parameters in the Trent. These higher levels may be due to animal contamination, though the only way to determine this is through bacterial source tracking techniques. This aspect may be further considered if funding allows, but was not the main focus of this study.

According to the phosphorous curve presented in Bothwell *et al.* (1992) (Fig 2) and phosphorous data collected in this study, chl α levels observed in this study could have been much higher than observed in all areas of the Trent downstream of MLC. This illustrates the influence of other natural environmental factors which influence the growth and accumulation of algae in streams (Nordin, 1985). In most coastal streams, orthophosphate is the limiting factor controlling algal accumulation and growth. However, in the Trent downstream of MLC, phosphorus levels are significantly elevated through the critical summer months. This results in elevated chlorophyll α levels throughout the lower Trent River. However, the physical characteristics of the Trent limit algal growth and accumulation. It is the steep sided nature of much of the channel, and the riparian vegetation, which limits the amount and duration of light exposure along the river. This in turn becomes the limiting factor to algal growth through much of the Trent River below MLC; however, despite the light limitation, algal growth and accumulation remains significant downstream of MLC.

The significant correlation of 2005 half-piece chl α values with original artificial chl α substrate was as expected considering the experimental nature of the first year of the pulsed discharge study. A successful decrease in algae by way of a successful pulsed discharge was expected to be reflected through a minimized or lack of relationship between the original and half-piece data.

Higher chl α values on the artificial than on the natural substrate suggest that the Styrofoam provides a better substrate texture for algal growth than do natural river rocks. The texture of the styrofoam causes an underestimation of the surface area during sampling which in turn overestimates the extent of algal growth on the river, and should be kept in mind in such studies. The peak natural substrate chl α value in early July 2005, that was much higher than all other values for that year, suggests the potential of human bias. Non-random sampling of rocks by individuals in the field and/or in-stream positioning of the block could have affected the anomalous data point. These types of errors could be minimized by more defined sampling protocols and increased replicates, but are likely unavoidable due to the natural variation of the river.

Clearly, the historical and 2005 chl α exceedences of both the aesthetic and recreation, and the aquatic life Guidelines indicate that an algal growth problem continues to exist in the Trent downstream of the Village of Cumberland STP. Lower values in chl α values in 2006 can be attributed to lower orthophosphate levels. In turn, lower orthophosphate levels were surprising given that orthophosphate in the discharge was comparable to that of previous years and VOC summer discharge flows (MOE files unpublished) were similar to previous years and actually higher than those in 2005. The cause of the reduced orthophosphate levels in 2006 should be investigated further, as these clearly resulted in a slight decrease in chl α in the Trent. Lower natural substrate chl α values in the 1-2 km d/s MLC section of the river reflect the narrowed geometry and higher flows of the river in these areas. Though minimal velocity data was collected during this study, higher velocities observed historically and in this study also support these observations. Data support the algal growth problem to be exacerbated in the wider, exposed lower reaches of the river, and in August when flows are usually lowest.

Dye testing

The low recovery rate of RWT could be attributed to the non-conservative behavior of RWT (it does not behave exactly like a water molecule) (Weiler pers. comm., 2006). However, this cannot account for all losses of RWT in the system. It was presumed that the recovery rate of the RWT represented the probability of a molecule not interacting with the hyporheic zone or ground water throughout the length of the stream gauged. This presumption was made because there did not appear to be a very high organic content in most of the Trent downstream of MLC. The only exception was at 400 m d/s MLC, where true colour data reflected high organic content. Therefore, for the majority of the Trent system, there was nothing for the RWT to adsorb to, unless adsorbing to particles in the streambed and surrounding soils. The low recovery rates brought to light the prevalence of hyporheic exchange in the Trent River. This was supported by the discharge measurements taken as they show there was substantial loss to ground water in the upper half of the Trent, followed by inflow of groundwater in the lower half.

This finding would indicate that a timing procedure at the sewage treatment plant would not be useful. A portion of the effluent, particularly during the summer low flow period, would enter the hyporheic zone, or ground water, in which there would be a much higher residence time. Then, the nutrients would be continuously re-entering the stream over time, prolonging the residence time, and confounding attempts to pulse the STP discharge.

The lack of RWT showing up in MLC was inconclusive. The creek appeared to have at least one fairly extensive beaver dam and a lot of in-stream vegetation. It is possible that all of the RWT adsorbed to organics in this situation. However, considering the low flow velocities, the concern is that there may be a large amount of hyporheic exchange. If there was substantial hyporheic exchange, it can be expected that some of the nutrients from the sewage treatment plant are entering the ground water. If this is the case, the nutrient path from the STP to the lower Trent may not be as direct as anticipated. The ground water could be mixing with the stream water at any point along the Trent River. Also, travel and residence time in this situation would be such that a pulsed discharge would not be sufficient to control nutrient levels in the lower Trent.

As the Village of Cumberland continues to explore options for decreasing the nutrient loading to the Trent River during the summer months, the groundwater regime and the interaction of groundwater and surface water flows in MLC and the Trent need to be better understood. The hyporheic exchange may influence the success of treatment options such as engineered wetlands.

Conclusions

Our results show that a timed discharge treatment will not be plausible to keep phosphorus levels down in the Trent River. The travel and residence time is much too long in MLC to be detected. Also, there is significant interaction with the groundwater and hyporheic zone, which prolongs residence times, as well as creating multiple pathways for the phosphorus to reach the area of concern. The concern is also raised that a man-made wetland may exacerbate the problem by creating more area and lower flows for surface-groundwater interactions. It is recommended that monitoring the groundwater for nutrients take place. Also, more intensive discharge measurements along the length of the streams should be done to establish the significance of surface-groundwater interactions. The cause for lower nutrient values in 2006 in the Trent needs to be investigated further.

Acknowledgements

We would like to thank the Village of Cumberland and Anderson Civil Consultants Inc. for their support and participation in this project, Markus Weiler and Stephanie Ewen at UBC for their contributions to the dye testing portion of the study, and MOE Fisheries staff for their insights and comments.

References

Anderson, D. 2006. Personal communication.

Anonymous, 2003. *Village of Cumberland Liquid Waste Management Plan Stage 2*. Anderson Civil Engineering, Nanaimo, B.C.

BCME (British Columbia Ministry of Environment*).* Science and Information Branch. 2006. *British Columbia Approved Water Quality Guidelines: 2006 Edition*.

Bothwell, M.L. 1988. Growth rate response to lotic periphytic diatoms to experimental phosphorus enrichment: the influence of temperature and light. *Canadian Journal of Fisheries and Aquatic Sciences* **45**: 261-270.

Bothwell, M.L., Derksen, G., Nordin, R.N. & Culp, J.M. 1992. Nutrient and grazer control of algal biomass in the Thompson River, British Columbia: A case history of water quality management. Eds R.D. Robarts and M.L. Bothwell. Natl Hydr. Res. Inst. Symposium Series 7, Environment Canada, Sask., Sask. pp. 254-266.

Dodds, W.K., Smith, V.H. & Lohman, K. 2002. Nitrogen and phosphorus relationships to benthic algal biomass in temperate streams. *Canadian Journal of Fisheries and Aquatic Sciences* **59**: 865-874.

Elser, J.J., Marzolf, E.R. & Goldman C.R. 1990. Phosphorus and nitrogen limitation of phytoplankton growth in the freshwaters of North America: a review and critique of experimental enrichments. *Can. J. of Fish. and Aq. Sci.* **47**: 1468-1477.

Environment Canada. 2006. http://waterquality.ec.gc.ca/EN/home.htm

Mainstone, C.P. & Parr, W. 2002. Phosphorus in rivers – ecology and management. *The Science of the Total Environment* **282-283**: 25-47.

Ministry of Environment. 2005. *British Columbia Environmental Laboratory Manual 2005 Edition*. British Columbia Ministry of Environment.

Ministry of Environment. 1998. *Manual of Standard Operating Procedures for Hydrometric Surveys in BC.* Resouces Inventory Branch. British Columbia Ministry of Environment.

Nordin, R*. Water Quality Criteria for Nutrients and Algae. Technical Appendix.* Water Management Branch, B.C. Ministry of Environment. October 1985.

Vannote, R.L. & Sweeney, B.W. 1980. Geographic analysis of thermal equilibria: A conceptual model for evaluating the effect of natural and modified thermal regimes on aquatic insect communities. *The American Naturalist* **115**(5): 667-695.

Weiler, M. 2006. Personal communication.

Wetzel, R.G. 2001. *Limnology Lake and River Ecosystems 3rd Ed*. Academic Press. USA.

Wilcock, R.J., Nagels, J.W., McBride, G.B., Collier, K.J., Wilson, B.T. & Huser, B.A. 1998. Characterisation of lowland streams using a single-station diurnal curve analysis model with continuous monitoring data for dissolved oxygen and temperature. *New Zealand Journal of Marine and Freshwater Research* **32**: 67-79.

APPENDIX 1

2002-2004 historical data

Water chemistry

Of the 2002-2004 water chemistry parameters considered in this study, all but pH showed very low or below MDL levels at Trent River sites upstream of the confluence of MLC. The pH (mean 8.0 SE 0.05) stayed relatively stable throughout the dataset with no apparent trends except a very slight decrease from 2002-2004. Below the confluence of MLC, all but pH and ammonia showed a clear decreasing trend with distance from the confluence. The dilution effect of the Trent River progressed downstream, reaching background levels for all but pH and ammonia by 2.5 km d/s MLC or 500 m u/s Trestle. Dilution occurred at a mean rate of approximately 20% per km.

True colour
True colour was less than or equal to 10 TCU above MLC, and peaked at 40 TCU downstream (Fig 23). Peak values downstream of MLC were at 400 m downstream of MLC, the first site after the confluence.

Turbidity
Turbidity ranged from 0.14 to 0.35 NTU upstream of MLC, and peaked at 2.34 NTU downstream (Fig 24). Peak values downstream of MLC were at 400 m downstream of MLC, the first site after the confluence.

Ammonia
Ammonia was below the MDL of 5 ug/L upstream of MLC, and peaked at 16 ug/L downstream (Fig 25). At three downstream sites, ammonia levels in 2003 were an average of 1.9 times higher than in both 2002 and 2004.

Nitrate+nitrite
Above MLC nitrate+nitrite ranged from 143 to 372 ug/L, and below MLC ranged from 331 to 1480 ug/L (Fig 26). With the exception of the Cumberland STP, in all years the highest nitrate+nitrite levels occurred at MLC u/s Trent (>3000 ug/L).

Orthophosphate
Orthophosphate was also less than or close to the MDL of 1 ug/L above MLC, and peaked at 101 ug/L downstream of MLC (Fig 27). Orthophosphate showed a decreasing trend from 2002-2004.

Total phosphorous
Total phosphorus levels were less than the MDL of 2 ug/L above MLC, and peaked at 266 ug/L below MLC (Fig 28). Total phosphorus showed a decreasing trend from 2002-2004.

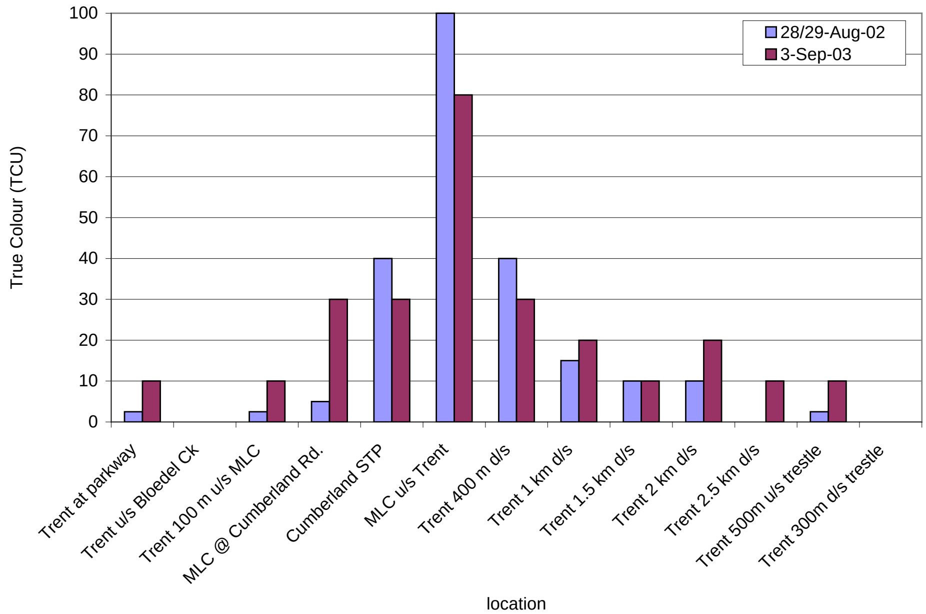


Fig 23 – True colour over two sample dates from 2002-2003 in the Trent River and MLC.

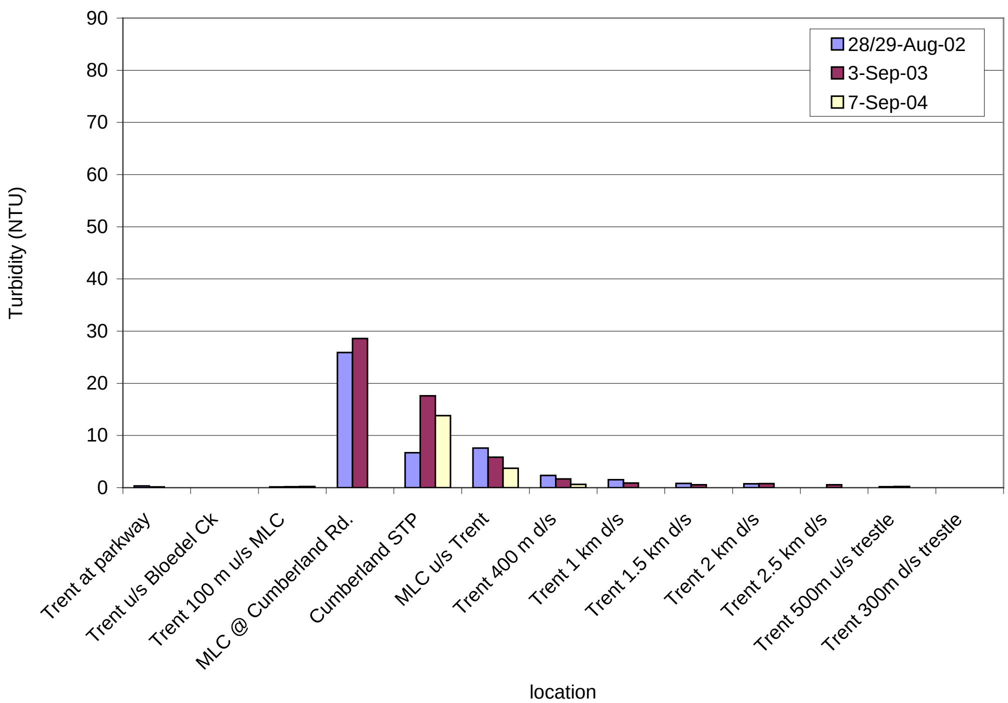


Fig 24 – Turbidity over three sample dates from 2002-2004 in the Trent River and MLC.

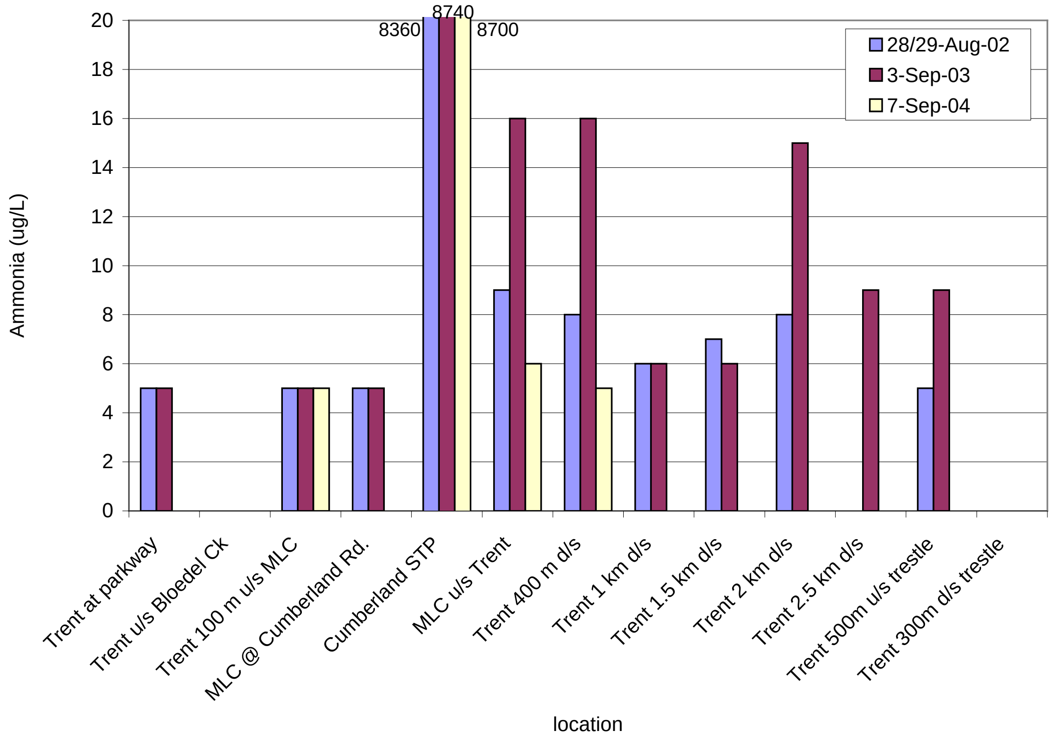


Fig 25 – Ammonia over three sample dates from 2002-2004 in the Trent River and MLC.

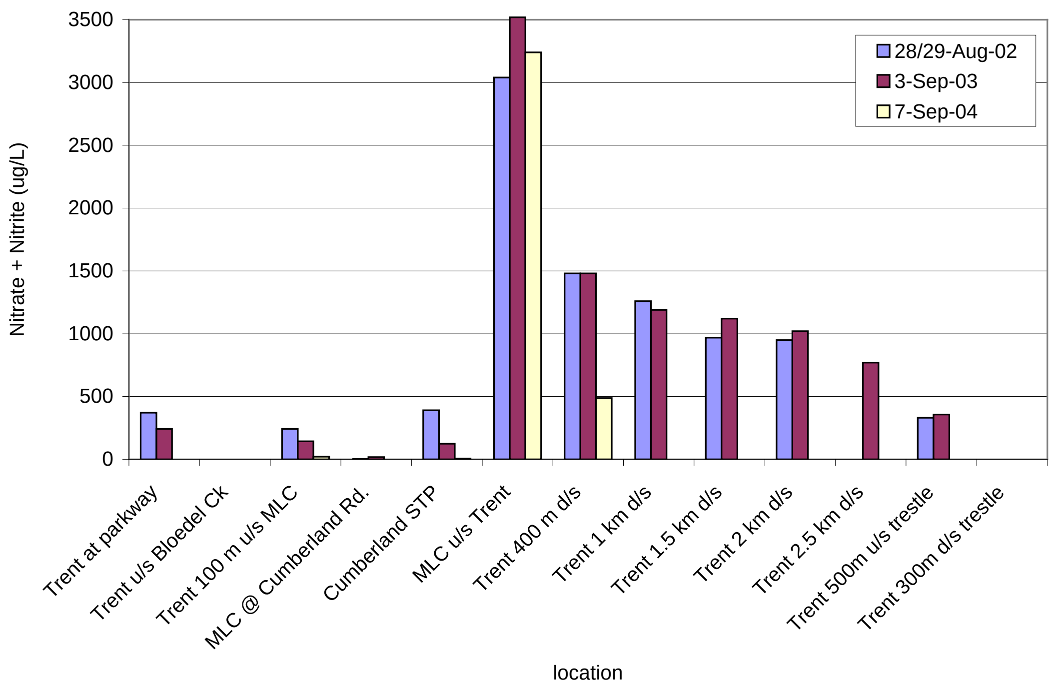


Fig 26 – Nitrate+nitrite over three sample dates from 2002-2004 in the Trent River and MLC.

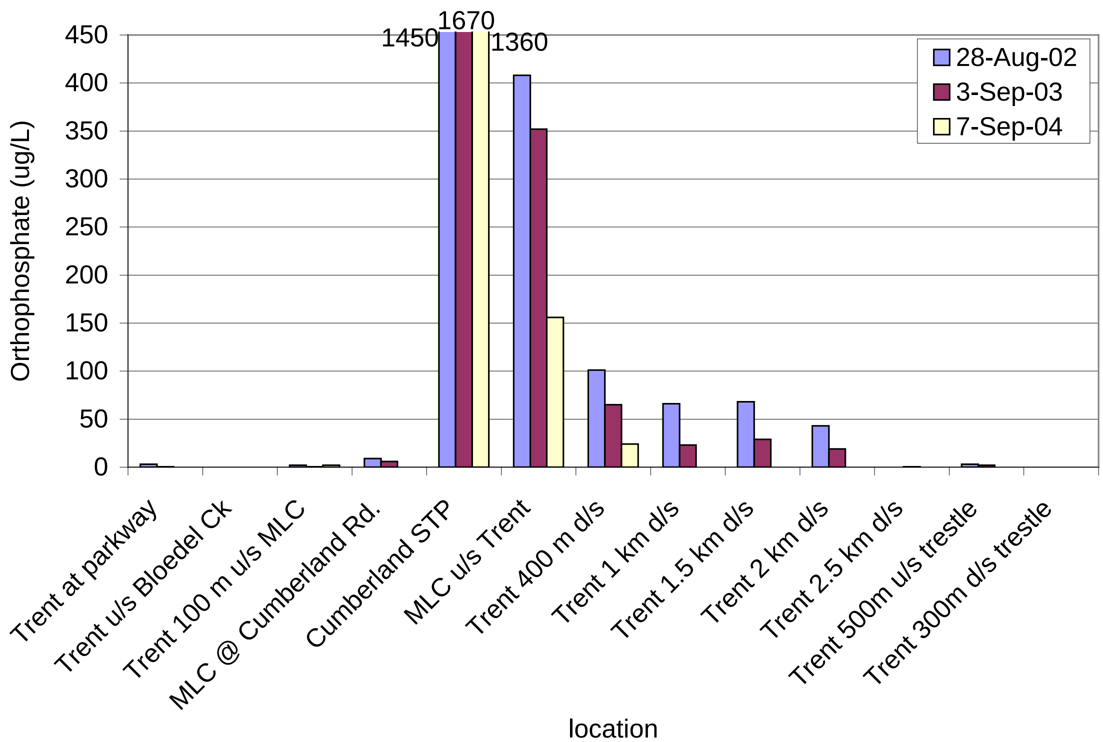


Fig 27 – Orthophosphate over three sample dates from 2002-2004 in the Trent River and MLC.

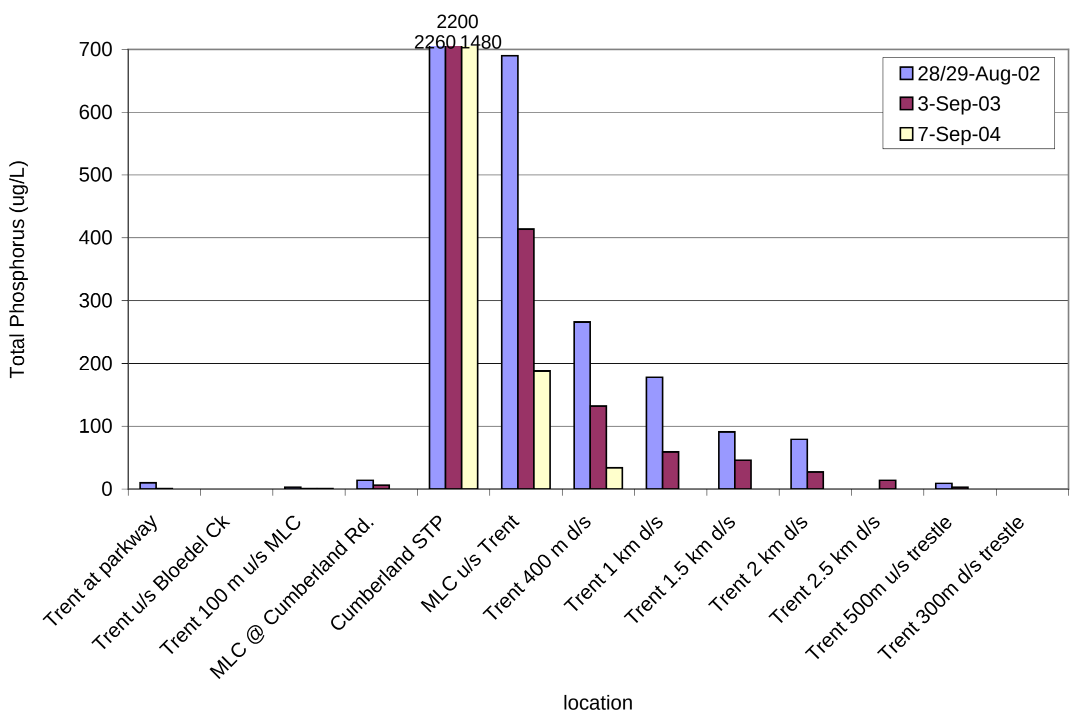


Fig 28 – Total phosphorous over three sample dates from 2002-2004 in the Trent River and MLC.

MLC@Cumberland Rd (upstream of the STP) and the Cumberland STP did not show trends similar to the Trent River sites or MLC u/s Trent. Both these unique sites generally had turbidity levels higher than all other sites (6.7-28.6 NTU). True colour levels were similar to those at 400 m d/s MLC. Though MLC@Cumberland Rd was similar to upstream sites for ammonia, orthophosphorus and total phosphorus, Cumberland STP had extremely elevated levels of all these parameters (>8000, >1000, >1400 respectively). Contrastingly, nitrate+nitrite levels at both sites were comparable to upstream sites.

Algal biomass

Chlorophyll α
Natural substrate chl α levels in 1997 and 1998 were only collected at 100 m u/s MLC and 400 m d/s MLC (Fig 29). Those at the downstream sites exceeded aquatic life Guidelines (100 mg/m^2) at both sample dates in 1997, and at one sample date in 1998. Algal biomass was observed at a maximum of 552 mg/m^2 in September 1997.

Natural substrate chl α decreased by an average of 60% from 2002 to 2003 at all sites sampled (Fig 29). In 2002, two of the six sites sampled (100 m u/s MLC at 6 mg/m^2) and1 km d/s of MLC at 37.3 mg/m^2) were below the aesthetic criterion of 50 mg/m^2. Two other sites, 1.5 km downstream of MLC and 500 m u/s trestle, had chl α levels of 119 mg/m^2 and 129 mg/m^2, respectively, and thus exceeded the aquatic life criterion. In 2003, all sites were below the aesthetic criterion.

Unlike with the other water quality parameters, no sites downstream of MLC reached upstream (background) levels in 2002 or 2003; rather, chl α at sites in the lower reaches of the Trent were on average nearly 6 times upstream levels.

Village of Cumberland 2005 Chl α data
Village of Cumberland natural substrate chl α values at 400 m d/s MLC did not exceed 18.0 mg/m^2 in 2005 during the April-October sampling period (Fig 30). MOE and Cumberland data could not be directly compared as dates of data collection were not always exactly the same. However, on dates falling within the same one week time period, MOE data at 400 m d/s MLC were higher than Cumberland's data on all three such dates. At 2 km d/s MLC, Cumberland's data were higher on 2 of 3 dates. April and October data indicate that algal growth at 400 m d/s MLC could be as high as or higher than August growth.

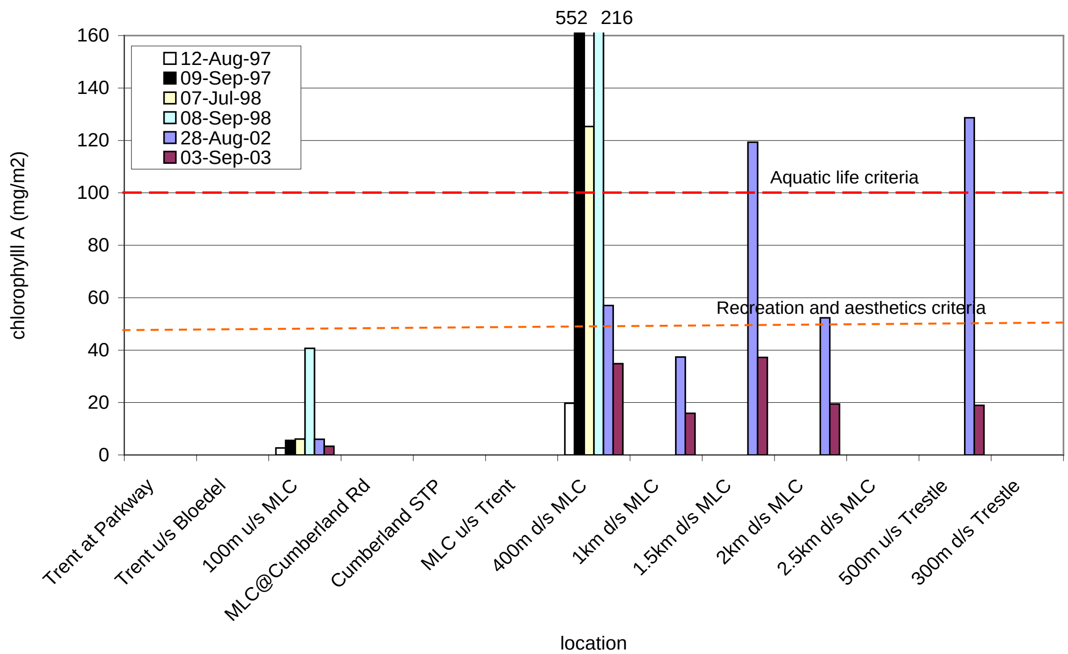


Fig 29 – Chl α natural substrate over six sample dates from 1997-2003 in the Trent River and MLC.

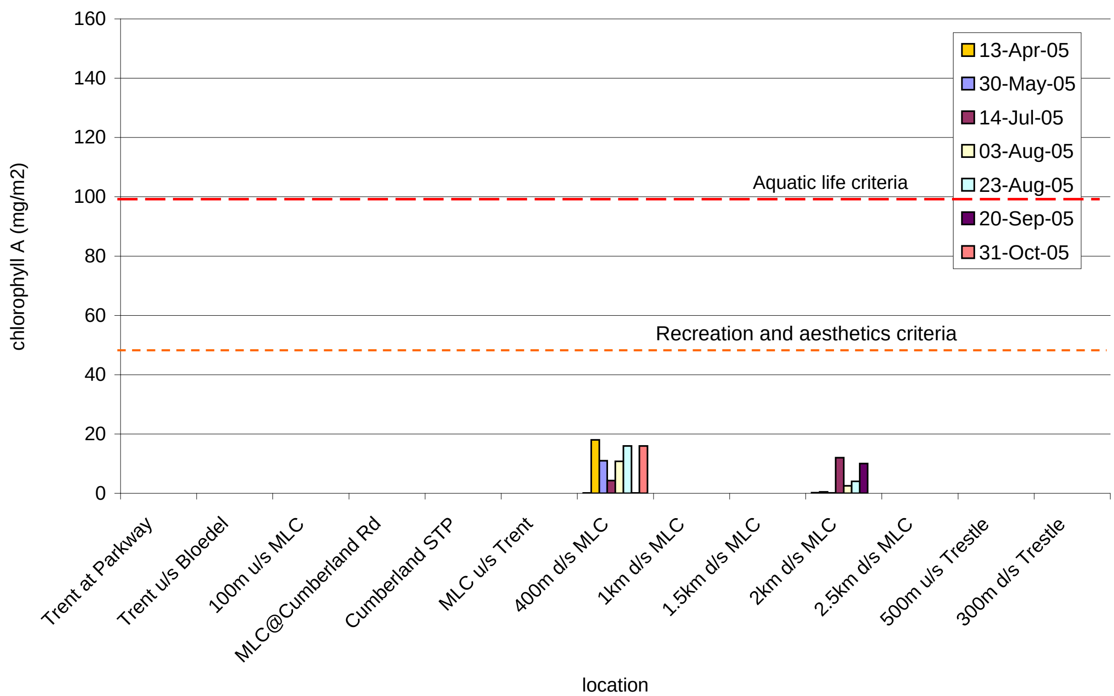


Fig 30 – Village of Cumberland (VOC) chl α natural substrate from April – October 2005 in the Trent River.