uOttawa

L'Université canadienne
Canada's university

FACULTÉ DES ÉTUDES SUPÉRIEURES ET POSTDOCTORALES

uOttawa

L'Université canadienne
Canada's university

FACULTY OF GRADUATE AND POSTDOCTORAL STUDIES

Leiserl Woods
AUTEUR DE LA THÈSE / AUTHOR OF THESIS

M.Sc. (Earth Sciences)
GRADE / DEGREE

Department of Earth Sciences
FACULTÉ, ÉCOLE, DÉPARTEMENT / FACULTY, SCHOOL, DEPARTMENT

Using Ecohydrology to Predict Algal Biomass in the Raisin River Watershed (Ont. Canada)

TITRE DE LA THÈSE / TITLE OF THESIS

M. Robin
DIRECTEUR (DIRECTRICE) DE LA THÈSE / THESIS SUPERVISOR

F. Pick
CO-DIRECTEUR (CO-DIRECTRICE) DE LA THÈSE / THESIS CO-SUPERVISOR

EXAMINATEURS (EXAMINATRICES) DE LA THÈSE / THESIS EXAMINERS

D. Fortin

F. Michel

J. Ridal

Gary W. Slater
Le Doyen de la Faculté des études supérieures et postdoctorales / Dean of the Faculty of Graduate and Postdoctoral Studies

USING ECOHYDROLOGY TO PREDICT ALGAL BIOMASS IN THE RAISIN RIVER WATERSHED, (ON. CANADA)

LIESERL M. E. WOODS

Thesis submitted to the
Faculty of Graduate & Postdoctoral Studies
University of Ottawa
in partial fulfillment of the requirements for the
M. Sc. degree in Earth Sciences

Ottawa-Carleton Geoscience Centre
and
University of Ottawa
Ottawa, Canada

Thèse soumise à
Faculté des etudes supérieures et postdoctorales
Université d'Ottawa
en de l'obtention de la maîtrise en sciences en
Sciences de la Terre

Centre Géoscientifique d'Ottawa-Carleton
et
Université d'Ottawa
Ottawa, Canada

Your file Votre référence
ISBN: 978-0-494-58205-3
Our file Notre référence
ISBN: 978-0-494-58205-3

Table of Contents

List of Tables

List of Figures

Abstract

The effects of groundwater and environmental variables on river ecology in terms of algal biomass (benthic and suspended) were determined across a watershed dominated by agricultural land-use (Raisin River, eastern Ontario). At the watershed scale, during summer base flows, light and temperature or Strahler stream order predicted suspended algal biomass (estimated by chlorophyll a) whereas turbidity and temperature predicted epilithic periphyton biomass in riffle zones; nutrients did not correlate with either algal community. Benthic algal biomass was negatively correlated with suspended algal biomass. At the reach scale, periphyton biomass and total phosphorus were temporally linked but the relationship became uncoupled by spates. Evidence for shallow hyporheic flow but not deep groundwater discharge was reported for one of two reaches studied. Benthic algal biomass accrual increased linearly with both a positive or negative hydraulic gradient, indicating that surface water/ groundwater interactions were important.

Résumé

Les effets des eaux souterraines et de variables environnementales sur l'écologie des cours d'eau en ce qui a trait à la biomasse d'algues (benthiques et planctoniques) ont été évalués dans un bassin versant à vocation agricole (la rivière Raisin dans l'est de l'Ontario). À l'échelle du bassin en période d'étiage, la lumière et la température (ou l'ordre de Strahler) pouvaient prédire, au cours de l'été, la quantité de biomasse d'algues suspendue; tandis que la turbidité et la température pouvaient prédire la quantité de périphyton épilithique des radiers; la quantité de nutriments n'était pas corrélée avec l'une ou l'autre communauté. Les quantités de biomasse benthique et planctonique était négativement corrélées. Cependant la quantité de biomasse périphytique et le phosphore total étaient corrélés dans le temps à l'échelle d'un tronçon mais le lien était absent lors de crues. Des indices d'écoulement hyporhéïque peu profond mais pas de décharge profonde d'eaux souterraines ont été observées pour un des deux tronçons étudiés. L'accroissement linéaire de la biomasse benthique en présence d'un gradient hydraulique positif ou négatif, indique que les interactions eaux de surfaces/ eaux souterraines étaient importantes.

Aknowledgements:

Funding for this study: The resources for this study were first provided by the Canadian Water Network to fund a holistic study of the Raisin River watershed. The group that emerged, the Watershed and Environmental Research Assessment Project (WERAP), aimed to produce an integrated view of the watershed that combined a variety of studies such as groundwater/ surface water interactions (E. Bustros-Lussier 2008), biochemical nitrate transformations in farm fields (M. Suchy in progress), in-stream nutrient dynamics of the watershed (S. Maharaj 2007), atrazine effects on amphibian sexual development (V. Langlois in progress) and this study. By focussing research efforts on one watershed we were able to share resources, data and knowledge that were common to multiple studies, providing a more complete picture than would otherwise have been possible.

Thank you to Frances and Michel for your knowledge and dedication, to the "warm-bodied" EST graduate students for support and commiseration, to my friends and family for their patience, and to Nick for always being there.

Chapter 1: Introduction to thesis objectives

Traditionally, the effect of groundwater on stream and river ecology has been poorly understood, likely due to academic divisions between biological and geological sciences. Lately however, many watershed studies have combined these sciences, creating the need for a new discipline: ecohydrology. There has been some debate as to the definition and adoption of the term ecohydrology (Hannah et al. 2004), but one of the most widely used definitions is the study of the functional interrelations between hydrology and biota at the catchment scale (Zalewski et al. 1997). Surface water and groundwater interact in the shallow substratum beneath the sediment-water interface, termed the hyporheic zone. The hyporheic zone can be limited or vast, depending on the hydraulic conductivity of the sediments, but is constrained by the extent to which surface water mixes with the groundwater. At the reach scale, shallow hyporheic flow is generally produced by downward flow of surface water into the sediments at the upstream end of the riffle and an upward flow of hyporheic water at the end of the riffle. This thesis will investigate the effects of reach-scale hyporheic flow on river ecology by studying an important primary producer: algae.

In rivers, algae can be epilithic (bound to the substrate), also known as periphyton or benthic algae or can be suspended in the water column (Allan & Castillo 2007). Algae can proliferate and become a nuisance when environmental conditions change such as increased light from clear-cutting (Lowe et al. 1986) or changes in nutrient concentrations due to agricultural, industrial or urban inputs (Stockner & Shortreed 1978, Slavik et al. 2004). In water bodies with excess algal production, dissolved oxygen can be depleted to the detriment of fish and other aquatic species (Edwards & Owens 1962, Kramer 1987). While the importance of quantifying and predicting algal biomass in streams and rivers to determine the degree of eutrophication and trophic status is well recognized (Dodds et al. 2002), unlike lakes, there is still no consensus on which models are the most reliable.

Many studies have focussed on light, nutrients and current velocity to predict periphytic algal biomass (Allen & Castillo 2007) however the effect of groundwater discharge into the river has only recently been investigated (Pepin et al. 2002, Wright et al. 2005). As management efforts focus on reducing nutrient concentrations, there is equal evidence to suggest that in many streams and rivers, nutrients do not predict algal biomass (Jones & Barrington 1985, Reynolds 1988, Sullivan et al. 2001, Figueroa-Nieves et al. 2006) and that variables such as turbidity and hydrology should also be monitored.

This thesis will attempt to address the different parameters (both chemical and physical) that can affect algal biomass in a small, lowland, agriculturally-dominated watershed, in south-east Ontario: the Raisin River watershed. Chapter 2 will present the watershed's physical characteristics (topography, geology, climate, urban impacts and hydrology) to provide the context for the following chapters. In Chapter 3 algal biomass (epilithic and suspended) will be investigated at the watershed scale in relation to environmental variables.

Most research in groundwater- surface water interactions has been done in the shallow hyporheic (the uppermost few cm). There have been calls to broaden the perspective of interactions studied in the hyporheic to include spatial variability at different scales (Sophocleous 2002). Chapter 4 will examine groundwater/ surface water interactions at the reach scale,

There is reason to believe that groundwater/ surface water interactions could impact benthic algae by introducing upward or downward flow across the sediment interface and thereby interfering with laminar flow in the boundary layer. Diffusion limitation at the boundary layer has long been studied (Whitford 1960), where a nutrient-depleted layer forms around the cell and diffusion overtakes active transport as the rate-limiting step in nutrient uptake. Chapter 5 will investigate how river discharge and groundwater/ surface water interactions affect periphyton biomass and accrual at the reach scale in the Raisin River. Results from Chapter 5 have been published, (Woods et al. 2009).

Chapter 2. Watershed Characteristics

This chapter provides physical context to the research presented in the following chapters. It is meant to give background information on topography, geology, climate, urban impacts and hydrology of the Raisin River watershed. Figures and their interpretations are drawn heavily from the Water Budget: Conceptual Understanding for the Raisin – South Nation Source Water Protection Region, (Raisin- South Nation SWPR 2007) unless otherwise cited.

Site Location and Topography

The Raisin River watershed is located in South Eastern Ontario and flows in a south-easterly direction into the St. Lawrence River at Lancaster (Figure. 1). The topography has a gentle slope from the North-West to the South-East ranging from about 120 m.a.s.l. to about 40 m.a.s.l. by the St. Lawrence (Figure. 2). The terrain is hummocky in areas, but overall a gentle gradient sloping towards the St. Lawrence predicts that deeper regional groundwater would likely be attaining the St. Lawrence, while overland flow and shallow groundwater are likely greater contributors to all but the mouth of the Raisin River.

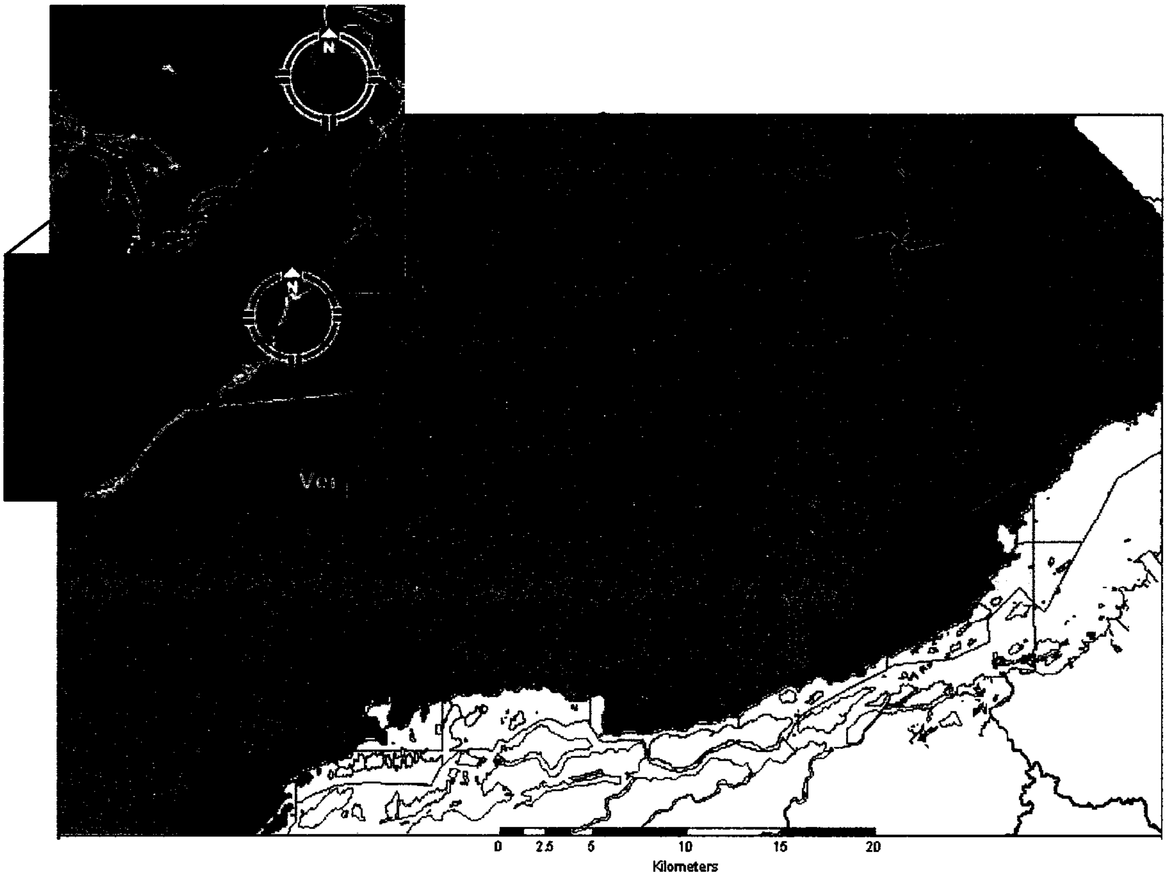


Figure 1. Subwatershed boundaries of the Raisin River watershed (subwatersheds outlined in green, river in dark blue). As defined by Raisin- South Nation SWPR (2007). Inlays from Google Earth.

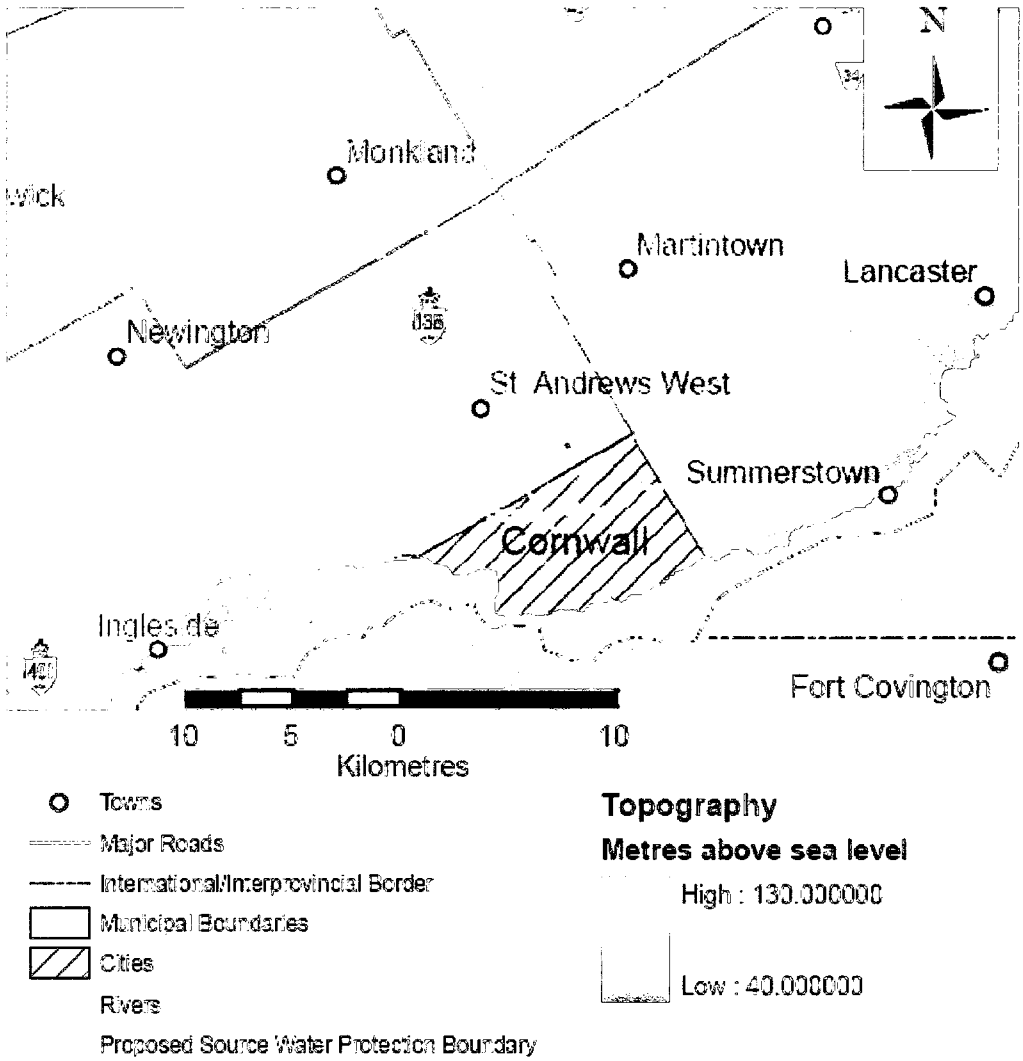


Figure 2. Topography of the Raisin River watershed. From Figure 11: Topography (Raisin-South Nation SWPR 2007).

The watershed is approximately 550 km^2 and contains three main branches, the North, Middle and South Raisin River that meet to form a 6th order tributary (Figure 3). Sites sampled in Chapter 3 throughout the watershed ranged from 3rd order to 6th (mostly 4th or 5th order branches). The two reaches studied in Chapters 4 and 5, St. Andrew's and Macintyre Rapids, were 4th and 5th order streams respectively. Streams of this mid-size, according to the serial discontinuity concept (Ward & Stanford, 1983), are highly influenced by meteorological events and show the most visible and unpredictable flow regimes. This becomes important in Chapter 3 where the regulation of algal biomass and accrual by flow, nutrients and discharge is investigated.

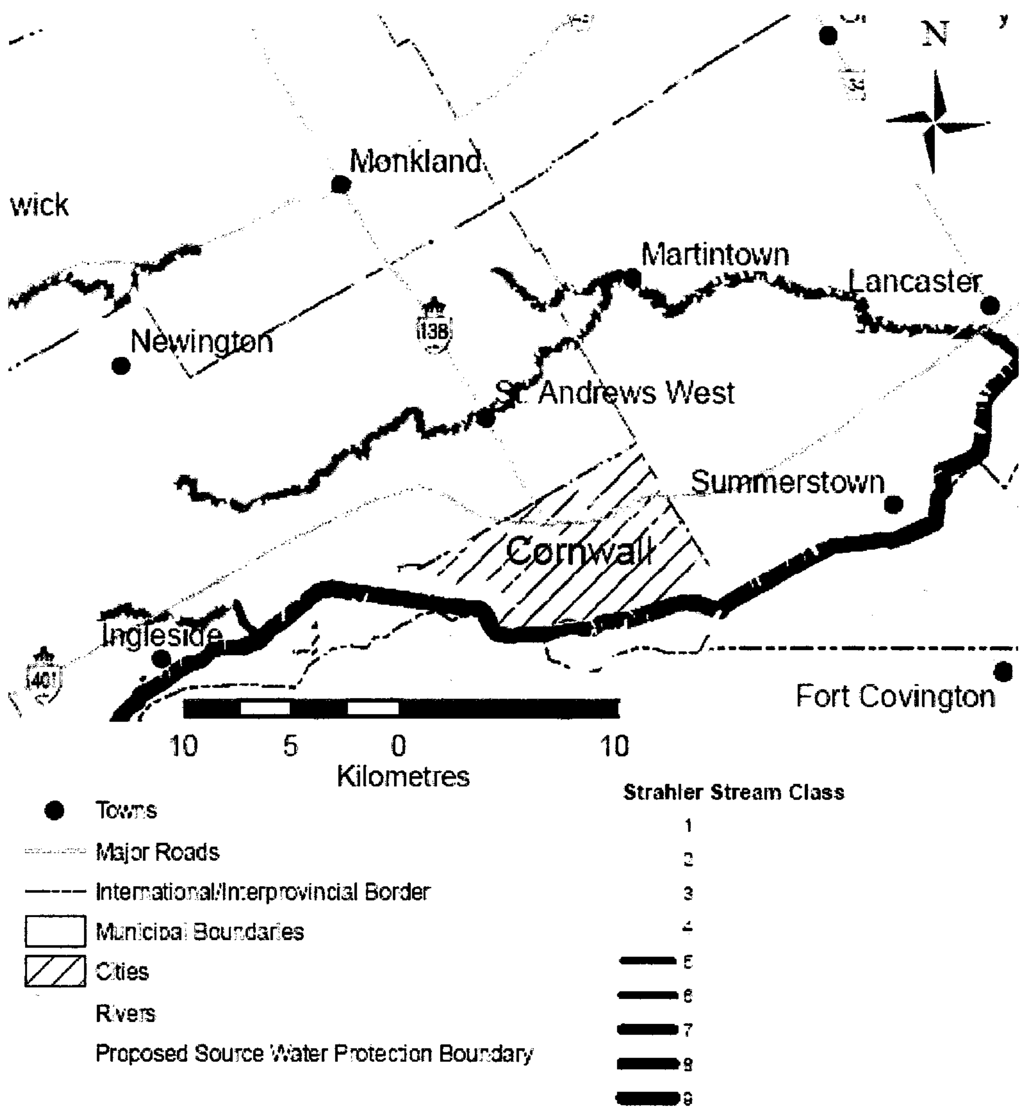


Figure 3. Stream Orders in the Raisin River watershed. From Figure 41: Strahler Stream Classification (Raisin- South Nation SWPR 2007).

Geology

The Raisin River watershed was eroded by the Laurentian ice sheet, and was later covered by the Champlain Sea between 11 500 and 10 000 years ago leaving behind marine sediments overlying glacial till and an overall hummocky landscape with little topography.

The North, Middle and South branches of the Raisin River are underlain by the Lower Lindsay formation; the Verulam formation; and the Bobcaygeon and Gull River formations, respectively (Raisin- South Nation SWPR 2007). These formations are Cambrian-Ordovician and are composed of limestone and shale except for the Gull River formation, which is also composed of dolostone. The major faults in the watershed run generally east/west, crossing the Middle Raisin River just upstream of Martintown and crossing the North + Middle three times between Martintown and Williamstown (Figure 4). A major fault intersects the Raisin about 1 km upstream of the Macintyre Rapids site studied in Chapter 4 and 5. This could potentially introduce deeper groundwater flow compared to other reaches.

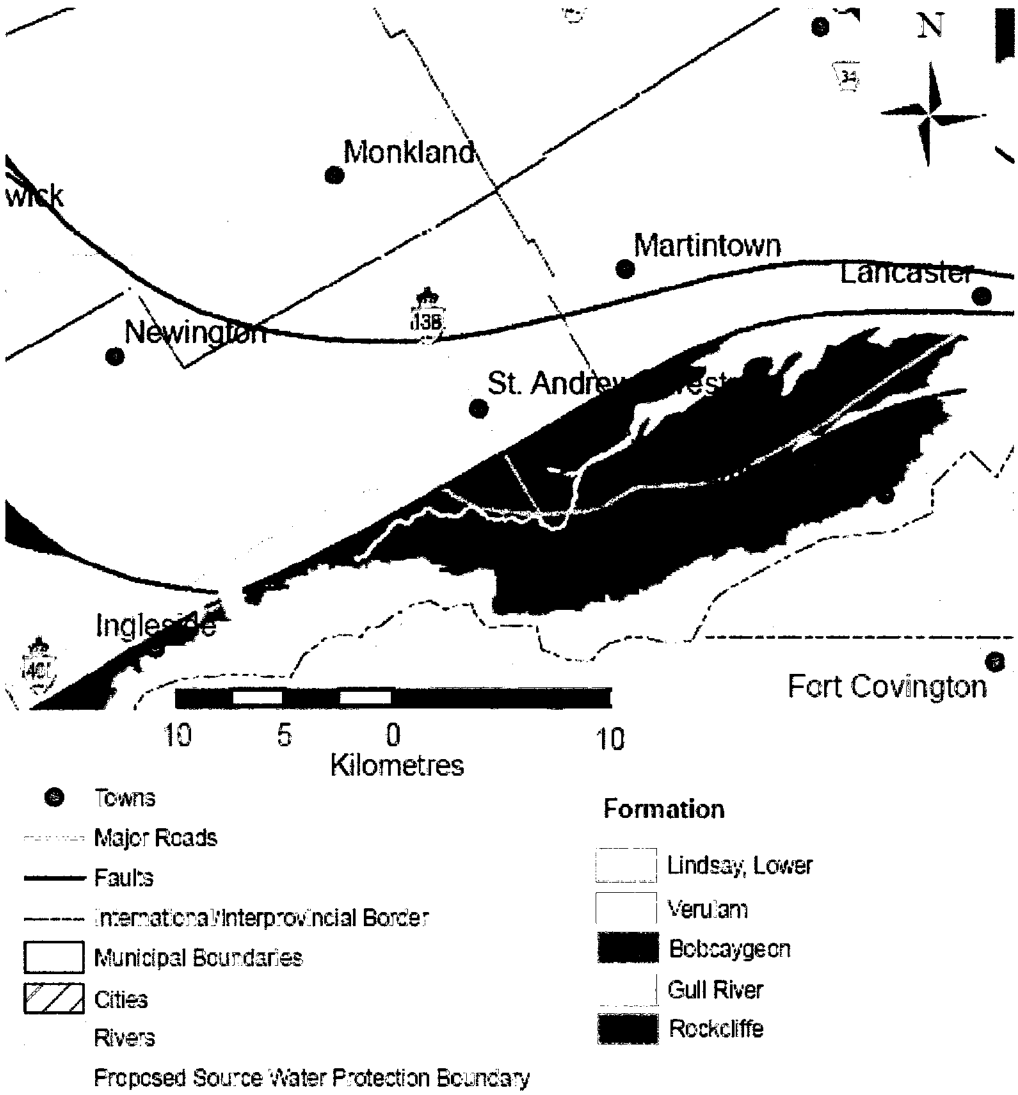


Figure 4. Bedrock formations and faults of the Raisin River watershed. From Figure 12: Bedrock Formations and Faults (Raisin- South Nation SWPR 2007).

The watershed lies mostly in the physiographic unit of the Glengarry Till Plain beginning at its headwaters and flows into the topographically lower Lancaster Flats toward the mouth at the St. Lawrence River (Figure 5). The surficial geology of the Raisin River is primarily Pleistocene glacial deposits of silty sand to silt till; with pockets of glaciomarine deposits and some more recent organic deposits especially near the Newington bog area (Figure 6). The upper reaches of the Raisin River watershed are dominated by silty sand and silt as well as the recent organic deposits while the lower reaches are dominated by sandy deltaic deposits.

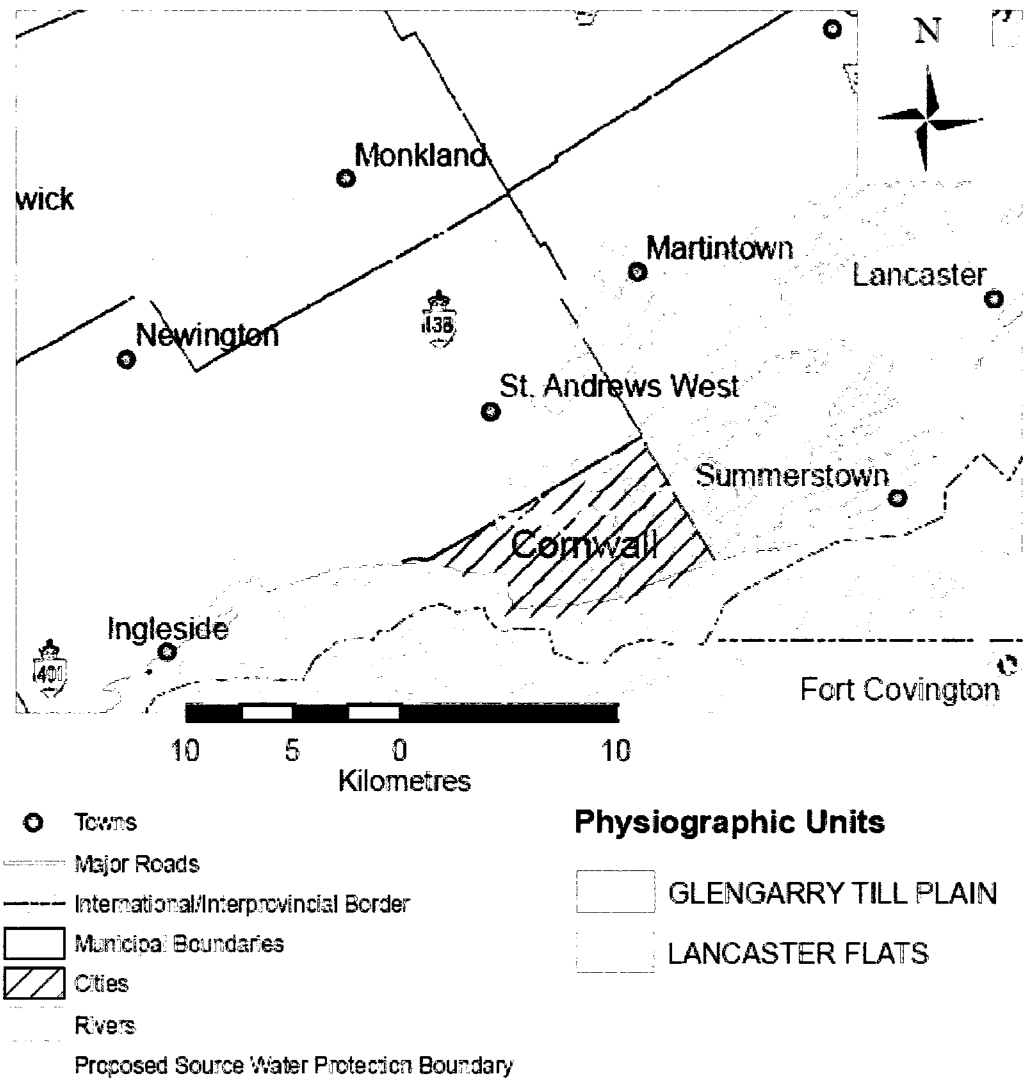


Figure 5. Physiographic units of the Raisin River watershed. From Figure 9: Physiographic Units (Raisin- South Nation SWPR 2007).

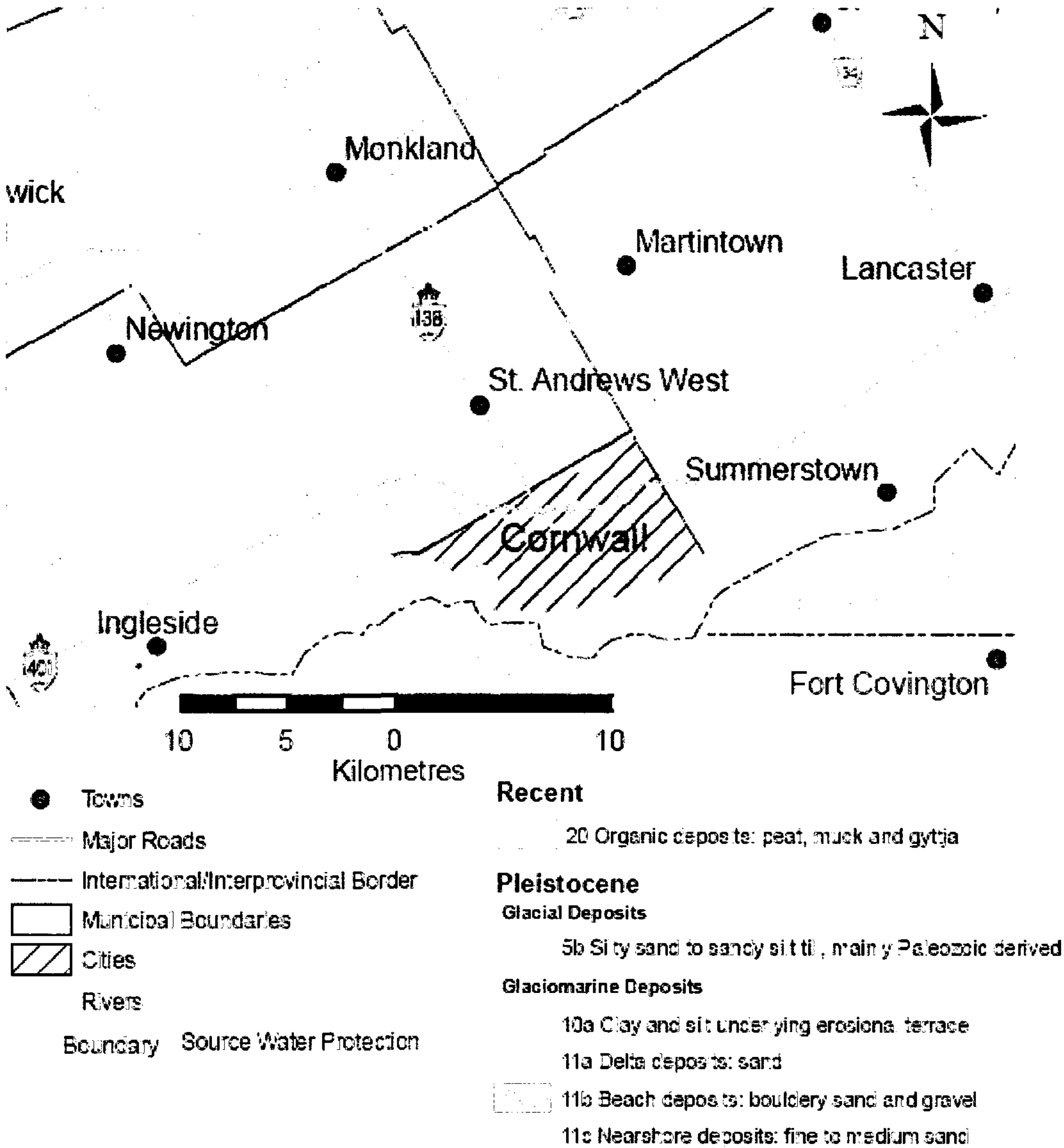


Figure 6. Surficial geology of the Raisin River watershed. From Figure 13: Surficial Geology (Raisin- South Nation SWPR 2007).

The main soils found in the watershed are primarily loam with clay along the river course, sandy loam near the mouth and muck pockets near the headwaters (Figure 7). The overburden thickness varies in the watershed from -18m (where significant erosion has occurred) up to 50m (Raisin- South Nation SWPR 2007).

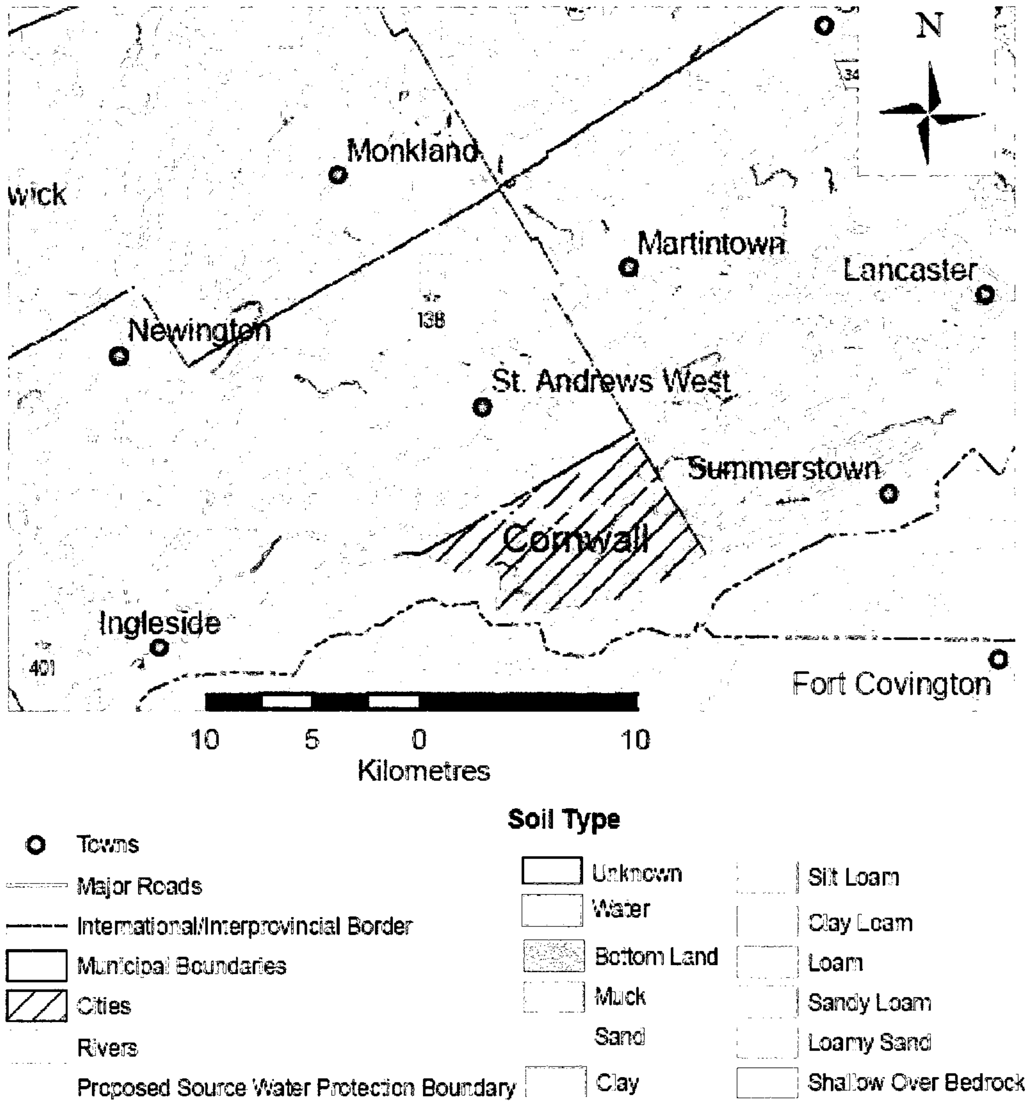


Figure 7. Soils of the Raisin River watershed. From Figure 8: Soils (Raisin- South Nation SWPR 2007).

Climate

The Raisin River watershed is characterized by a continental humid climate. Appendix B shows average climate normals for the Environment Canada Cornwall climate station from 1971-2000. The average daily temperature is 7.2 °C with daily maximums and minimums in the hottest month, July, reaching on average 26.7 °C and 16.4 °C respectively.

The coldest month, January, has a daily maximum and minimum of -4.6 °C -12.9 °C respectively. The average yearly precipitation is 794.8 mm of rain and 207.1 mm of snow. The Raisin River will freeze in patches over the winter months (January to mid-March).

Land cover in the Raisin River watershed is dominated by agriculture and deciduous, mixed and coniferous forests (Figure 8). Water loss due to evapotranspiration has been estimated for this region as between 579 and 664 mm/year (Raisin- South Nation SWPR 2007). Due to the forested cover, there is a high potential for evapotranspiration, especially in the growing months of June to August, which make up approximately 60% of the yearly loss due to evapotranspiration.

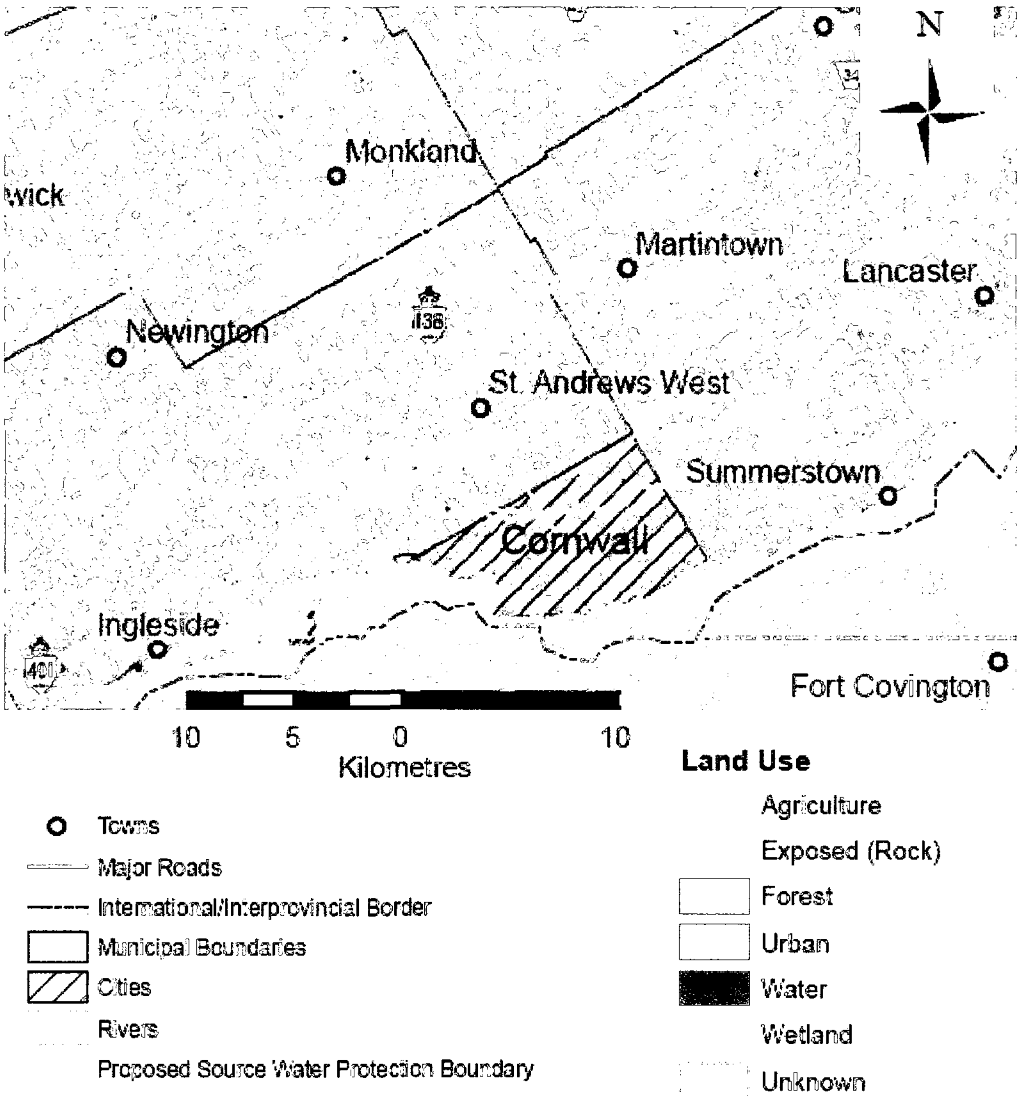


Figure 8. Land use of the Raisin River watershed. From Figure 7: Land Use (Raisin- South Nation SWPR 2007).

Urban Impacts

The municipality of Cornwall is located in the South-West of the Raisin River watershed with a population of 45 640 (Statistics Canada 2001). North Cornwall borders the South Raisin River watershed and so urban runoff and potentially contaminants such as road salts are expected contributions to the river.

The surrounding stretch of the St. Lawrence has been identified as an area of concern (AOC) by Environment Canada due to environmental issues such as industrial contaminants (mercury) and habitat destruction among others. A remedial action plan (RAP) has been in place since 1986 (Environment Canada 2007). The main industry in Cornwall has historically been pulp and paper, though in 2006 this industry closed down. A pulp mill was a contributor of mercury to the St. Lawrence River though in the final years of production, remedial action was taken to alleviate this environmental stressor. Other industries remain within the watershed, including a cheese factory located downstream of Williamstown. The sewer outflow from the City of Cornwall is located west of the city, in Long Sault where it is treated and then released to the St. Lawrence River.

The headwaters of the South Raisin River begin at the St. Lawrence River at Long Sault (45°1'46" N, 74°53'20" W), (upstream of the sewer outfall) where water is periodically diverted from the St. Lawrence in order to maintain flow to the South Raisin during base flow. This diversion was monitored from May 30th, 1972 until September 20th, 1996. Over the 25 years that the diversion was monitored, the amount of water diverted decreased from a peak yearly flow of about 0.7 m^3/sec down to less than 0.15 m^3/sec (Water Survey of Canada 2008). The peak yearly flow on average decreased by 0.024 m^3/sec per year, (R^2=0.9) with four years where virtually no flow released were removed from the data set (Appendix C).

The flow of water from the St. Lawrence might give cause for worry that the South Raisin River ecosystem would be affected by invasive species from the St. Lawrence. In fact one dead zebra mussel was found near the headwaters (Pichard et al. in Press), indicating that the St. Lawrence may have had some influence on the headwater river ecosystem.

Hydrology

During the summer months, when the ground is sufficiently saturated, an intense storm event can cause the river to flood its banks. The result of an extreme storm event is typically a dilution of the ionic concentration of the river water due to increased runoff (Golterman 1975).

The hydrological response to a precipitation event of a particular watershed can be characterized by the centroid lag time, T_{LC} which can be derived from a storm hydrograph by subtracting the time at which the centroid of the response hydrograph, t_{qc} from the centroid of the effective water input t_{we}.

$$T_{LC} = t_{qc} - t_{we} \qquad (1)$$

In addition to the centroid lag, the time of concentration T_C gives the time taken by precipitation that falls at the furthest reaches of the watershed to reach a selected downstream location given by:

$$T_C = t_{qe} - t_{we} \qquad (2)$$

where t_{qe} is the time at the end of runoff on the hydrograph and t_{we} is the time that precipitation ends (Dingman 2002).

Both the centroid lag and the time of concentration are watershed-specific parameters that do not depend on the timing or duration of precipitation input and so are useful to compare between watersheds.

Another way of characterizing the hydrology of a watershed is to look at the ratio of effective discharge (or event flow), Q_{ef} to total rainfall, W_t. For most watersheds, this ratio has been found to generally be lower than 0.5, with many watersheds as low as 0.1 (Dingman 2002). A lower ratio indicates that less rainfall is contributing to the effective discharge and thus is either infiltrated or evaporated before it reaches the stream. This approach assumes that the contributing area is constant and more rain gauges will give a better indication if this is indeed so.

Four storm hydrographs from the discharge gauge at the Williamstown Environment Canada station were analysed over the sampling period when both hourly precipitation data from the Cornwall Environment Canada rain gauge and hourly discharge data were available. The approximate drainage divide for the Williamstown discharge gauge is shown in Figure 9. Table 1 shows hydrograph characteristics for the four hydrographs analysed for 2005 and 2006 data sets obtained. Effective discharge, Q_{ef} to total rainfall, W_t is between 0.02 and 0.04, which is very low but could be explained by the low gradient of the watershed, and the prevalence of agriculture that requires high volumes of water such as corn. Appendix D contains the raw hydrograph data for each of the four storms analysed. Care was taken to exclude double-peaked hydrographs and hydrographs where precipitation occurred over the recession curve before base flow could be reached.

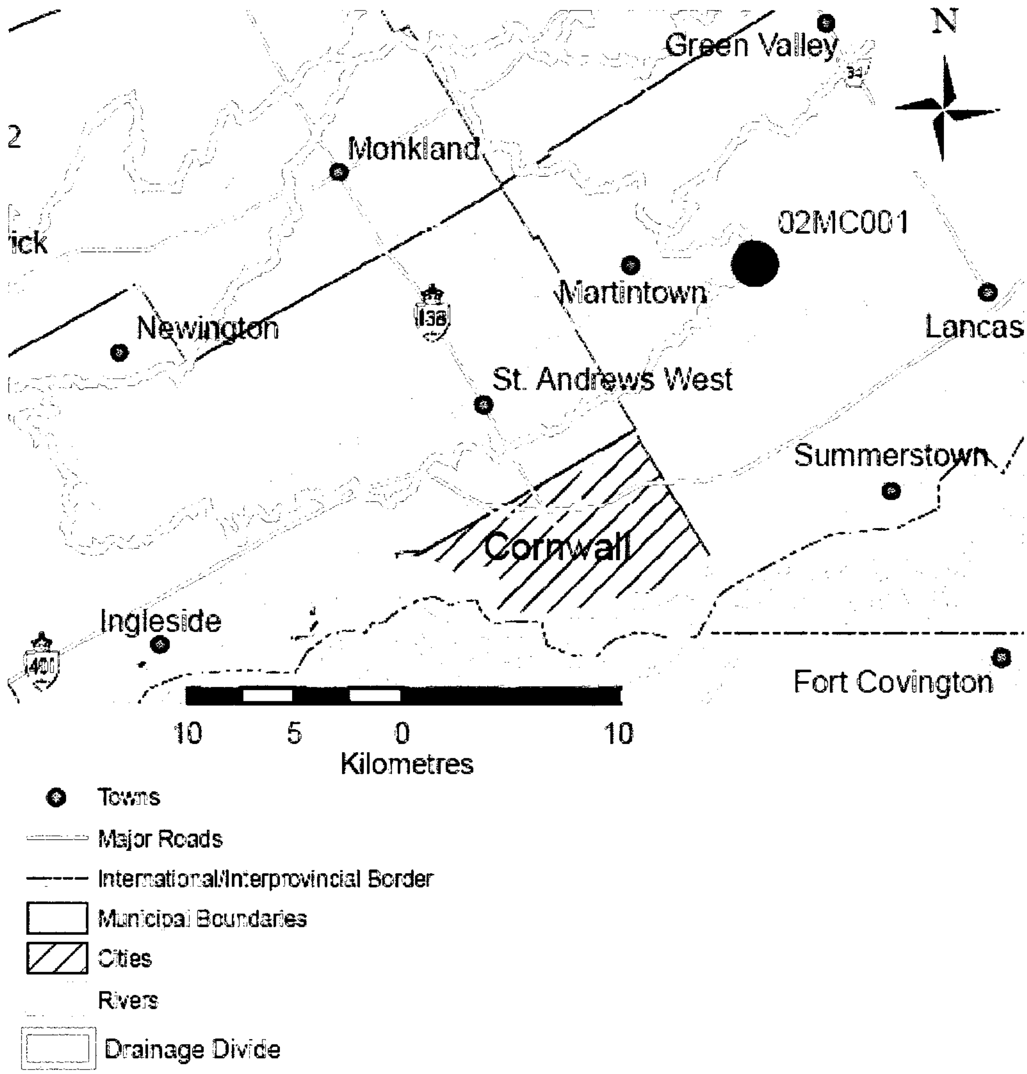


Figure 9. Approximate drainage divide for Williamstown stream gauge. From Figure 39: Approximate Drainage Divides of Selected WSC Stream Gauges (Raisin- South Nation SWPR 2007).

Table 1. Hydrological parameters of selected single-peaked hydrographs from the Raisin River watershed analysed for the Williamstown Environment Canada discharge gauge and the Cornwall Environment Canada precipitation gauge. t_{qef} is the duration of effective discharge (days), t_{win} is the duration of precipitation (days), T_c is time of concentration (days), T_{LC} is the centroid lag (days), Q_{ef} is the sum of effective discharge for the hydrograph, Wt is the total precipitation input.

Event	t_{qef}	t_{win}	T_C	T_{LC}	Q_{ef} (m^3)	W_t (mm)	W_t (m^3)	Q_{ef}/W_t
June 16, 2005	4.46	0.63	4.38	2.01	6112323	434	176048845	0.0347
June 30, 2006	4.08	0.54	4.67	2.63	1062275	170	68959225	0.0154
July 29, 2006	1.88	0.04	1.79	0.69	341956	24	9735420	0.0351
October 28, 2006	6.46	1.25	5.21	1.96	4337595	330	133862025	0.0324

Flows recorded at the Williamstown gauge from 1960 to 2004 range from 0 to 131 m^3/sec. Flows exceed 13 m^3/sec less than 10 percent of the time and fall below 0.1m^3/sec less than 10 percent of the time (Figure 10). The flow duration curve is fairly steep, which suggests that most discharge is due to runoff and not derived from groundwater discharge to the river. This corresponds well with the known surficial geology of mostly sandy silt deposits which would tend to have low dynamic storage, as found in other glaciated watersheds (Thomas 1966), (Figure 11).

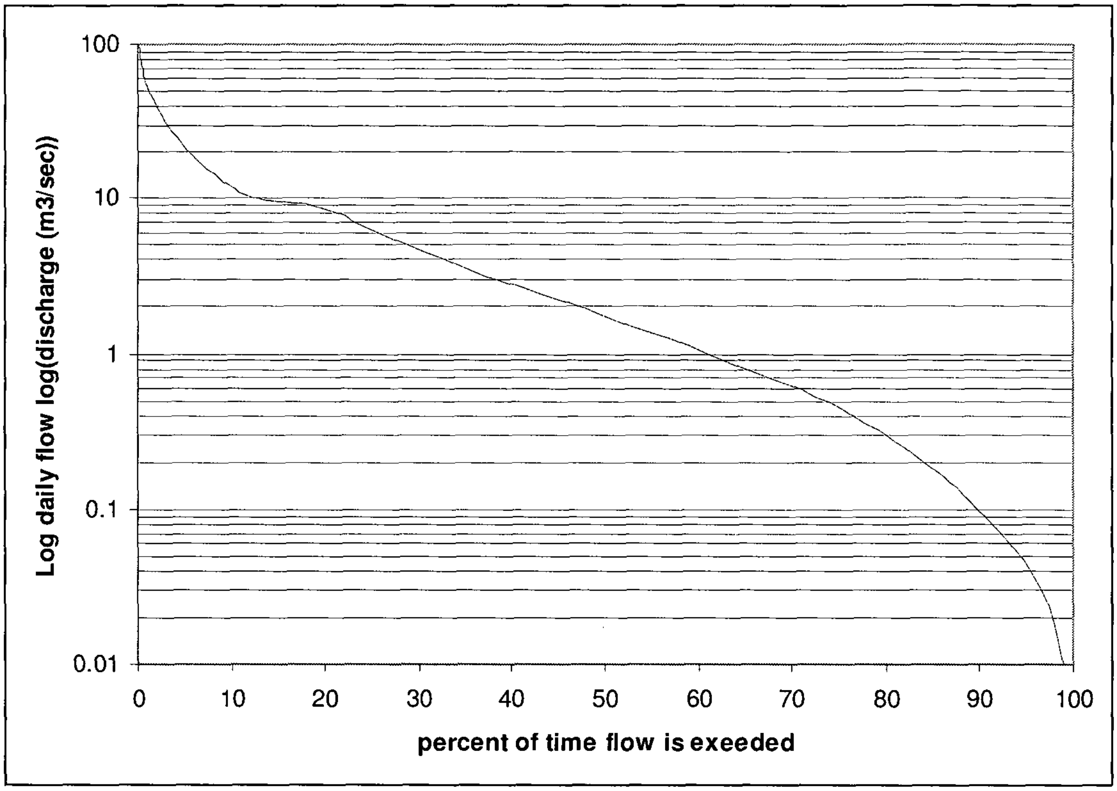


Figure 10. Flow-duration curve for the Environment Canada Raisin River gauge at Williamstown from 1960 to 2004.

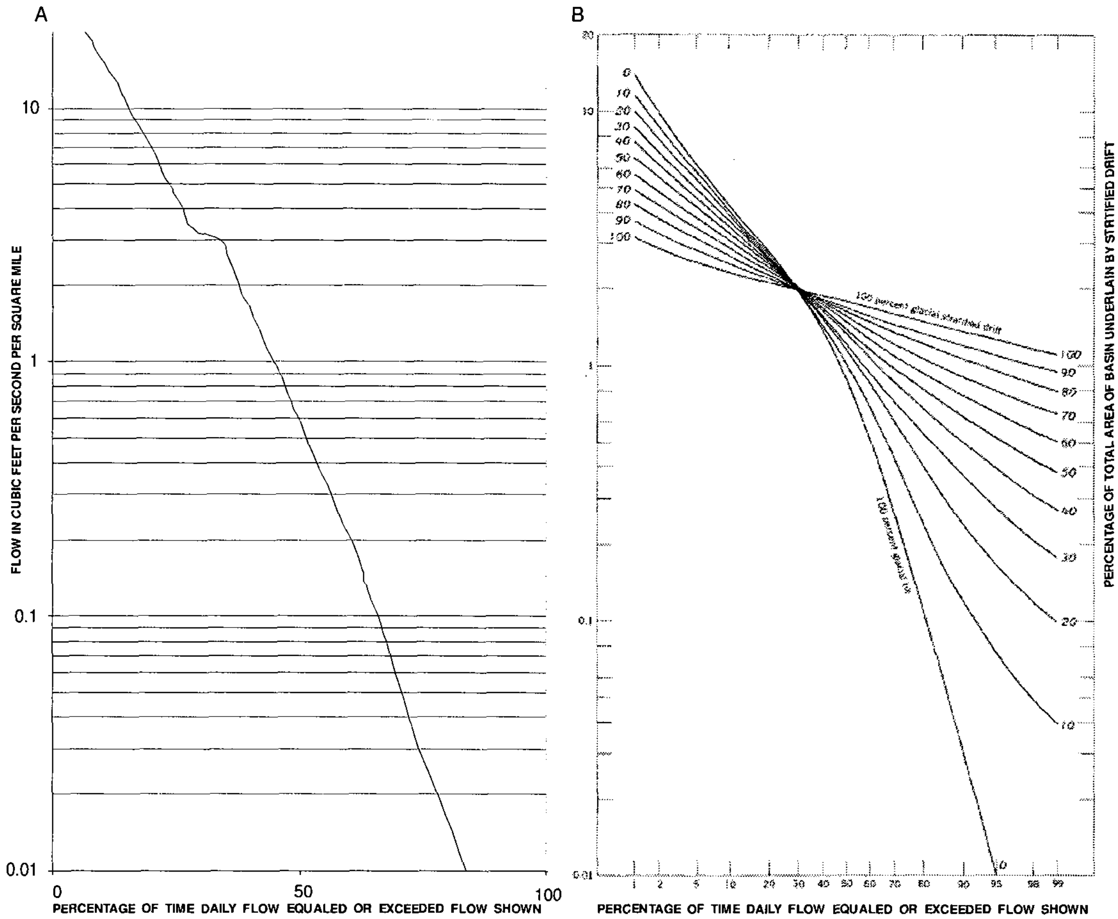


Figure 11. Comparison between shapes of flow duration curves in A) the Raisin River watershed at the E.C. Williamstown gauge from 1960 to 2004 and B) varying types of glaciated terrain (Thomas 1966). All flow data were normalized to a total historical average discharge of 1.8 cfs per sq. mile in order to compare shapes of each curve independently of climate.

Hydrogeology

Lithology, structure and stratigraphy provide a reasonable estimate of the behaviour of aquifers within a watershed (Freeze & Cherry 1979). The groundwater within the regional aquifer underlying the Raisin River watershed generally flows in a northwest to southeast direction, while local aquifers generally flow in a similar direction but vary according to the local topography (Figure 6 and 12). The low hydraulic conductivities of the glacial and glaciomarine sediments of the unconfined aquifer found throughout most of the basin (Figure 6 and 13) combined with the low topographical variation of the study area produce low vertical hydraulic gradients in sediments especially in and around the streambed of the Raisin River (Figure 14). The potential for groundwater recharge to the aquifer due to the low hydraulic conductivity of the sediments is generally low and is in the range of about -100 to 10 mm/yr along the banks and streambed of the Raisin River (Figure 15). In the context of this figure, a negative hydraulic gradient indicates upward flow of groundwater to the surface and a positive hydraulic gradient indicates downward flow of surface water to the phreatic aquifer whereas later in this thesis, hydraulic gradients will be defined in the reverse direction for practicality. Because of the low hydraulic conductivity of the unconfined aquifer, most of the groundwater wells (87.7%) in the region draw water from the regional bedrock aquifer (Singer et al. 2003).

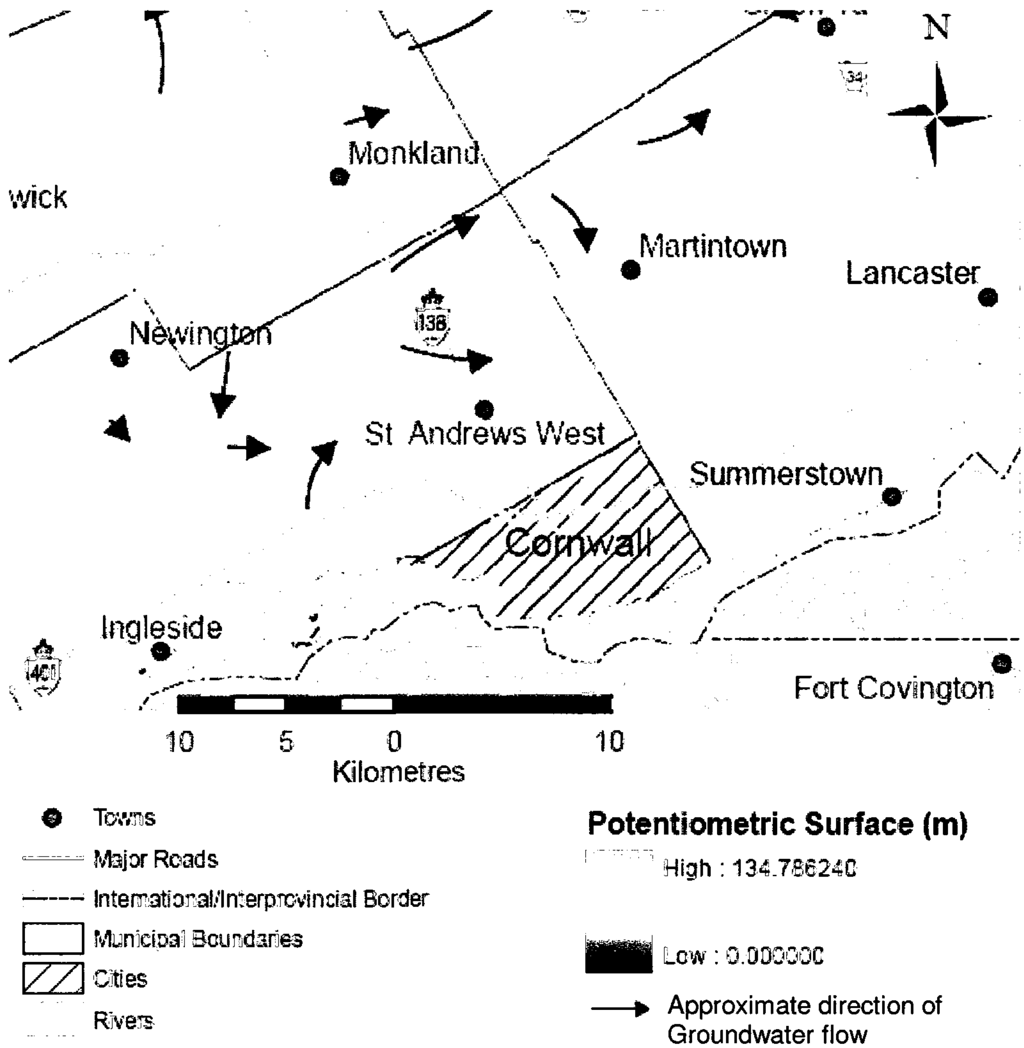


Figure 12. Water table and groundwater flow direction in the Raisin River watershed. From Figure 21: Overburden Potentiometric Surface and Direction of Groundwater Flow (Raisin-South Nation SWPR 2007).

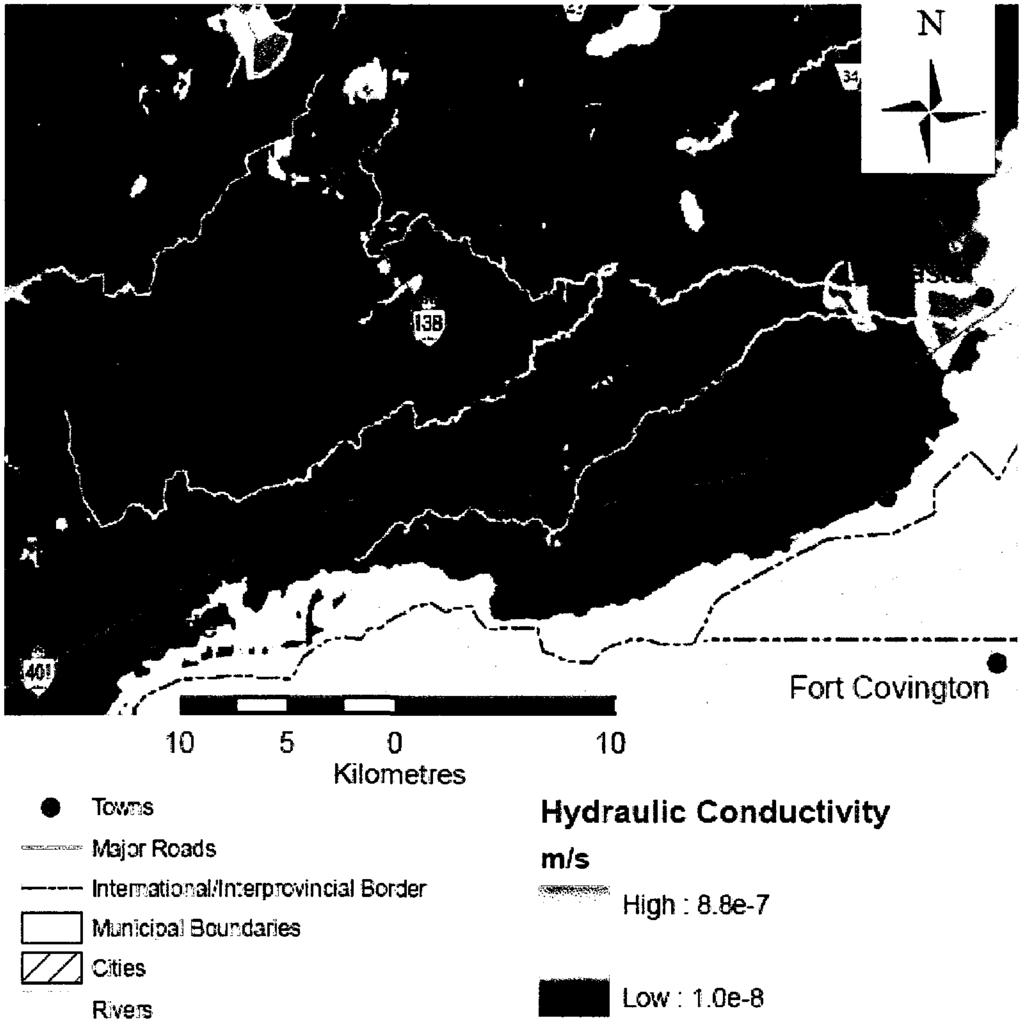


Figure 13. Hydraulic conductivity throughout the Raisin River watershed. From Figure 26: Hydraulic Conductivity of the Confining Unit (Raisin- South Nation SWPR 2007).

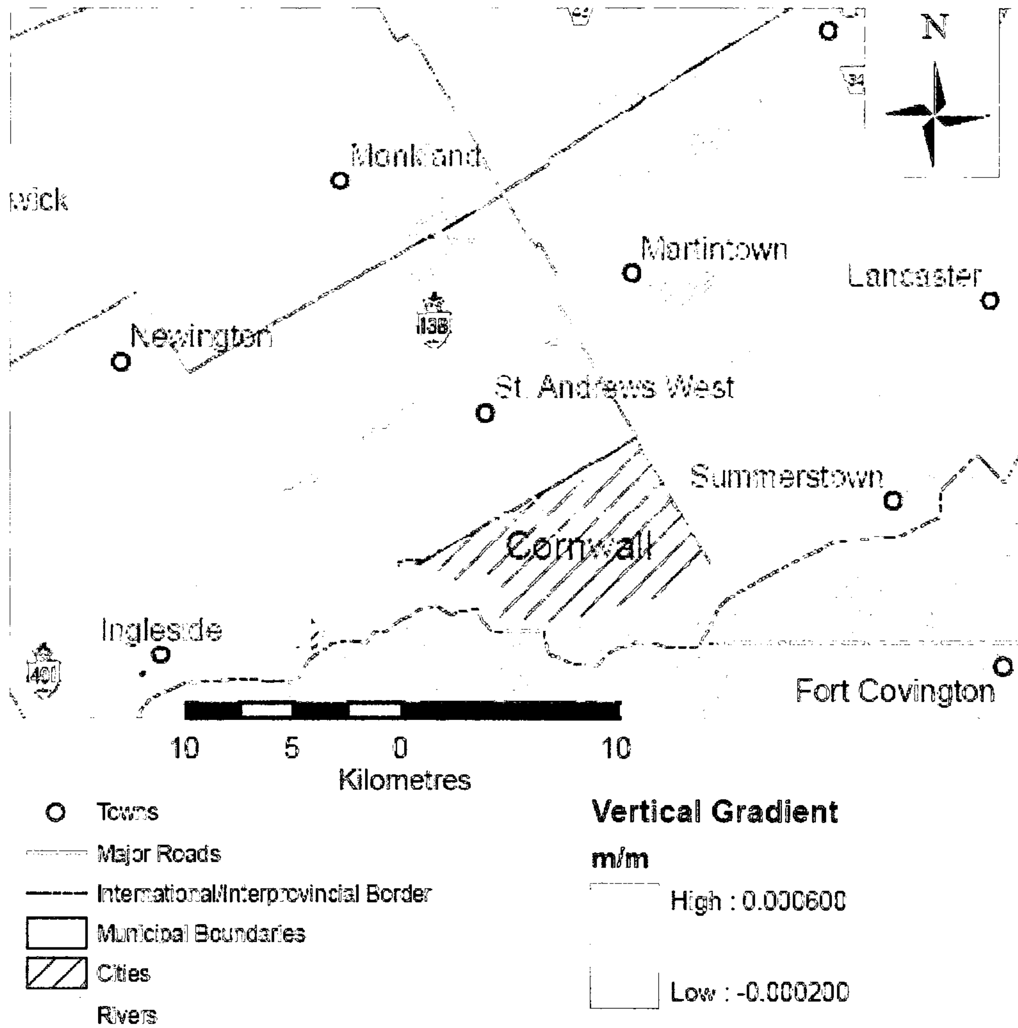


Figure 14. Vertical hydraulic gradients in the Raisin River watershed. From Figure 25: Vertical Gradient Recharge to the Contact Zone Aquifer (Raisin- South Nation SWPR 2007).

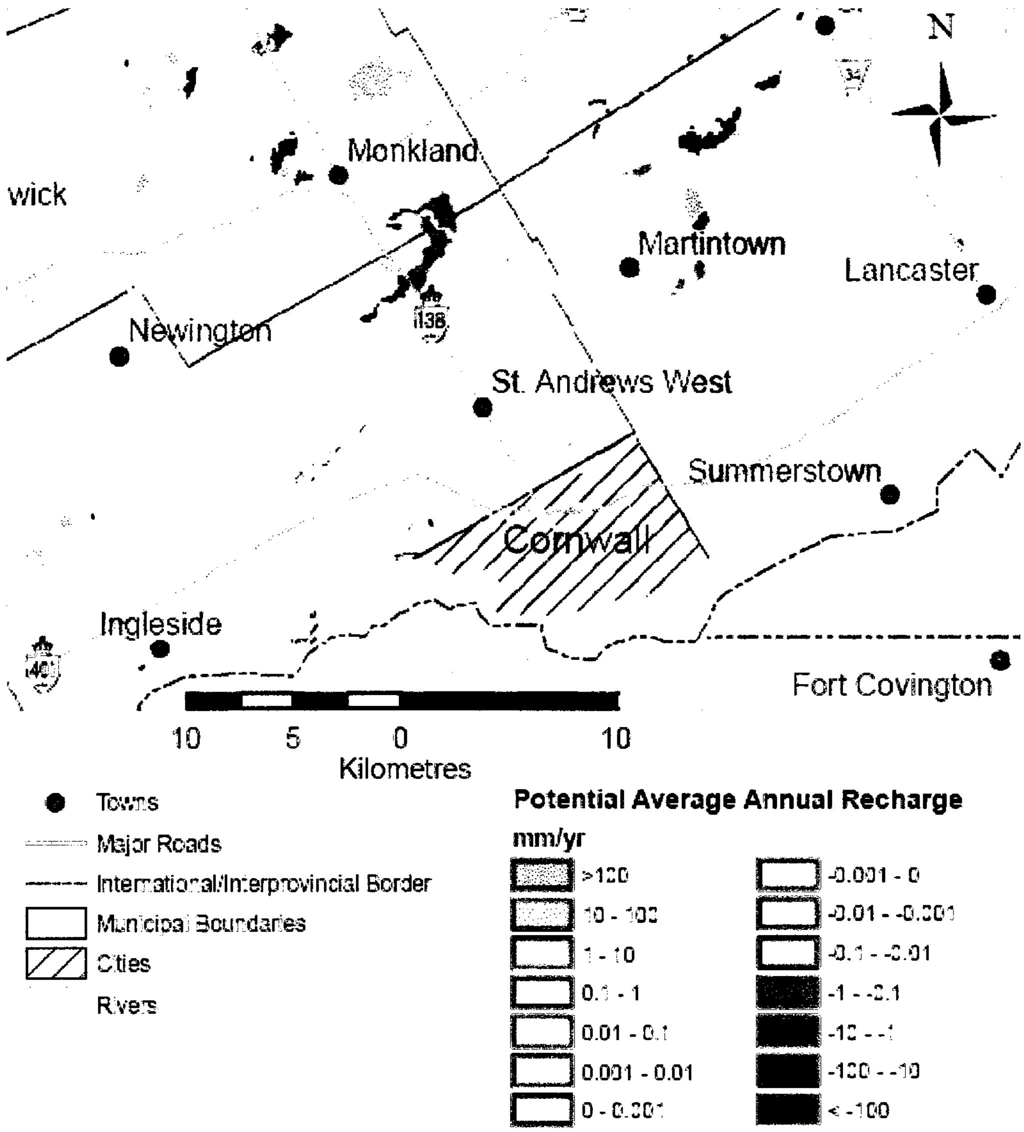


Figure 15. Potential groundwater recharge in the Raisin River watershed. From Figure 27: Potential Average Annual Recharge to the Contact Zone Aquifer (Raisin- South Nation SWPR 2007).

Though the hydrology indicates that periodic high discharge events should force groundwater/ surface water interactions, it appears that the low hydraulic conductivity of the geology is working to prevent such interactions.

Chapter 3: Algal-nutrient relationships

Introduction

Algae in rivers are an important part of the food web as primary producers. Algae can be found suspended in the water column (phytoplankton), or attached to rocks (periphyton). Compared to lakes, the factors determining suspended algae and periphyton biomass are less well understood for river and stream ecosystems. In general, light, nutrients and current are the main three factors limiting algal biomass. Light may become limiting, especially in heavily forested streams (Quinn et al. 1997), though when nutrients are limiting, several studies show that shading has no effect on algal biomass (Stockner & Shortreed 1978, Mosisch et al. 1999).

Nutrient limitation in streams primarily arises from either nitrogen and/ or phosphorus, (Pringle & Bowers 1984, Carpenter et al. 1998, Flecker et al. 2002). Since there is a fairly constant ratio of C:N:P in algal cells, it is widely held that when ratios in water fall below intracellular concentrations, nutrients can become limiting. In a meta-analysis of 158 studies of temperate lotic systems, 13% of streams were N-limited, 18% were P-limited, 44% were co-limited and 26% showed no nutrient limitation (Dodds & Welch 2000). Another meta-analysis of 237 studies found similar results where 17% were N-limited, 19% P-limited, 23% were co-limited and 43% were not affected by nutrients (Francoeur 2001). No clear pattern has emerged geographically or temporally in terms of nutrient limitation. The present study adds to the body of literature that will ultimately define these patterns.

Current velocity has been described as having a subsidy-stress relationship with algae as the current delivers a constant supply of nutrients but can cause sloughing at high velocities due to shear stress (Biggs et al. 1998). From the river continuum concept, the mid-order Raisin River was expected to be dominated by periphyton since in headwater streams where currents are stronger, attached algae are more resilient while suspended algae tend to dominate in the slower current of lowland rivers (Vannote et al. 1980).

For suspended algae, researchers are divided as to whether river hydrology is more important in determining biomass (Jones & Barrington 1985, Reynolds, 1988, Sullivan et al. 2001) than nutrient concentrations (Basu & Pick, 1996, Van Nieuwenhuyse & Jones 1996, Bukaveckas & Crain 2002, Dodds 2006). For periphyton, there is a similar debate as to whether nutrient loading (e.g. Chételat et al. 1999) or flow (Figueroa-Nieves et al. 2006) is the most important factor influencing algal biomass. Using a multiple river watershed model that included the Raisin River, Chételat et al. (1999) found a significant correlation between epilithic periphyton and water column total phosphorus (TP) concentrations in Eastern Ontario and Western Quebec rivers.

In the Raisin River, as in many mid-order rivers with a wide network of low-order headwaters, periphyton could be the most important primary producer before macrophytes or suspended algae. This study was designed to estimate the variability of both benthic and suspended algal biomass within the Raisin River watershed and determine the significance of relationships between algal biomass, nutrients and flow. Very few studies have sampled and analyzed these relationships for both communities of primary producers simultaneously.

Methods

Field methods:

During the summer of 2005, 41 riffle sites covering the entire watershed were sampled on the Raisin River over a 5 week period (Figure 16). Since sampling took place over a period of five weeks, it was a concern that the amount of daylight hours may have affected our results. Daylight hours were calculated for each sampling day from the CBM model described in Forsythe et al. (1995) using latitudes for each site taken from the Garmin© eTrex® H GPS. There were no significant correlations between daylight hours and biomass for either algal community (data not shown).

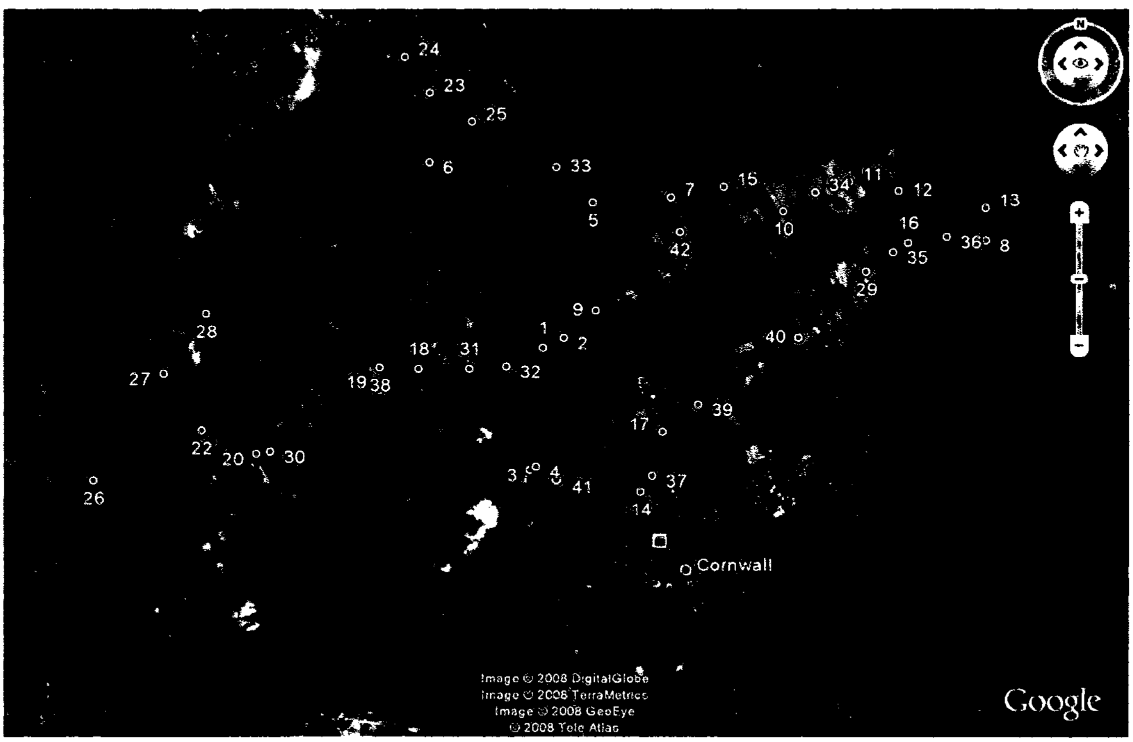


Figure 16. Locations of sites sampled in the Raisin River watershed during base flow, 2005.

River discharge was measured at each reach using a current velocity meter (Swoffer model 2100) and then multiplying by cross-sectional area of the reach. Hourly discharge data was available online from the Water Survey of Canada for the "Raisin River near Williamstown" gauge (45°9'19" N, 74°38'16" W).

In-situ measurements of temperature, specific conductivity (SpC), (where SpC is conductivity that has been compensated to 25 °C.), oxidation-reduction potential (ORP) and pH (Hydrolab Minisonde 4 Multiprobe Logger) were taken at each sampling. Water samples were taken from pool sections (0.1 –1.5 m in depth) before and after the riffle sites and kept cool, out of the light and processed within 24 hours for water chemistry analysis of nitrate (NO_3 + NO_2), ammonium (NH_4), total phosphorus (TP) and reactive phosphorus (SRP). Dissolved organic carbon and colour (440 nm) measurements were also made.

Suspended algal biomass was sampled in adjacent upstream and downstream pool sections. Water samples were filtered through GF/C Whatman filter paper to collect suspended chlorophyll a (Chl a).

Light attenuation through the water column at each site was measured with light extinction coefficient (K_d) calculations obtained by measurements of photosynthetically active radiation, PAR (400-700nm) (μmol/s/m^2) using a LI-COR Underwater Spherical Quantum Sensor.

Epilithic periphyton was sampled at 4 locations at each site to cover the riffle area (~1-5 rocks/ location at 0.05-0.44 m depth); water velocities and water column depth were recorded over each rock using a Swoffer velocity meter. Periphyton were not collected in shaded areas nor after heavy precipitation events. Although shaded areas were avoided, depending on the time of day that each site was visited, an area could be mistakenly interpreted as not shaded, even when it was shaded at other times of the day. Rock samples were kept cool, out of the light and processed within 24 hours.

Lab analysis:

All nutrients were analysed with a Lachat Instruments QuickChem FIA+ 8000 series. The QuickChem® Method In line 10-115-01-3-A with a method detection limit (MDL) of 0.007 mg P/L and a range of 0.1-10.00 mgP/L was used for total phosphorus (TP) and soluble reactive phosphorus (SRP). Methods 10-107-06-1-J and 10-107-04-1-C with an MDL of 0.002 mg N/L and a range of 0.01-2.00 mg N/L were used for NH_4 and NO_3 + NO_2 respectively. Samples that were higher in concentration than detection limits were diluted and re-analyzed.

Periphyton and suspended algal chlorophyll a (Chl a) were extracted using DMSO (Burnison 1980) and concentrations were estimated from absorbance measurements at wavelengths given by the equations of Jeffrey and Humphrey (1975). The rock surface area was used to normalize chlorophyll a concentrations to surface area. Rock surface area was estimated using Al foil to cover the rock (Steinman & Lamberti 1996).

Statistics:

The objective of this study was to determine the environmental variables that best explain the variability at the watershed scale using multiple regression models. All data were analysed using Systat II software. Most variables measured were not normally distributed and so had to be transformed in order to be used in linear regressions. Environmental variables were tested for normalcy using the Shapiro-Wilk test where distributions with $p > 0.01$ were considered to be normally distributed. Transformations used were for right-skewed data including log, square-root, quad-root, inverse functions; the McCalls area transformation was also used.

Correlations between suspended algae or periphyton and environmental variables were examined for significance and then ranked by p-values. Stepwise multiple regressions were used to define the most appropriate predictive models (Kutner et al. 2005). The alpha limit for a parameter to enter the stepwise multiple regression model was set at $p < 0.05$; $p > 0.05$ was the limit set to remove a parameter from the model. Collinearity between any pair of independent variables was limited to $R < 0.60$, a limit which has been used for similar models of algal biomass in rivers of western Quebec and southern Ontario (Basu & Pick 1996).

Two multiple regression models that used the best explanatory variables were produced for suspended algae and periphyton. These models were evaluated based on highest adjusted R^2 values for the regression, while maintaining a significance of greater than 95% ($p < 0.05$) for each variable within the regression. When two parameters were highly correlated, the parameter with the most biological significance was chosen for the multiple regression.

Results

Water Chemistry across the Watershed:

Table 2 and Table 3 show that physical and water chemistry variables encompass a 10 to 100 fold variation across the watershed with the exception of temperature and conductivity (SpC) (1.5-fold and 2 fold variations respectively). Dissolved oxygen (DO) data are not presented as samples were taken at different times of day; the range of DO was 6.2 – 14.8 mg/L. Unfortunately, total nitrogen could not be examined in this study, so the influences of nitrogen could not be fully observed. The spatial distribution of total phosphorus (TP) concentration showed no obvious upstream/ downstream pattern, for example, all branches of the river show high variability (Figure 17). When multiple samples were taken at each site, TP, NO_3, periphyton and suspended algae all showed more variability between sites than within sites ($p<0.01$ for all sites, from single-factor ANOVA).

Table 2. Descriptive statistics for physical variables

	N of cases	Min.	Max.	Median	Mean	Std. Dev	
Strahler Stream Order	42	3	6	4	4.5	0.8	
Distance upstream from Mouth of Raisin (km)	42	6.92	66.27	33.56	33.66	16.73	*
Site Discharge (m^3/s)	40	0	0.709	0.061	0.121	0.185	
Daily Discharge at Williamstown Gauge (m^3/s)	42	0.065	1.028	0.287	0.333	0.29	
Average velocity in riffle taken above periphyton samples (m/s)	42	0	0.61	0.27	0.26	0.14	*
Depth of pool (m)	42	0.10	0.50	0.30	0.30	0.10	*
Temperature (°C)	41	18.0	29.7	23.9	24.0	2.5	*
Extinction Coefficient	41	0.67	7.94	3.45	3.21	1.69	*
Turbidity (NTU)	42	0.94	14.47	6.28	6.39	3.52	*
Specific Conductivity (µS/cm)	41	265.1	766.7	479.1	495.5	108.9	*

* - Variable is normally distributed according to Shapiro-Wilk test where $p>0.01$.

Table 3. Descriptive statistics for chemical and biological variables.

	N of cases	Min.	Max.	Median	Mean	Std. Dev.	
Nitrate (μg/L)	37	5	300	60	71	72	
Ammonia + Ammonium (μg/L)	28	5	70	20	26	14	
Total Phosphorous (μg/L)	42	10	180	50	58	27	
Soluble Reactive Phosphorous (μg/L)	37	5	60	30	29	14	*
Dissolved Organic Carbon (mg/L)	42	1.5	40.6	19.0	18.3	10.6	*
Colour (440 nm)	42	0.014	0.710	0.141	0.180	0.162	
Periphyton biomass (mg/m^2)**	42	17.3	170.1	48.1	56.6	31.3	
Suspended Algal biomass (mg/m^2)**	42	0.003	0.068	0.013	0.017	0.012	
Suspended Algal biomass (μg/L)	42	1.9	22.8	5.2	5.9	3.7	

* - Variable is normally distributed according to Shapiro-Wilk test where $p>0.01$.

**- Periphyton biomass is expressed in mg/m^2 of rock surface whereas suspended algal biomass is expressed in mg/m^2 of sediment interface.

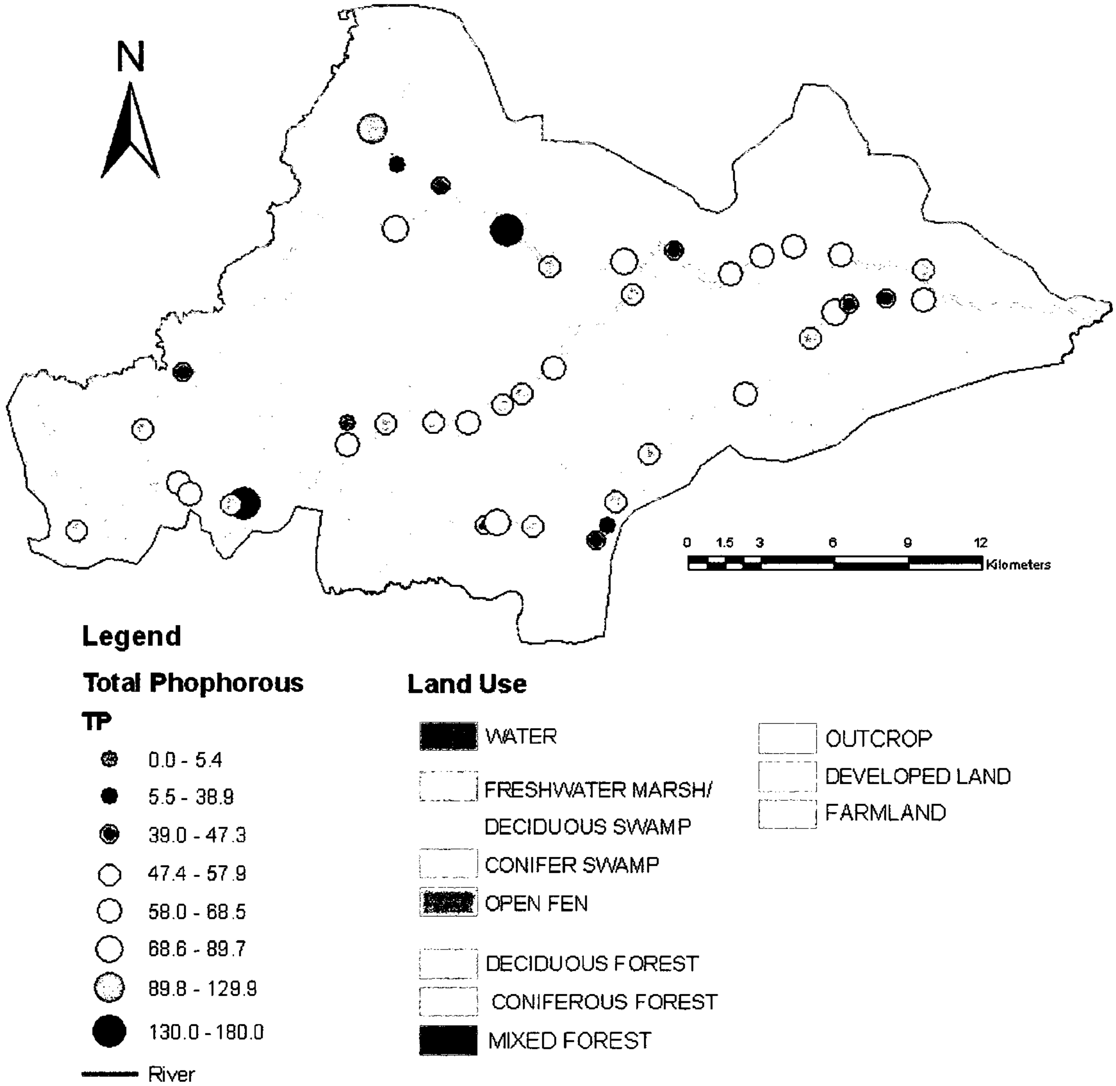


Figure 17. Total phosphorus variability (µg/L) at 42 sites distributed throughout the Raisin River watershed

Spatial trends within the watershed could be distinguished by comparing average concentrations of environmental variables between branches of the Raisin River (Table 4). Turbidity was highest in the Middle Raisin and this contributed to a higher extinction coefficient. Conductivity was highest in the South, possibly due to a greater contribution from road salt, due to urban land-use. However, this branch was also located in a separate geologic formation, which could have affected the conductivity of the surface water as described previously in Chapter 2.

Table 4. Summary of all variables by branch of the Raisin River.

	North		Middle		South		N+M*	
Variable	Avg	N	Avg	N	Avg	N	Avg	N
Distance to mouth of Raisin [km]	36.19	5	49.05	15	21.01	13	16.31	6
Site Discharge [m^3/s]	0.04	5	0.09	13	0.06	13	0.45	6
Daily Discharge at the Williamstown gauge [m^3/s]	0.38	5	0.29	15	0.29	13	0.61	6
Averag Velocity measured in the riffle [m/s]	0.14	5	0.25	15	0.28	13	0.36	6
Depth of pool adjacent to riffle [m]	0.23	5	0.28	15	0.35	13	0.32	6
River Temperature [°C]	22.42	5	23.70	15	24.33	12	27.52	6
DOC [mg/L]	20.96	5	27.99	15	5.98	13	18.04	6
Colour at 440 nm [a.u.]	0.16	5	0.33	15	0.04	13	0.14	6
Extinction Coefficient [$\mu mol/s/m^2$]	3.41	5	4.60	14	1.65	13	3.11	6
Turbidity [NTU]	4.27	5	7.63	15	5.80	13	6.70	6
Specific Conductivity [μS/cm]	471.54	5	422.47	15	573.95	12	460.25	6
Nitrate (NO_3) [μg/L]	62.00	5	101.67	15	50.00	10	49.00	5
Ammonium (NH_4) [μg/L]	26.67	3	30.00	10	18.33	9	28.00	5
Total Phosphorus [μg/L]	54.00	5	59.33	15	50.00	13	55.00	6
Soluble Reactive Phosphorus [μg/L]	34.00	5	30.67	15	17.00	10	38.00	5
Periphyton biomass as Chl-a [mg/m^2]	57.65	5	45.45	15	65.10	13	65.66	6
Suspended algal biomass as Chl-a [μg/L]	5.90	5	5.50	15	3.83	13	8.97	6

* N+M is downstream of the confluence of the North and Middle Raisin but before the confluence of the South Raisin

Nitrate concentrations were highest in the Middle Raisin. Ammonia was lowest in the South Raisin, where DOC, colour and the extinction coefficient were on average also lowest among all branches. This is not surprising since all other branches had their headwaters in wetlands where higher ammonia concentrations could be attributed to the decomposition of organic matter and humic acids that contribute to colour and increased opacity (as suggested by the higher extinction coefficient for other branches of the Raisin River).

Soluble reactive phosphorus was highest in the Raisin after the confluence of the North and Middle (herein referred to as N+M Raisin) but no such trend existed for total phosphorus. The fact that SRP was higher in this branch but not TP suggests that point sources along this branch may have been delivering higher concentrations of available P to the river. Also, SRP was lower in the South Raisin than in the North or Middle Raisin Rivers, possibly due to less agricultural activity.

Originally when all sites along the main branches were averaged, the North Raisin had the highest average total phosphorus concentrations in the watershed (77 μg/L). This ranking was primarily due to concentrations found at sites 33 and 24 on the North Raisin which had the highest and third highest total phosphorus concentrations surveyed in this study (180 μg/L and 90 μg/L, respectively). These two sites were removed from subsequent models for the reasons outlined below.

Site 33 was located just after the confluence of a fourth order tributary to the North Raisin that drained a combination of forested and agricultural land. Measurements taken on August 10^{th} indicated that discharge was 80 L/s, however on the same day the Environment Canada gauge located downstream on the North + Middle Raisin near Williamstown reported a discharge of only 66 L/s. This combination of an extremely high total phosphorus concentration and a discharge measurement that was inconsistent with other discharge measurements reported for that day were the reasons that site 33 was not included in subsequent models.

Site 24 was located just east of Monkland at the headwaters of the North Raisin, where two streams combine to form the North Raisin River. Samples were taken after the North Raisin passed through a culvert underneath the train tracks where flow was estimated at less than 1 L/s and specific conductivity was very high (702 μS/cm) compared to other samples on the North Raisin River (ranging from 391 μS/cm to 640 μS/cm). As suggested by the anomalies in SpC, TP and discharge, the site may have been affected by the train tracks, and was therefore discarded from subsequent analysis.

Overall, while some variables appeared to differ greatly between the South Raisin and all other branches (colour, DOC, extinction coefficient), when all variables were taken into account, MANOVA results indicated that the South Raisin and other branches were not significantly different ($p > 0.05$) and so all branches of the Raisin River were analysed together in regression analyses for the watershed.

Distribution of data:

Since 7 out of 37 values for nitrate were below the analytical detection limit (10 μg/L), data could only be verified as log-normally distributed above the detection limit. Values below the detection limit were set at half the detection limit and log-transformed along with the rest of the data.

Total phosphorus could not be transformed such that it was normally distributed using any conventional parametric function and so it was left untransformed. This non-normality was likely due to the narrow range of TP concentrations sampled in the Raisin River, where over 60 % of concentrations were either 50 μg/L or 60 μg/L.

The McCalls area transformation (Chase 1976) was used to normalize the distribution of two variables: daily average river discharge at the Williamstown Environment Canada gauge and site discharge. Flow (discharge) data from the Williamstown gauge could not be otherwise transformed because of a few relatively higher values that corresponded with samples taken in early July. A few downstream sites located after the confluence of the North and Middle branches of the Raisin River were relatively much greater in flow than the remaining sites, which also prevented the site discharge data from being transformed using conventional parametric functions.

Strahler stream order was the only discrete, ordinal variable and was distributed according to a poisson distribution ($p > 0.01$) but could not be estimated by a normal distribution ($p < 0.01$). This variable was left untransformed. A list of transformations made for each variable, listed by their abbreviations as they are referred to in Tables 5 to 8 can be found in Appendix D, Table D1.

Comparing Environmental Variables with Algal Biomass:

The Raisin River watershed showed great variability in algal biomass, both epilithic and suspended (Table 3). When suspended algal biomass was expressed in mg/m^2 of sediment interface, and periphyton was expressed in mg/m^2 of rock surface an approximate comparison could be made between periphyton and suspended algal biomass that showed periphyton to have the greater biomass over the reaches studied (Table 3). Although there is certainly more periphyton per m^2 of sediment interface than per m^2 of rock surface, there is still evidently more periphyton biomass than suspended algal biomass.

Correlations between all environmental and biological variables are presented in a correlation matrix (Table 5). Variables are grouped with physical variables listed first, followed by chemical and finally biological variables. Generally, correlations greater than 0.30 were significant.

Table 5. Pearson Correlation Matrix of multiple R between pairs of independent variables used to limit collinearity. Pearson correlation coefficient, R ≥ 0.55 was used as the limit for pairs of independent variables entering the step-wise regression models for either periphyton or suspended algal biomass.

	STRAHLER	DISTOMTHKM	MATSITEDSCH	MATWILLDAYDI	PERIVEL	POOLDEPTH	TEMP	DOC	SQRTCOLOUR	EXTCOEFF	TURBIDITY	SPC	LOGNO3	LOGNH4	TP	SRP	LOGPERI	LOGSUSPUGL	LOGSUSPMGM2
STRAHLER	1																		
DISTOMTHKM	-0.27	1																	
MATSITEDSCH	0.52	-0.39	1																
MATWILLDAYDI	0.40	-0.12	0.66	1															
PERIVEL	0.29	-0.11	0.53	0.31	1														
POOLDEPTH	0.14	-0.56	0.37	0.12	0.07	1													
TEMP	0.52	-0.49	0.38	0.24	0.33	0.22	1												
DOC	0.35	0.68	0.04	0.27	0.04	-0.35	-0.06	1											
SQRTCOLOUR	0.27	0.71	-0.04	0.14	0.03	-0.35	-0.05	0.96	1										
EXTCOEFF	0.35	0.61	0.02	0.21	0.06	-0.39	-0.06	0.83	0.82	1									
TURBIDITY	0.40	-0.01	0.41	0.28	0.30	0.06	0.20	0.21	0.16	0.31	1								
SPC	-0.30	-0.47	0.03	-0.23	-0.10	0.30	-0.26	-0.74	-0.76	-0.64	-0.29	1							
LOGNO3	0.13	0.38	0.34	0.34	0.35	-0.27	0.08	0.47	0.49	0.39	0.30	-0.40	1						
LOGNH4	0.03	0.43	-0.23	0.07	-0.21	-0.36	-0.04	0.54	0.50	0.50	0.02	-0.20	0.26	1					
TP	0.30	0.08	0.05	0.12	-0.03	-0.03	0.32	0.28	0.27	0.39	0.39	-0.40	0.31	0.30	1				
SRP	0.50	0.16	0.25	0.38	0.27	-0.15	0.21	0.54	0.46	0.29	0.12	-0.41	0.60	0.22	0.11	1			
LOGPERI	-0.16	-0.13	0.05	-0.01	-0.06	0.00	0.12	-0.23	-0.22	-0.22	-0.44	0.21	-0.23	-0.13	-0.26	-0.09	1		
LOGSUSPUGL	0.51	-0.05	0.10	0.26	0.05	-0.10	0.35	0.29	0.24	0.40	0.26	-0.40	0.05	0.05	0.26	0.29	-0.38	1	
LOGSUSPMGM2	0.49	-0.37	0.21	0.25	0.09	0.46	0.48	0.02	-0.01	0.04	0.21	-0.18	-0.15	-0.17	0.14	0.14	-0.33	0.79	1

Predicting Suspended Algal Biomass:

Variables that significantly predicted suspended algal biomass are listed in Table 6. Nutrient variables such as TP, SRP, NO_3 and NH_4 were not significantly correlated with suspended algal biomass ($p > 0.05$). Suspended algal biomass was less than predicted by TP in a published model of eastern Ontario watersheds (Basu & Pick 1996) (Figure 18). Strahler stream order was the strongest predictor (Adjusted $R^2 = 0.240$, $p = 0.001$) where higher order streams correlated with greater suspended algal biomass (Figure 19). The next strongest predictor was specific conductivity that was negatively correlated with suspended algal biomass. The extinction coefficient and river temperature were also positively correlated with suspended algal biomass. Also, suspended algae and periphyton were negatively correlated (Figure 20).

Table 6. Simple regressions of suspended algae and periphyton as a function of abiotic factors and as a function of each other.

Variable							
Dependant	Independent	n	Constant	Coefficient	R^2	Std. error	p
LOGSUSP	STRAHLER	40	0.135	0.122	0.260	0.176	0.001
	SPC	39	1.068	-0.001	0.161	0.189	0.011
	EXTCOEFF	39	0.551	0.047	0.157	0.186	0.013
	LOGPERI	40	1.274	-0.342	0.147	0.188	0.015
	TEMP	39	-0.019	0.029	0.125	0.193	0.027
LOGPERI	TURBIDITY	40	1.885	-0.029	0.196	0.205	0.004
	LOGSUSPUG/L	40	1.998	-0.429	0.147	0.211	0.015

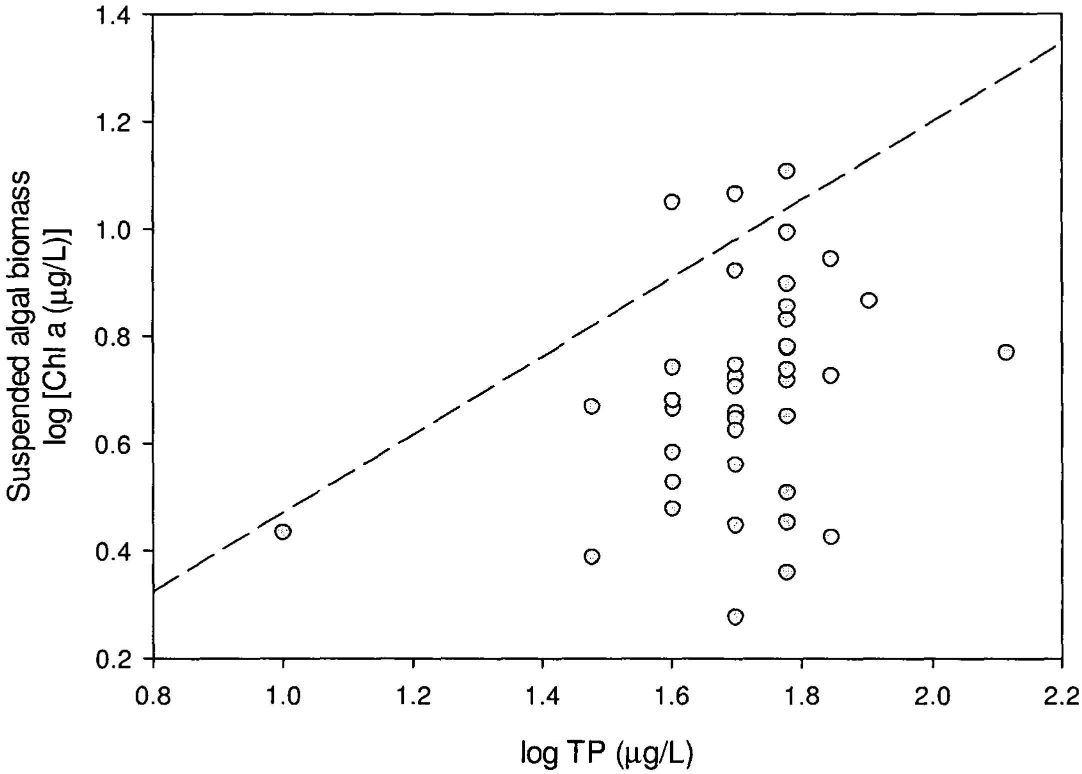


Figure 18. Raisin River suspended Chl a vs. TP compared to previous regression model for eastern Ontario watersheds from Basu and Pick (1996) (dotted line, log Chl a = -0.26 + 0.73log TP).

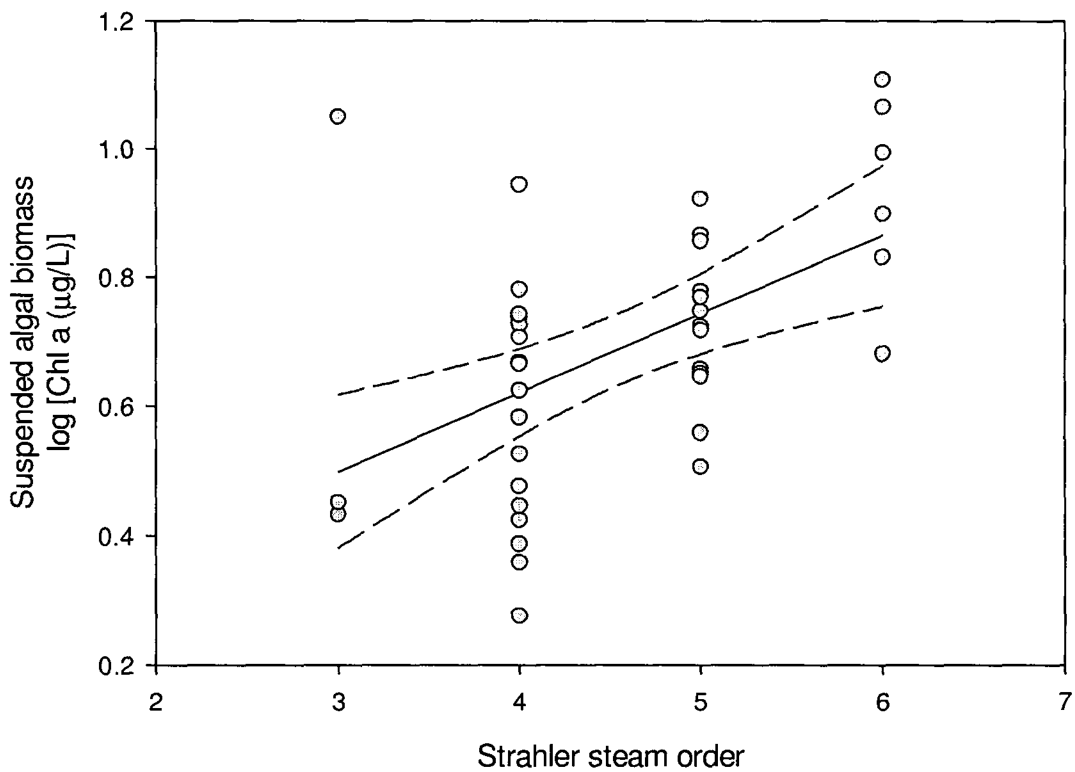


Figure 19. Positive correlation between Strahler stream order and suspended algal biomass (p = 0.001, R2 = 0.260). Dashed lines show 95% confidence intervals on the regression line.

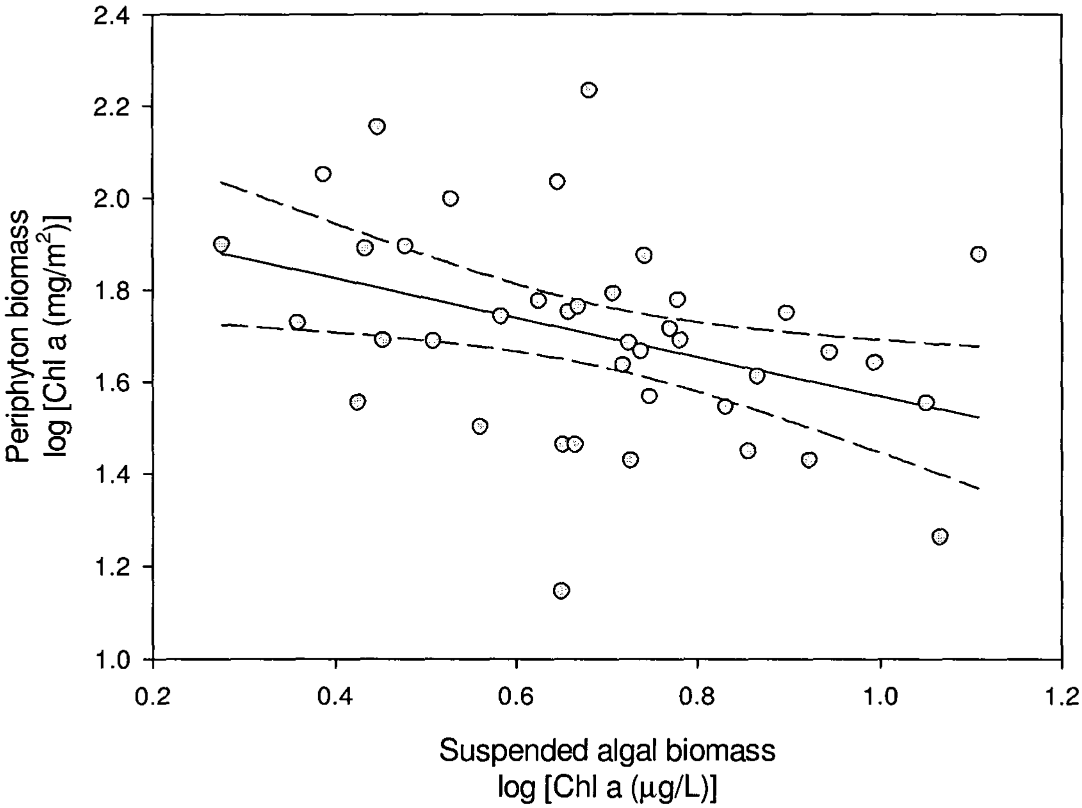


Figure 20. Negative correlation between periphyton Chl a concentration and suspended algal Chl a concentration (p = 0.015, R2 = 0.147). Dashed lines show 95% confidence intervals on the regression line.

Strahler stream order was correlated ($R \geq 0.30$) with the light extinction coefficient, temperature and SpC and so no additional regressors improved the model, while it remained in the analysis (Table 5). The model that best explained suspended algal biomass using only abiotic variables included the extinction coefficient and temperature (Adjusted $R^2 = 0.266$, $p = 0.002$), (Table 7). When periphyton was included as a predictor variable in the model, three other models significantly predicted suspended algal biomass including one model with regressors temperature and periphyton biomass (Adj. $R^2 = 0.273$, $p = 0.001$) and one with Strahler stream order and periphyton biomass (Adj. $R^2 = 0.318$, $p < 0.001$), (Table 7). The strongest model used regressors temperature, the extinction coefficient and periphyton biomass (Adj. $R^2 = 0.386$, $p < 0.001$).

Table 7. Simple and Multiple Regressions predicting suspended algal biomass as Chl a (μg/L)

Model	Coefficients					Regression			
	Effect	B	Std Error	Std. B	P(2 Tail)	N	R^2	Adj. R^2	Sig.
1						40	0.26	0.24	0.001
	CONSTANT	0.135	0.155		0.389				
	STRAHLER	0.122	0.033	0.510	0.001				
2						38	0.306	0.266	0.002
	CONSTANT	-0.214	0.287		0.461				
	EXTCOEFF	0.05	0.017	0.422	0.005				
	TEMP	0.031	0.011	0.382	0.01				
3						39	0.311	0.273	0.001
	CONSTANT	0.536	0.331		0.114				
	TEMP	0.034	0.012	0.407	0.006				
	LOGPERI	-0.389	0.125	-0.434	0.004				
4						40	0.353	0.318	<0.001
	CONSTANT	0.66	0.27		0.02				
	STRAHLER	0.11	0.032	0.460	0.001				
	LOGPERI	-0.277	0.12	-0.310	0.026				
5						38	0.436	0.386	<0.001
	CONSTANT	0.289	0.318		0.37				
	TEMP	0.034	0.011	0.423	0.003				
	LOGPERI	-0.326	0.117	-0.371	0.008				
	EXTCOEFF	0.041	0.016	0.346	0.013				

Predicting Periphyton Algal Biomass

When only abiotic factors were taken into account, turbidity was the only variable that was significantly correlated ($p < 0.05$) with periphyton biomass (Table 6) and the correlation was negative (Figure 21). No clear linear relationship existed between log periphyton and logTP (Figure 22), whereas a previous model that included sites from the Raisin River as well as other watersheds in eastern Ontario and western Quebec did produce a positive linear relationship (Cheletat et al. 1999).

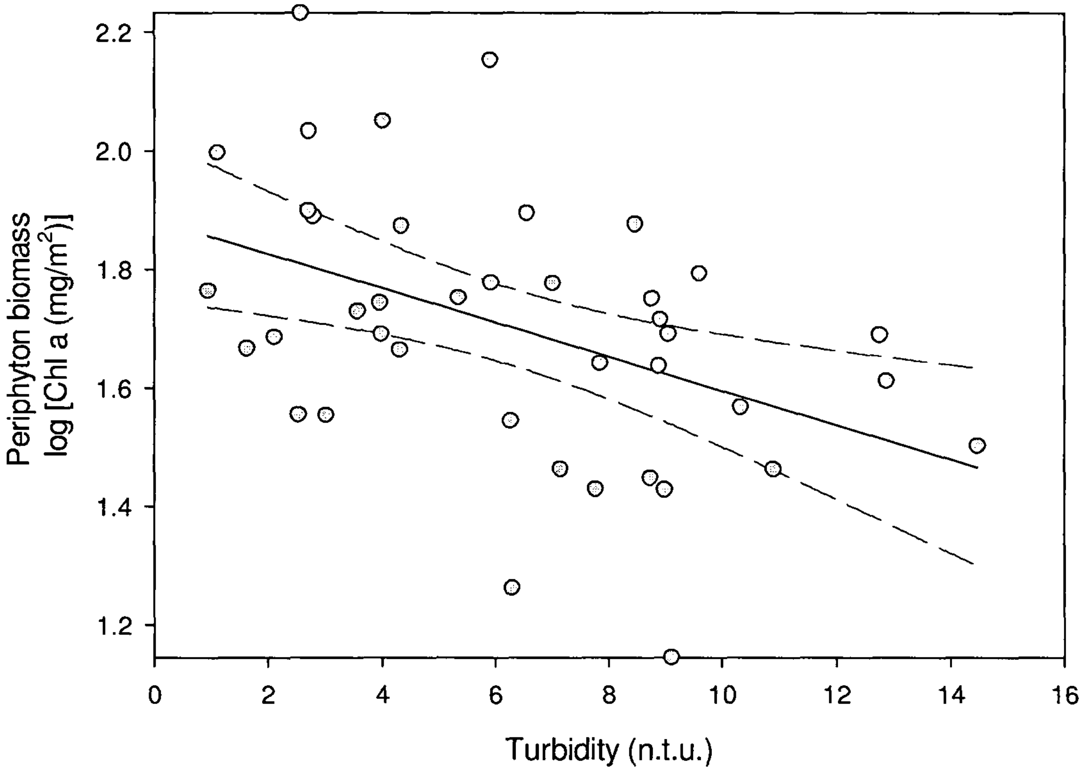


Figure 21. Periphyton biomass is negatively correlated with turbidity ($p = 0.004$, R2 = 0.196). Dashed lines show 95% confidence intervals on the regression line.

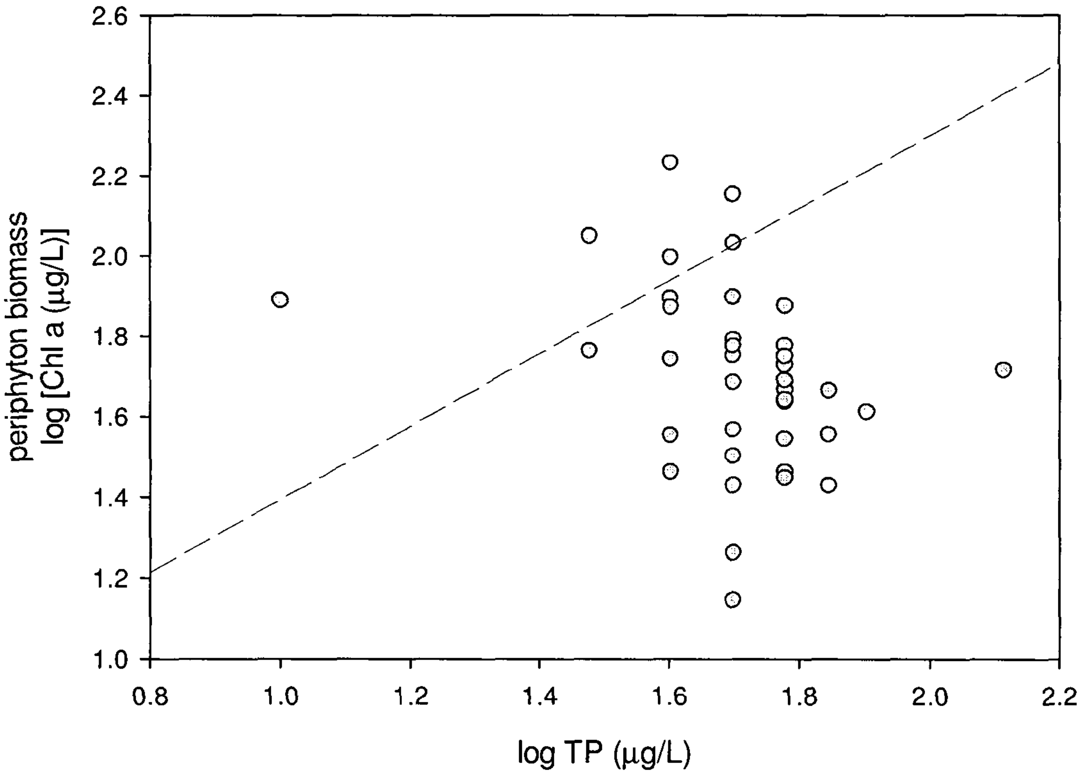


Figure 22. Scatter plot of periphyton Chl a and TP for the Raisin River compared to regression model of Chételat et al. (1999) (dashed line, log Chl a = 0.490 + 0.905log TP).

Suspended algal biomass was also significantly correlated with periphyton (Table 6), and the correlation was negative as stated above (Figure 20). When suspended algal biomass was included, a multiple linear regression model combining turbidity, suspended algal biomass and temperature, best explained variation in periphyton biomass (Adjusted R^2 = 0.339, $p = 0.001$) (Table 8). While temperature alone did not correlate with periphyton biomass ($p > 0.05$) (Appendix D, Table D5), it did explain the residuals of the multiple regression that included suspended algae and turbidity better than any other variable.

Table 8. Simple and Multiple Regressions predicting periphyton biomass as Chl a (mg/m2)

Model	Coefficients					Regression			
	Effect	B	Std Error	Std. B	P(2 Tail)	N	R^2	Adj. R^2	Sig.
1						40	0.196	0.174	0.004
	CONSTANT	1.885	0.069		0.000				
	TURBIDITY	-0.029	0.01	-0.442	0.004				
2						39	0.391	0.339	0.001
	CONSTANT	1.405	0.297		0.000				
	TURBIDITY	-0.028	0.009	-0.426	0.004				
	LOGSUSPUGL	-0.445	0.161	-0.399	0.009				
	TEMP	0.032	0.013	0.348	0.020				

Discussion

Suspended Algal Biomass:

Compared to other river systems across Eastern Ontario and Western Quebec, the Raisin River appears to have less suspended algae per unit TP than predicted by Basu and Pick's 1996 model (Figure 18), (where only 5 out of 40 data points from the Rasin River were above the published model line). Two possible explanations for the apparent paucity of suspended algae in the Raisin River are 1) the overall shallow depth of the system and 2) grazing losses.

The depth of the Raisin River was shallow through most of its length but natural riffle zones existed throughout that provided shallow sections where suspended algae could come into contact with the substrate and settle. Pools in the Raisin River located adjacent to sampled riffle zones had a maximum depth of half a meter whereas loss rates of suspended algae in rivers have been shown to increase when water depths decrease, especially below 1 m (Köhler et al. 2002).

Native clams were abundant at many sites (Pichard et al. 2008), and can be significant filter feeders. In the Rideau River, Basu and Pick (1997) found that exotic zebra mussels could filter about 40% of suspended algae per day assuming a 3m water column. Caraco et al. (2006) showed reduction in suspended algae with the intrusion of zebra mussels to the Hudson River, which can explain the lack of a significant suspended algae-TP relationship.

Suspended algal biomass was most strongly correlated with Strahler stream order, where higher order branches contained higher concentrations of suspended algae. This is not surprising according to the River Continuum Concept (Vannote et al. 1980), which predicts that suspended algae will become increasingly important and periphyton will become less important moving downstream through increasing stream orders.

Also correlated with suspended algal biomass but then as well correlated with Strahler stream order were specific conductivity, the extinction coefficient and temperature. Specific conductivity can sometimes be used as general estimate for nutrient concentrations; however in the Raisin River, SpC was negatively correlated with TP and SRP. Since there tends to be a higher proportion of groundwater discharge (with higher SpC) in headwaters than further downstream, the negative correlation between Strahler stream order and SpC may explain why SpC is negatively correlated with suspended algal biomass. The extinction coefficient is positively correlated with suspended algal biomass; although suspended Chl a contributes to the observed light attenuation, dissolved organic matter (colour) is the most important factor influencing light attenuation in the Raisin River (Holsworth 2005).

Conductivity and the extinction coefficient were negatively correlated and so cannot be used together in a multiple regression to explain suspended algal biomass. Instead, temperature (which is also correlated with Strahler stream order) and the extinction coefficient combine to form the best model to explain suspended algal biomass. Temperature, which typically increases algal growth rates (Allan & Castillo 2007), is positively correlated with suspended algal biomass. For this spatial study it would make sense that larger, more downstream sites have higher in-stream temperatures, likely due to less shading (Rutherford et al. 1997) and relatively less cooling from groundwater discharging into the stream that reduces stream temperatures in the summer (Mellina et al. 2003). Since temperature is positively correlated with Strahler stream order, the overall depth of the river is likely the main driver in predicting suspended algal biomass, however because this study examined only riffle zones and adjacent pools, the overall depth of the river was not well characterized.

Periphyton biomass:

Periphyton biomass was not significantly related to nutrient concentrations, unlike other studies in the region that found TP to be an important predictor of periphyton (Chételat et al. 1999) (Figure 22). However, it should be noted that since nutrient samples were taken concurrently with periphyton sampling, the nutrient concentrations of that day may not correlate well with periphyton biomass because the latter represents an integrated signal of conditions over the growth period. Some researchers have got around this problem by incorporating land-use as a predictor variable with more success (Carr et al. 2005), since different land-uses can be attributed to varying nutrient inputs to the river.

Less periphyton biomass was measured in the Raisin River than predicted by the model of Chételat et al. (1999). The dearth of periphyton in the Raisin River could be explained by 1) grazer losses or 2) limited metabolite removal and nutrient delivery provided by groundwater/ surface water interactions.

The deficit in periphyton biomass might be due to effective grazing by benthic invertebrate communities as reported by Hillebrand (2002). The texture of the streambed is important in creating refuges during extreme flows for various flow-sensitive invertebrates and other organisms (Whiting 2002). Many of the sites chosen as riffle sites on the Raisin River were armoured (that is the substrate at the surface is coarser than that found at greater depths). Because of the refuges created at these armoured sites, there may be more grazers, which could help explain the observed deficit in periphyton biomass.

Alternately, periphyton being intimately linked with the substrate could be responding to isolated zones of groundwater discharge as found in other alluvial river systems and floodplains (Stanford & Ward 1993, Ward & Tockner 2001). Zones of groundwater discharge (upward flow of groundwater to the river) could deliver additional nutrients to the river whereas zones of groundwater recharge (downward flow of river water to the hyporheic zone) may provide flow at the sediment-water boundary that could stimulate algal growth through metabolite removal from and nutrient delivery to the boundary layer (Allen and Castillo 2007). If the Raisin River has less groundwater/ surface water interactions than other rivers in the area, this may decrease the amount of benthic algal biomass overall, but may still explain some of the variability in periphyton biomass within the watershed.

Periphyton biomass was significantly, negatively correlated with turbidity. One mechanism that may explain this relationship is sloughing due to periodic elevated discharge events, since turbidity was positively correlated with site discharge. While extreme discharge events may be important for periphyton removal, no sampling took place during or directly after such events. Despite attempting to avoid sampling during or after spates, the Raisin River did experience some fluctuations in water levels throughout the sampling period.

Turbidity levels rise in streams after precipitation events, and sloughing of periphyton during these episodes of higher flow could explain the negative correlation between periphyton biomass and turbidity that was observed. Turbidity caused by fine clays might also interfere with nutrient uptake, thereby preventing algal growth. Although this watershed does contain areas with silt and clay, they are not the dominant substrate; other researchers have found that clay-rich turbidity does not affect SRP uptake by algae (Wolfe & Lind 2008).

Periphyton biomass was also negatively correlated with suspended algae. A negative relationship could be explained by light limitation; however periphyton collected in riffle zones would not experience significant light limitation. Stream depth is likely important in determining the dominant community. Suspended algae could be competing with periphyton for available nutrients. It is possible that sloughing of periphyton due to spates may contribute to suspended algal biomass and reduce periphyton biomass. In rivers, suspended algae have been shown to contain varying amounts of re-suspended periphyton (Allen & Castillo 2007). Since this work did not include analysis of the algal communities in suspended samples, it is possible that much of the suspended algae was actually re-suspended periphyton and not true river phytoplankton (potamoplankton).

Turbidity and suspended algae were the only significant regressors for periphyton biomass as Chl a. For the Raisin River watershed, a model using turbidity alone predicts only 17% of the variation in periphyton biomass (Table 8). The combination of suspended algal biomass as Chl a, turbidity and stream temperature make the strongest model for predicting periphyton biomass in the Raisin River, explaining 34% of variation in periphyton biomass. Temperature may be an indicator for periphyton biomass as it plays a role in cellular metabolism and brightly lit reaches may have higher stream temperatures than shaded ones irrespective of river discharge.

Conclusion

This work in the Raisin River watershed did not find any significant correlations between periphyton or suspended algal biomass with nutrients or flow. Light and temperature were the most important predictors of suspended algal biomass but light limitation was likely not a significant driver for periphyton biomass because of the river's shallow depth. Periphyton biomass was instead predicted by turbidity and temperature. Periphyton and suspended algal biomass were negatively correlated. This may reflect competition between these two communities for available resources.

Both suspended and benthic algal biomass in the Raisin River was less than predicted by models for eastern Ontario and western Quebec watersheds, based on total phosphorus concentrations. Suspended algal biomass may have been reduced by contact with the substrate due to the shallow depth of the river or losses due to an abundance of native mussels found at many sites. Periphyton biomass may have been reduced by grazer losses that were not investigated or may be diffusion limited, thereby requiring eddy diffusion to replenish the nutrient-depleted boundary layer, which could potentially be delivered through surface water/ groundwater interaction.

It was hypothesized that the effects of nutrients on periphyton biomass might be observed through time at the reach scale and furthermore that groundwater may be influencing algal biomass at this scale. This investigation is pursued later in Chapter 5 at two reaches along the Raisin, while Chapter 4 will first describe an investigation of groundwater-surface water interactions at these two reaches using hydrogeological field methods.

Chapter 4. Surface water-groundwater interactions investigated at the reach scale using piezometers and streambed temperature gradients.

Introduction

Variability in reach-scale groundwater-surface water interaction has been generally attributed to stream topography, hydraulic conductivity of the geologic and geomorphologic features, (Brunke & Gosner 1997). Monitoring techniques such as the seepage meter by Lee (1972) and the mini-piezometer by Lee and Cherry (1979) have long been used to study surface and groundwater interactions showing that local variability in recharge and discharge can exist within a reach (Porter 1996, Storey 2003). Sediment temperature profiling has also been used as a tool for detecting hyporheic flow (White 1987) and groundwater discharge sites at a reach scale (Conant 2004). A recent review of heat as a groundwater tracer can be found by Anderson, (2005).

Porter (1996) found reversals in the vertical hydraulic gradient (VHG) on the Raisin River, indicating a shift from gaining to losing reaches and vice-versa. Other reversals in gaining/ losing sections of a stream detected by use of mini-piezometers and temperature profiles have recently been documented at a tributary to Lake Tahoe, NY (Allander 2003) but no explanation for their cause was provided. The term hysteresis in hydrogeology is typically used to refer to difference in behavior depending on the "history" of the system, for example the difference in moisture characteristic curve depending on whether the soil is wetting or drying. However, it has also more recently been used to describe the response of the pressure head in a sub-riverine aquifers to dramatic fluctuations in river water level, which are delayed due to low hydraulic conductivity of the sediments. This delay produces a hysteretic relationship or loop when hydraulic gradient values are plotted against discharge, giving a curve for the rising limb of the hydrograph that is different than the curve for the falling limb (Evans & Davies 1998, Rose 2003). Reversals of the VHG have been documented in large, highly regulated rivers whereby hysteresis is shown for previous water levels creating a lag in the response of the VHG that is dependent on the hydraulic conductivity of the sediments and river discharge (Arntszen 2006). Our work supports the results of Arntszen and shows that the hysteresis noted in large regulated rivers can also apply to small mid-order streams and rivers.

In order to study the groundwater- surface water interactions in the Raisin River, monitoring of traditional mini-piezometers was combined with temperature profiles to determine discharge and seepage areas. Thermochrome iButton® temperature data loggers were used to monitor surface water to see if they could detect reversals in hydraulic gradient measured by the mini-piezometers and to see if discharge and recharge sites could be detected.

Since previous results indicated that deep groundwater discharge could not be detected along the chosen reaches (Bustros-Lussier 2008), we would expect any surface water/ groundwater fluxes to involve shallow groundwater possibly driven by hyporheic flow.

Study site

The Raisin River comprises the North, Middle and South branches, which combine and flow east into the St. Lawrence River near Lancaster (45° 07' 32" N, 74° 29' 30"W). In the Raisin River drainage basin (546 km^2) land use is dominated by agriculture except near the City of Cornwall to the south. Two study sites were selected based on their well-defined, meandering, pool-riffle sequences and ease of access: St. Andrew's (45° 5'49.16"N, 74°47'55.87"W), an upstream reach on the Middle Raisin, and Macintyre Rapids (45° 9'1.10"N, 74°36'57.10"W), a reach located 22.4 km downstream, after the confluence of the North and Middle branches. Macintyre Rapids was also selected because it was a benthic invertebrate and walleye study site for the Raisin River Conservation Authority, so data that were collected could be used locally. Both sites were on mid-order channels ranging in discharge from 0.4 to 0.9 m^3/sec. At both sites, sediments consisted of glacial till deposits (sandy and clay-rich Malone till) underlain predominantly by limestone, shale, and dolomite bedrock (Terasmae & Mott 1965).

Methods

Mini-piezometer installation and monitoring:

A total of 36 mini- piezometers were installed (Lee & Cherry 1979) over the two reaches to determine groundwater inputs/outputs (July-Oct, 2006). Two to three piezometers (each < 30 cm apart) were installed in 7-8 clusters located throughout each reach (Figure 23 & Figure 24). The location of piezometers within the reach were chosen to sample groundwater- surface water interactions throughout the reach, however due to the high concentration of cobbles encountered in the upper 60 cm of sediment, locations were often limited to where piezometers would go in. The difficulty in installing piezometers could have also been attributed to a shallow overburden, although this was not investigated. The implications of the inability to install piezometers in certain locations are that a bias is created in the results where sediments that either have less cobbles or more overburden are preferentially sampled, although the nature of the bias is not known since it would require at least some knowledge of the hydraulic conductivity of the bedrock aquifer. Groundwater inputs were estimated from hydraulic gradients based on measured hydraulic heads in piezometers at both sites and on hydraulic conductivity estimates. These mini-piezometers were also used as access points to the streambed for temperature-probe measurements at depth.

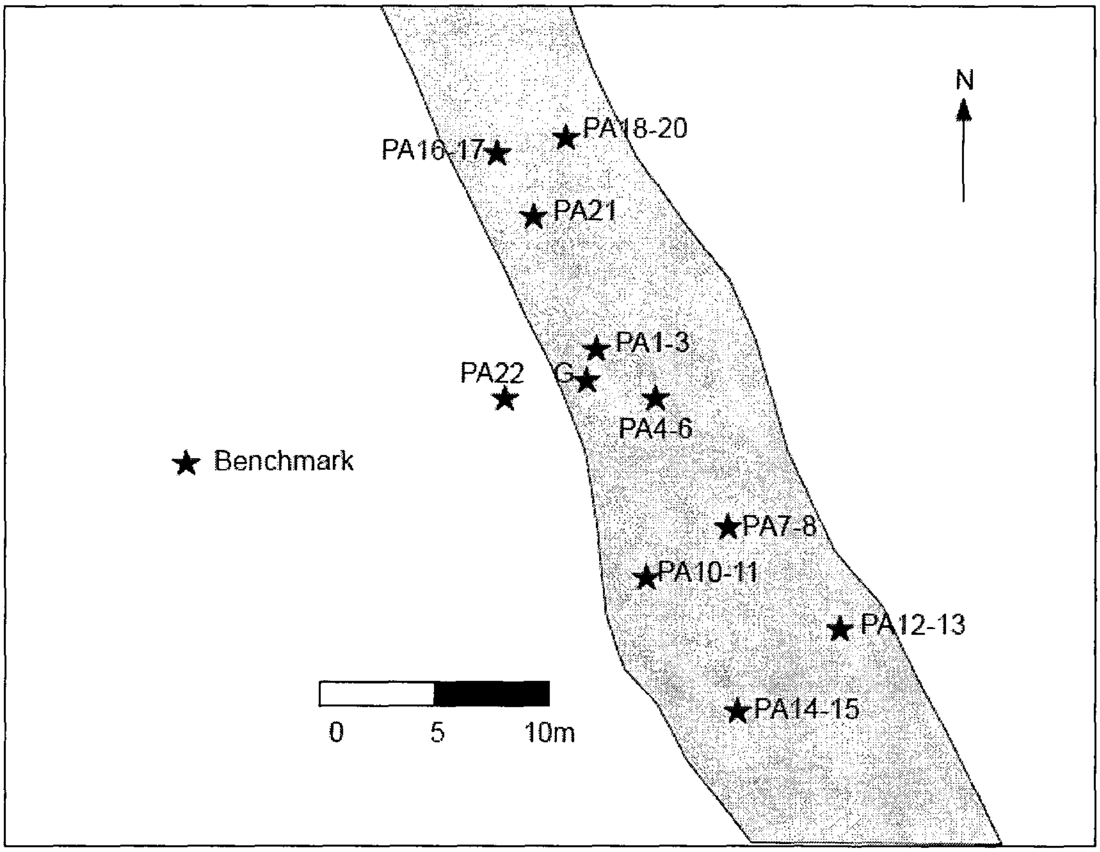


Figure 23. St. Andrew's site with locations of mini-piezometers. PA refers to piezometers installed at the St. Andrew's site. Each marker represents one cluster of between 1 and 3 piezometers. Benchmark is the location of the south-east screw on a bench at 45 5' 48.68789" N, 74 47' 56.852384" W located using the Thales Z-max RTK GPS system. G indicates position of water level gauge. River banks are indicated by blue lines and flow is towards the South-East. Scale bar is 10m.

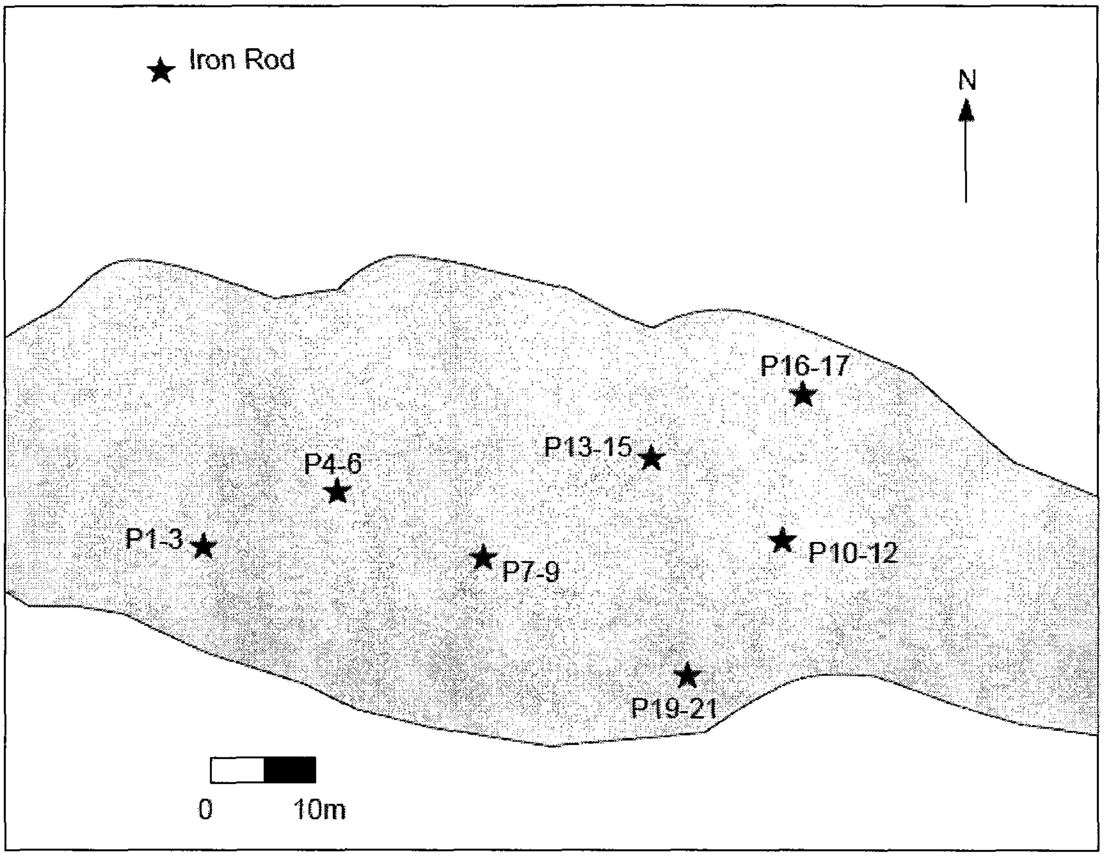


Figure 24. Macinytre rapids site with piezometer locations. P refers to piezometers installed at the Macintyre site. Each marker represents one cluster of between 1 and 3 piezometers. An iron rod was installed as a benchmark at 45 9' 2.546086" N, 74 36' 58.449769" W located using the Thales Z-max RTK GPS system River banks are indicated by blue lines and flow is towards the East. Scale bar is 10m.

Due to the high concentration of cobbles in the upper 60 cm of the streambed, seepage meters could not be installed over the reaches and so groundwater discharge into the river could not be measured directly. Reliable in-situ hydraulic conductivities could not be obtained for the same reason, and representative samples could not be obtained for ex-situ measurements. Instead, hydraulic gradients were used as surrogates for flux, based on the assumption that the hydraulic conductivity was approximately constant. The relationship between the flux, the hydraulic conductivity and the hydraulic gradient is given by Darcy's Law as described in (Freeze and Cherry 1979),

$$q = -K\frac{\Delta h}{\Delta l} \quad (1)$$

where q is the Darcy flux, K is the hydraulic conductivity of the sediment, and $\Delta h/\Delta l$ is the hydraulic gradient, which is the driving force for groundwater movement. The Δh component is the hydraulic head differential between the river bed and the piezometer mid-screen depth and Δl is the depth below the sediment interface where the piezometer mid-screen depth tip is located. The hydraulic head in the river is simply the water level of the river and the hydraulic head at the piezometer mid-screen depth is the water level in the piezometer. For a piezometer installed in the river Δh is simply the difference in water level between the river and the piezometer; if the water level in the piezometer is higher than that of the river then the vertical hydraulic gradient indicates groundwater discharge into the river. In the context of this thesis, discharge is upward flux of water from the aquifer to the river whereas recharge is downward flux of river water to the aquifer. The hydraulic conductivity of the sediment was assumed to be constant throughout the reach (qualitatively verified by digging pits of approximately 30cm deep in the riverbed where sediments were compared and found to be similar). If this assumption was found to be invalid, depending on the range that of hydraulic conductivity throughout the reach, the interpretation of the magnitude but not the direction of groundwater- surface water interaction would change.

Streambed temperature profiles

Streambed temperature profile measurements can be used to estimate groundwater flux through the advective heat transfer from water within the shallow aquifer and the conductive heat transfer through the stationary aquifer material. Temperature measurements are taken at depth intervals of for example, 5cm to a pre-determined depth to obtain a profile. Saturated thermal conductivity of the sediment can be measured in the lab or estimated from literature values so that theoretical flux profiles can be calculated using the equation developed by Turcotte and Schubert (1982) (Conant 2004, Schmidt et al. 2007):

$$q_z = -\frac{K_{fs}}{\rho_f c_f z} \ln \frac{T_{(z)} - T_L}{T_0 - T_L} \tag{2}$$

where:

q_z = specific discharge or Darcy flux [m/s] in the vertical (z) direction

K_{fs} = thermal conductivity of the streambed sediment [$Js^{-1}m^{-1}K^{-1}$]

$\rho_f c_f$ = volumetric heat capacity of the fluid (water) [$Jm^{-3}K^{-1}$]

z = is depth of temperature measurement [m]

$T(z)$ = is the streambed temperature at depth z [°C]

T_L = groundwater temperature [°C]

T_0 = surface water temperature [°C]

Fluxes were compared against hydraulic gradients to corroborate evidence of groundwater discharge and recharge. The thermal conductivity of the streambed sediments was estimated from samples taken at another site on the Raisin River.

Temperature profiles must be taken during summer or winter months where there is a significant difference between temperatures of surface water and groundwater. Since groundwater from wells in the Raisin River watershed has been found to be approximately 9.8 °C for November and December, 2005 and 10.5 °C year-round (Suchy 2008), surface water temperatures below 5 °C are suitable for these measurements; surface water in the winter also does not experience the same daily fluctuations as in the summer (Schmidt et al. 2007). While the Turcotte and Schubert equation (2) makes the assumption of only vertical flux, this equation has been used to describe fluxes in rivers, where a small horizontal component is always present, in the direction of river flow (Conant 2004, Schmidt et al. 2007). Our study must also assume that horizontal flow will not interfere with calculations, and that the horizontal component of the flux will not dominate.

Since the sediments of the reaches chosen on the Raisin River were dominated by cobbles, direct insertion of the temperature probe without damage would have been impossible. Access to the sediments at depth was therefore obtained by inserting the probe down the previously installed mini-piezometers. Temperature of the sediments at a depth of up to 60cm was taken depending on the depth of the piezometer and successive measurements were made at 5cm depth intervals. Compensation for piezometers installed at an angle was made after field measurements were taken. Temperature profiles were taken on Nov. 29th at St. Andrew's and on Dec. 7th at Macintyre rapids.

For measuring streambed temperature profiles, we used a YSI Model 418 temperature probe (YSI Incorporated, Yellow Springs, Ohio) attached to a Digi-Sense Model 93210-50 ThermoLogR Thermister Thermometer (Cole-Palmer Instrument Company, Vernon Hills, Illinois). The probe measures temperature 0.5cm from the tip and is accurate to within +/- 0.03 °C. The wire connecting the thermistor to the temperature probe was duct taped to a metal pole to ensure rigidity and to assure accuracy of depth measurements when inserting the probe down the mini-piezometers (Figure 25A, B).

(A) (B)

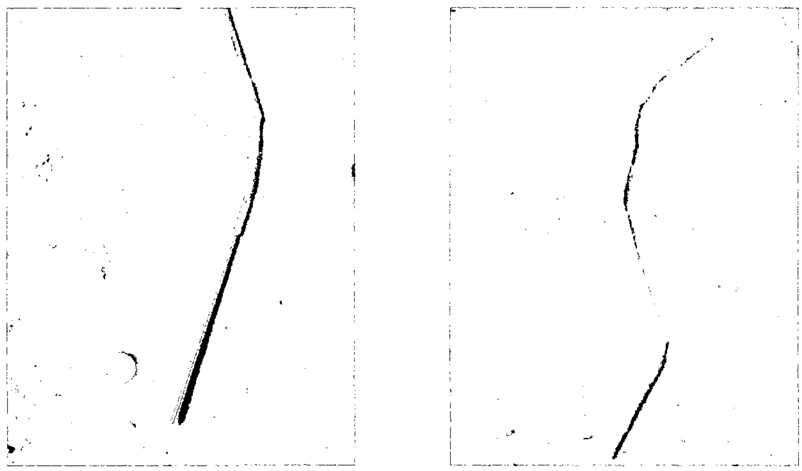


Figure 25. Temperature-probe used to obtain streambed temperatures at depth. Close-up of tip (A) and entire length of rigid section (B) is shown with penny for scale.

IButton® installation:

Thermochrome iButton® temperature data loggers model DS1922L were inserted into a heat-sealed plastic bag and then encapsulated in a 60mL Nalgene© container with approximately six 0.5cm diameter holes and attached to a stake using a tie-wrap. Three iButtons were installed at the reaches. All were installed near the sediment-water interface. IButton K1 was installed on the water level gauge at St. Andrew's (45° 5'49.16"N, 74°47'55.87"W) on August 2nd, 2006 at 1pm and was removed on Dec. 8th, 2006 at 7pm. IButtons K2 and K3 were installed at the Macintyre site at an upstream location on P1-3 (45° 9'1.10"N, 74°36'57.10"W) and at a downstream location on P10-12 (45 9' 1.050749" N, 74 36' 55.668158" W) location within the reach on August 2nd, 2006 and removed Dec. 7th, 2006 at 8pm and on Sept 8th, 2006 and removed Dec. 7th, 2006 at 9pm respectively. Two-factor ANOVAs were used to compare temperatures recorded on each date between sites. Verification of the accuracy of each iButton after they were removed from the river showed that there was some discrepancy between the reported temperature by each iButton and the actual temperature of the surrounding bath water. Temperatures were corrected according to the calibration curves shown in Appendix E. iButtons were verified at the University of Waterloo.

Water levels and river discharge:

Raw data for water levels and river discharge at the Williamstown gauging station was obtained from the Water Survey of Canada. River water levels at St. Andrew's and Macintyre rapids sites were monitored by measuring water levels on stakes installed at one site at St. Andrew's and at two sites at Macintyre rapids. The iButtons were installed on these stakes at the sediment water interface.

Results:

Water levels

Since iButtons were installed at the sediment water interface, water depths measured on the stake that each iButton was attached to could be used to monitor the depth of water above each iButton, over time. Since the goal was to determine if temperature monitoring at the sediment water interface using iButton could help determine locations of groundwater discharge, having similar depths of water above each iButton would assure approximately equal insulation against fluctuating air temperatures. River water level from gauge data showed that iButton depths were greatest at the downstream Macintyre site (iButton K3 attached to the stake for P10-12), followed by the St. Andrew's site (iButton K1) and then the upstream Macintyre site (iButton K2) (Figure 26). All sites followed the trend set by the hourly water level measured by the Environment Canada (EC) Williamstown river gauge and so this gauge was used as a proxy for water levels for both the Macintyre and St. Andrew's sites. The hourly water levels from the EC Williamstown gauge were then justifiably used to define water level fluctuations to compare against piezometric responses at each site.

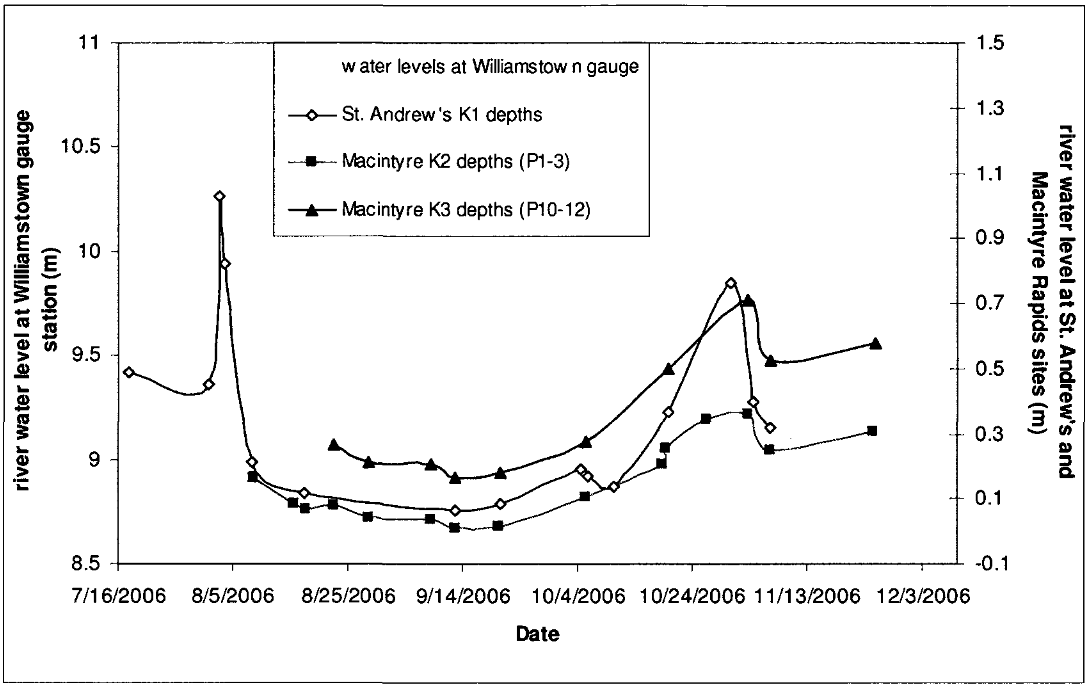


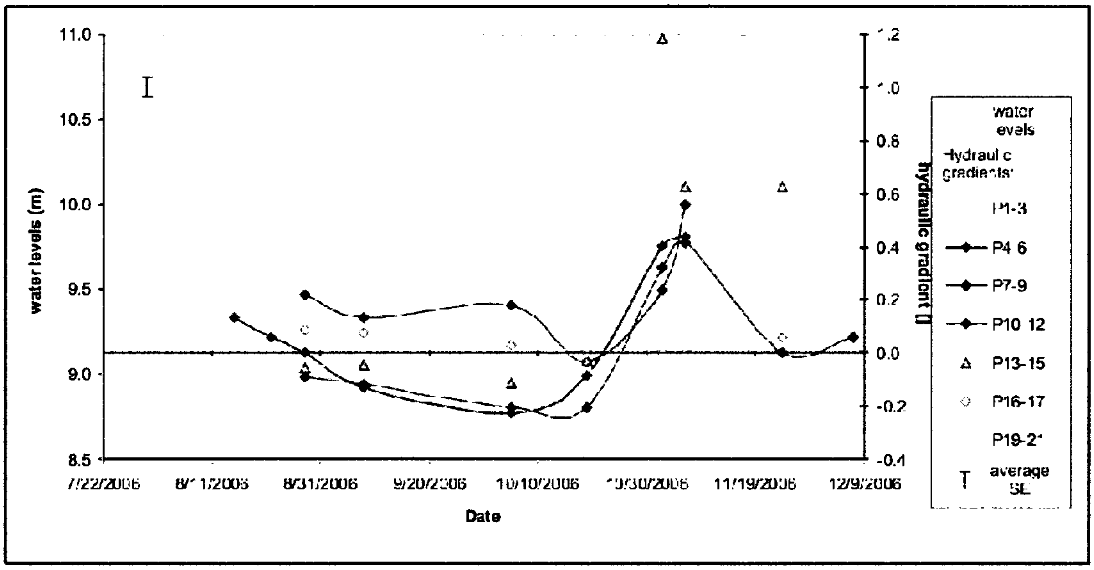
Figure 26. River water levels at locations where iButtons were installed at St Andrew's and Macintyre rapids sites with water levels from Williamstown gauge.

VHGs from piezometers:

Reversals in VHGs were first noted between Aug 15th and Aug 28th at two locations (P1-3 and P4-6) on the Macintyre rapids reach when upward driving forces reversed to downward driving forces upon decreasing water level (Figure 27A). A second reversal was noted between Oct 5th and Oct 19th at the more downstream piezometer P10-12 from an upward driving force to a downward driving force. Finally, four piezometers (P4-6, P7-9, P10-12 and P13-15) all showed a reversal between Oct 19th and Nov 2nd from downward to upward driving forces. Upward driving forces were measured during base flow at downstream piezometers P10-12 and P16-17 indicating possible groundwater discharge; all other piezometers showed downward driving forces, indicating possible groundwater recharge. After Oct 19th when water levels rose, all piezometers measured upward driving forces.

VHG measurements at St. Andrew's began October 5^{th} and therefore measurements did not capture base flow conditions. All VHGs measured at the St. Andrew's site indicated upward driving forces that is the water levels measured in the piezometers were always higher than the water level of the river (Figure 27B). Monitoring frequency at this site was insufficient to report reversals of the hydraulic gradient. From these data no recharge or discharge zones could be interpreted.

(A)



(B)

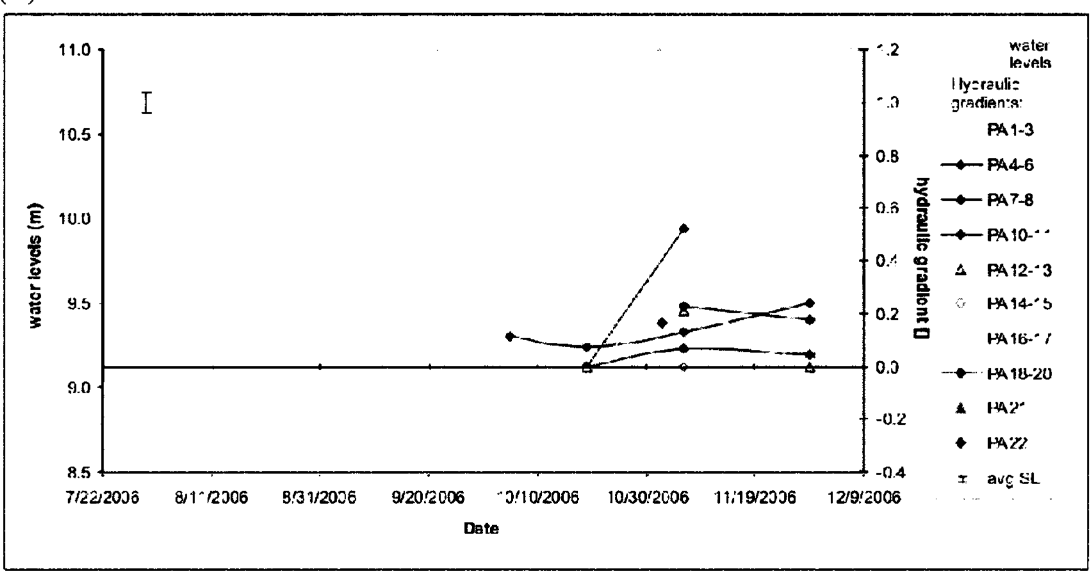


Figure 27. Macintyre rapids (A) and St. Andrew's (B) average VHG per cluster of mini-piezometers over time (n=1-3) with Williamstown river discharge and water level. Positive and negative hydraulic gradients indicate upward and downward driving forces respectively.

About 30m upstream of the Macintyre site piezometers, the limestone bedrock outcrops at the riverbed and so overlying sediments are expected to be shallow at these piezometers (less than 1.5m of overburden). A fault in the bedrock about 2 km upstream of this area is noted on Figure 28, likely indicating a possible connection point to the deeper groundwater in this area. Despite the proximity of this fault, and the shallow overburden in the area, this study lacks the flux measurements and geochemistry data both upstream and downstream of the fault to ascertain whether this fault could represent a preferential pathway for deep groundwater discharge or surface water recharge to the deeper aquifer. A concurrant study in which an electrical conductivity drag-probe at the sediment-water interface was used to detect increases in surface water electrical conductivity as a proxy for deep groundwater discharge did not find any evidence for deep groundwater discharge at this fault (Bustros-Lussier 2008).

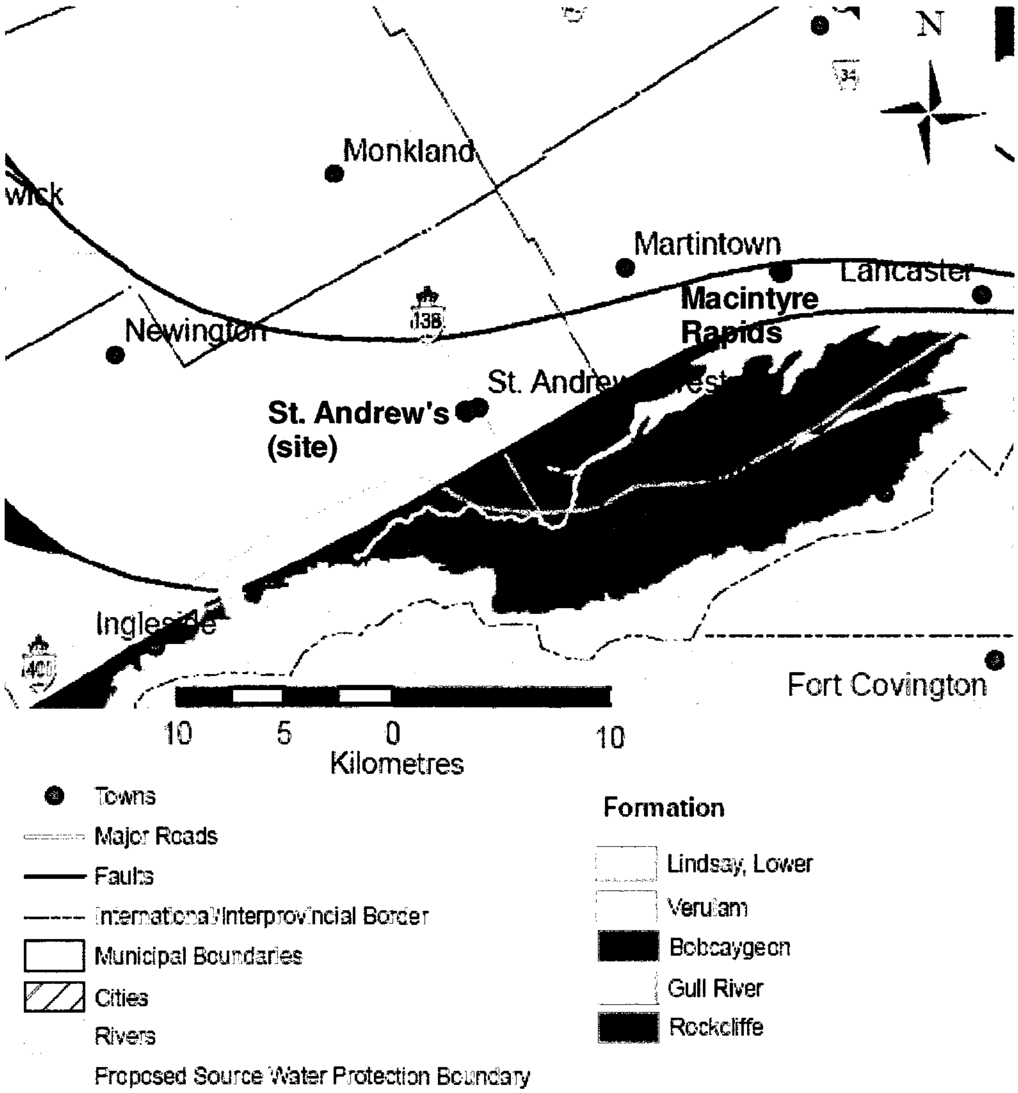


Figure 28. Bedrock Formations and Faults map of the Raisin River Watershed. Green and purple dots show approximate sampling sites locations, as noted. Edited from Map 12: Bedrock Formations and Faults from the Water Budget: Conceptual Understanding for the Raisin – South Nation source water protection region.

Streambed temperature profiles and calculated fluxes:

The Turcotte and Schubert equation was applied to all temperature measurements using the same values for all variables except for surface water temperature, which varied between sites as they were sampled on different dates and as they are situated 20km apart

(Table 9). Sampling ended at approximately 4:00 pm on Nov. 29th at the St. Andrew's site and the surface water temperature used was an average of the temperatures recorded by the iButton at that site over the previous twenty-four hours. The surface water temperature used for the Macinytre site was an average of temperatures from both upstream and downstream iButtons located at this site over the twenty-four hours previous to the end of sampling at approximately 4:00 pm on Dec. 7th.

Table 9. Values used in the Turcotte and Shubert equation (2) including: surface water temperatures (T_0), groundwater temperatures for November (T_L), thermal conductivity of the streambed sediment (K_{fs}) and the volumetric heat capacity of the fluid (water) ($\rho_f c_f$).

$T_0$24g avg at St. Andrew's site	3.796	°C
$T_0$24g avg at Macintyre Rapides site	1.29	°C
T_L	10.4	°C
K_{fs}	0.93	J/ms°C
$\rho_f c_f$	4 190 000	J/m^3°C

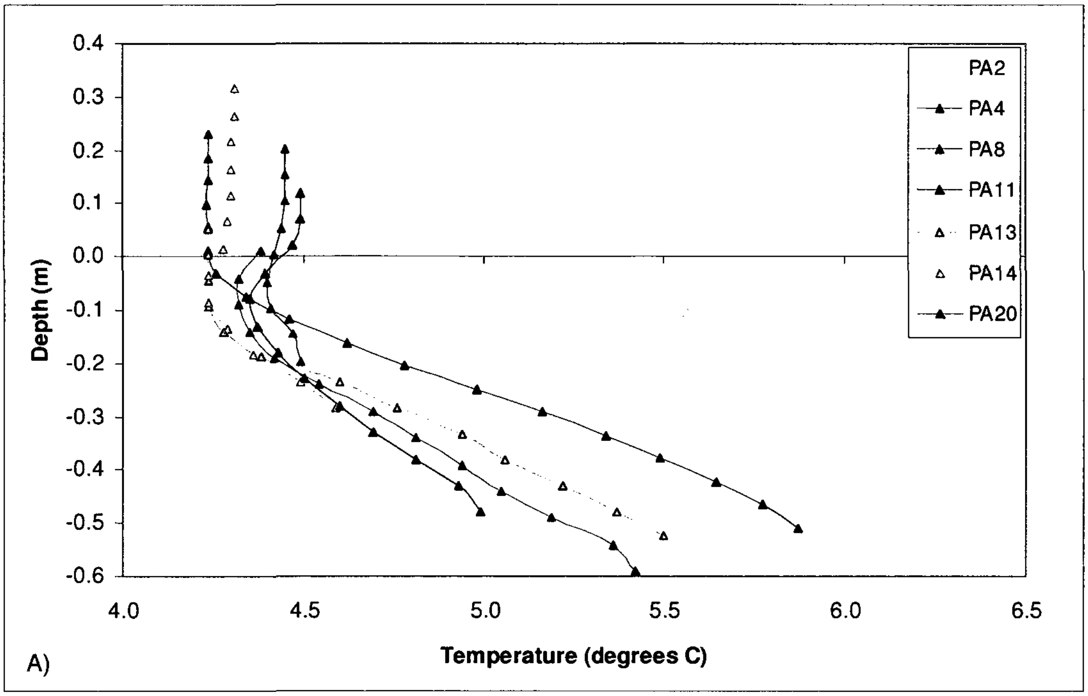
Since the temperature probe was inserted into the piezometers all the way to the bottom and temperatures were read from the bottom up, there was likely a bias toward warmer temperature readings due to deeper warm water displaced by the probe. In retrospect, measurements should have been made from the top of the water column to the bottom. As a result, temperatures measurements at all depths except the deepest were likely warmer than they should; calculated fluxes should therefore represent high-end estimates. However, since this improper technique was used at each piezometer in the same way, one can still compare between piezometers. Because of the silty nature of the sediments, it would be expected that the elevated water level in the mini-piezometer caused by the displacement of water from the probe did not cause water in the mini-piezometer to enter the sediments over the time that measurements were taken (as occurs in a slug-test). Also, temperature stratification within the water column was observed at the Macintyre site, which does not make sense because water should be sufficiently mixed. This observed temperature stratification was likely due to errors in either estimating the depth of each piezometer, or else is a result of mixing of warmer deeper water that has been displaced by the probe. It should be noted that assuming only vertical flux, for a given depth/flux profile, flux should remain static over depths greater than the range of diurnal temperature oscillation (0.2m) found previously (Lapham 1989, Stonestrom & Constantz 2003, Conant 2004, Schmidt et al. 2007). However, in river sediments, we would expect the horizontal component to be relatively large. As a result, horizontal mixing could possibly affect calculated fluxes.

PA2 shows wild variations in temperature and flux in the superficial sediments that may indicate mixing of the water inside the mini-piezometer (Figure 29A and 30A). Such mixing would disturb the readings of the temperatures at depth; however the flux measurements are greater at this location than any others recorded at this site. The measured temperatures and calculated fluxes for PA11 are greater than others except for PA2, but they do not fluctuate as widely, which points to this location (PA11) as the most likely for groundwater discharge at the St. Andrew's site. The temperature measurements for PA4 are very shallow and so we cannot draw conclusions about this location. PA8, 13, 14 and 20 all show similar temperature and flux curves that cannot be differentiated, but present lower temperature and fluxes.

Due to a rougher streambed at the Macintyre Rapids site, we encountered difficulty in determining the depth at which temperature measurements were recorded. As a result, installation depths of piezometers were used as a reference point but since cobble may have moved around over the summer, this measurement was very imprecise. Most profiles do reach temperatures near surface water temperatures recorded for that day; however it is unclear as to whether the temperature probe was above or beneath the substrate when this temperature was recorded due to uncertainties in depth. At best, we can compare the temperatures recorded for the estimated depths and note the greater patterns that were likely not affected by the errors in depth recorded.

The most downstream piezometers, P16 and P17 recorded the warmest temperatures in the substrate and the greatest upward fluxes were calculated for them (Figure 29B and 30B). P16 and P17 were installed in a low-current section at the end of the riffle, off to the side of the channel, where the overburden was thin (as determined by numerous attempts at trying to install at a greater depth). They were also installed at an angle that was greater than most piezometers, as we had to navigate around cobbles and possibly boulders within the sediment. The fluxes calculated for these piezometers are not constant at depths greater than 20cm below the sediment interface, which could be an artefact of the improper technique as described earlier or else horizontal mixing (Figure 30B). However, the warmest temperatures were still recorded for these piezometers, suggesting that warmer groundwater may be discharging into the river at this location.

The next-most downstream piezometer P10 recorded the second warmest temperatures in the substrate and the second greatest positive fluxes were calculated Figure 29B and 30B). This piezometer was installed at the transition between the riffle and pool where the streambed gradient was the steepest, but it was not directly located in the fastest moving surface water. The most upstream piezometer P2 recorded the coldest temperatures for all depths and had the lowest calculated fluxes (Figure 29B and 30B). All other piezometers located in the central riffle area recorded temperatures that were relatively similar over the corresponding depths as well as having similar fluxes and both fluxes and these temperatures were intermediate between the most upstream and most downstream piezometers.



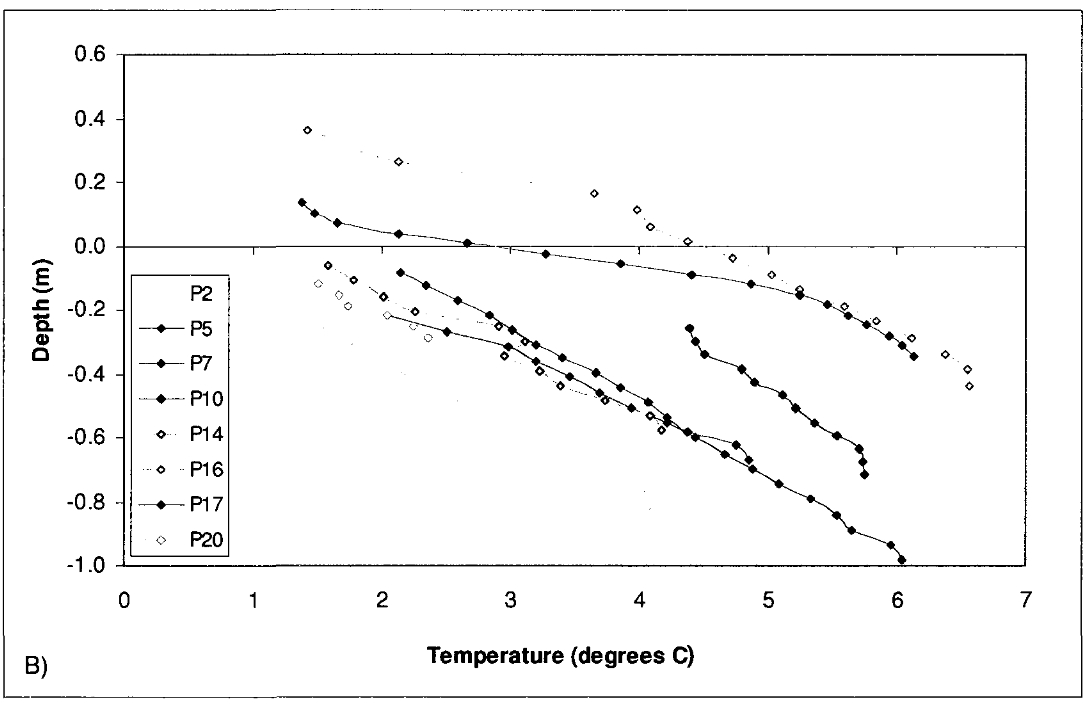


Figure 29. Streambed temperature probe profiles at St. Andrew's. (A) and Macintyre rapids (B) taken on Nov 29th, 2006 and Dec 7th, 2006 respectively (note differences in axis). Temperature in the streambed decreases as groundwater mixes with cooler surface water near the surface.

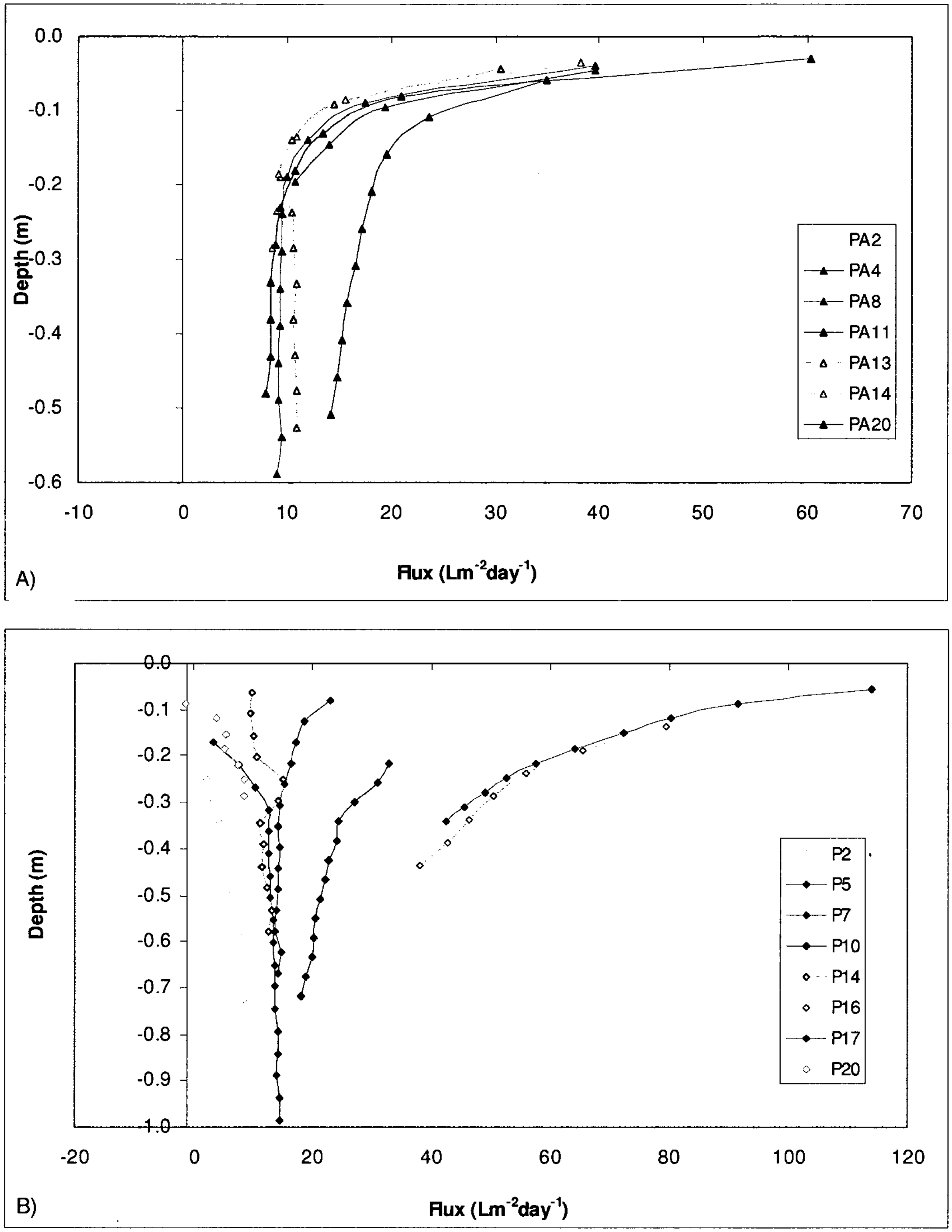


Figure 30. Calculated flux at St. Andrew's (A) and Macintyre rapids (B) sites from temperatures profiles shown in Figure 29 (note differences in axis). Upward flux is represented as positive numbers

IButton® data:

Surface water temperatures can vary with inputs of groundwater discharge (Walling & Webb, 1992). We used iButtons to record hourly surface water temperatures to see if groundwater discharge could be detected. We expected surface water temperatures to be colder during the summer where there was more groundwater discharge. Surface water temperatures were recorded using I-button© at both Macintyre and St. Andrew's sites for the period of Aug. 2 to Oct. 17th 2006, or during base flow. One iButton was installed at the upstream St. Andrew's site and the other two were installed at an upstream location and at a downstream location at the Macintyre site. The three sets of iButton hourly temperature data were analysed to determine if temperatures were significantly different between locations and the results are presented in Table 10. Two-factor ANOVAs were used to determine whether temperatures were significantly different between sites, over time. Overall, St. Andrew's (iButton K1) had colder surface water temperatures than the Macintyre site (iButtons K2 and K3). Among the two locations at the Macintyre site, the downstream location (iButton K3) has warmer temperatures than the upstream location (iButton K2) from Sept. 8th to Oct 17th, 2006 (Table 10). From Oct. 17th to Dec. 7th 2006, as water levels rise, the downstream Macintyre site continued to have the highest recorded temperatures while there was no difference between temperatures recorded at St. Andrew's compared with the upstream Macintyre site. (Table 10).

Table 10. Average differences in hourly surface water temperatures recorded by iButtons K1 (on the gauge at St. Andrew's), K2 (on P1-3 at the Macintyre Rapids site) and K3 (on P10-12 at the Macintyre Rapids site). P-values reported from two-factor ANOVAs.

	dates evaluated	Average	standard error	n	p
all values					
K3-K2	Sept 8th 4pm to Dec. 7th, 7pm	0.799	0.005	2164	0
K2-K1	Aug 2nd, 5pm to Dec 7th, 7pm	0.758	0.025	3051	2.3E-174
K3-K1	Sept 8th 4pm to Dec. 7th, 8pm	1.126	0.019	2164	0
base flow (ends Oct 17th, at 5pm)					
K3-K2	Sept. 8th, 4pm to Oct 17th, 5pm	0.831	0.011	938	0
K2-K1	Aug 2nd, 5pm to Oct 17th, 5pm	1.297	0.035	1825	1.9E-223
K3-K1	Sept. 8th, 4pm to Oct 17th, 5pm	1.647	0.010	938	3.1E-291
fall river discharge increase (starts Oct 17th, at 5pm)					
K3-K2	Oct 17th, 5pm to Dec. 7th, 7pm	0.774	0.005	1227	0
K2-K1	Oct 17th, 5pm to Dec. 7th, 7pm	-0.044	0.018	1227	0.013
K3-K1	Oct 17th, 5pm to Dec. 7th, 8pm	0.728	0.018	1228	3.2E-224

Changes in air temperatures preceded changes in surface water temperature as expected (Figure 31). High intensity storms in July and early August showed correlation with peaks in river water level whereas base flow resumed in late August and September to be followed by a rise in water levels in mid-October. (Since Environment Canada tipping gauge data was unavailable for winter months, precipitation data could not be shown after October.)

Daily fluctuations in surface water temperatures were dampened as water levels rose in early fall. This reduction of diurnal temperature fluctuations were likely due to greater insulation from increased water levels as iButtons were placed at the bottom of the river.

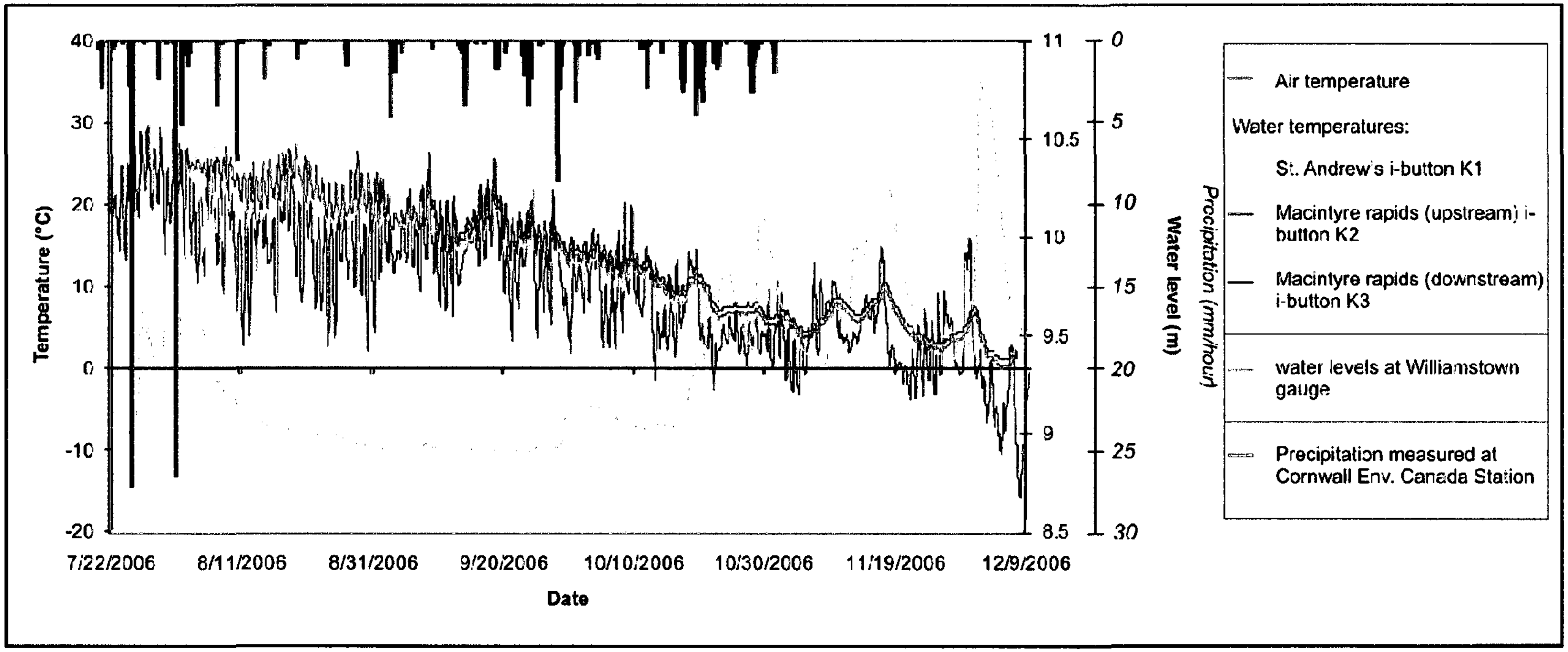


Figure 31. IButton temperature loggers show hourly record of surface water temperatures (Aug-Dec 2006). Water level (measured at Williamstown, precipitation and air temperature data (measured at Cornwall's Environment Canada station). No precipitation data was available after October since Environment Canada removes its tipping bucket gauges for the winter months.

Discussion:

Discharge and recharge zones derived from hydraulic gradient and temperature profiles:

Despite the difficulties in determining the depth of each temperature probe measurement in the profiles recorded at the Macintyre site, the data showed a consistent pattern with the hydraulic gradient data, which suggested the presence of reach-scale recharge and discharge patterns. The downstream piezometers P10-12, P16 and P17 generally showed the largest hydaulic gradients (Figure 27B) as well as the warmest temperature profiles and greatest upward fluxes (estimated from temperature gradients) (Figure 29B and 30B), indicating possible groundwater discharge at the end of the riffle. The more upstream piezometers P1-3, P4-6 and P7-9 show the strongest downward gradients (Figure 27B), some of the coldest temperature profiles and lowest fluxes (Figure 29B and 30 B) indicating that the upstream portion of the riffle might have been a site where surface water recharges the phreatic aquifer. It should also be noted that the temperature probe measurements from December 7th were taken at a time when water levels had recently receded after a sudden increase in water levels and so discharge might have been artificially high due to lagged inputs from bank storage due to low hydraulic conductivity of the sediments.

Longitudinal differences in water levels and the presence of pools and riffles in rivers cause downstream flow and localized small discharge and recharge zones from the hyporheic zone (Brunke & Gosner 1997). In mountain streams, reach-scale hyporheic flow has been well monitored and models predict spacing of up and down-welling segments according to stream order (Gooseff 2005) or along the continuum (Anderson 2005). Low gradient rivers with low permeability sediments however, do not necessarily show evidence for hyporheic flow (Wright 2005). While we cannot generalize to the entire Raisin River watershed, hyporheic flow at the reach scale appears to be present at the Macintyre site whereby surface water infiltrates at the upper reaches of a riffle zone and shallow groundwater is discharged to the river at the end of the riffle as supported by our hydraulic gradient and temperature probe data. This finding is consistent with others, studying a similar mid-order river (White 1987).

By contrast with the Macintyre site, we can derive relatively little from the hydraulic gradient and temperature probe measurements obtained at the St. Andrew's site mostly due to scarcity of data in hydraulic gradients. While Macintyre Rapids was monitored from August to November, St. Andrew's was only monitored from mid-October to November, showing the importance of monitoring during base-flow when groundwater likely plays a proportionally greater role than during the fall. Also, since the St. Andrew's site is smaller, the length of the riffle may have inhibited our ability to detect hyporheic flow at the riffle scale. Some evidence does exist for localized zones of discharge and recharge from the temperature profiles, as the most upstream piezometer PA20 shows recharge, at least at superficial depths and PA11, a piezometer near the end of the riffle, is the most likely candidate for groundwater discharge to the river (Figure 29A and 30A). Calculated fluxes from other piezometers on this reach show similar low discharge values that are likely a result of shallow groundwater discharge returning from bank storage after the most recent precipitation event. We could not conjecture that they represent discharge zones at other times such as during base flow.

iButton data:

iButton data from the St. Andrew's site consistently shows lower temperatures during the summer months and relatively higher temperatures during the winter months than both iButtons from the Macintyre site. This could be due to a greater groundwater input that would reduce the surface temperatures in the summer and increase temperatures in the winter. Since groundwater in the region is approximately 10.5 °C year round (Suchy in progress), summer surface water temperatures will be cooled by discharging groundwater and conversely groundwater will warm surface water temperatures that are cooler than 10.5 °C. However, if this were the case, we would expect surface water temperature loggers (located on the sediment/water interface) to detect colder water during hydrograph recession, when shallow groundwater discharge should be at its highest, and this was not observed (Appendix F). If groundwater discharging to the river was from shallow sources, having a temperature that is near precipitation, surface water temperatures might not have been affected and so iButtons might not have been able to detect shallow groundwater recharge to the river. Also, during the fall there is a transition period when decreasing surface water temperatures would no longer be cooled by discharging groundwater.

Because the two sites chosen were very different in width, shading could have played a role in the surface water temperature differences recorded by the iButtons. Shading can have an important effect on surface water temperatures, especially daily peak temperatures (Johnson 2004). The lower summer surface water temperatures at St. Andrew's could have been due to greater shading at the upstream St. Andrew's site compared to the downstream Macintyre site. Both sites had similar water depths, but upstream sites are typically narrower and so have more shade from the riparian zone. To test this theory, a prediction was made that while surface water temperatures would vary with air temperature, upstream surface water temperatures would be consistently cooler than those measured at the downstream site during the daytime. At night, this difference in upstream and downstream temperatures should drop to near zero as both sites would cool, but the warmer downstream site should cool more quickly as it is relatively warmer compared to the night time air temperature. This pattern was evident when comparing either of the iButtons at the Macintyre site with the iButton at the St. Andrew's site, also no such pattern existed between upstream and downstream iButton temperatures at the Macintyre site. For this reason, the two iButton logs at the Macintyre site were averaged and compared against the log from the St. Andrew's site. The average temperature of the iButton logs at the Macintyre site, minus the temperature from the iButton log at the St. Andrew's site is herein referred to as ΔT. Spectral analysis was used to show that the periodicity of ΔT over time indeed correlated with air temperature daytime peaks during base flow with a period of 24 hours (Figure 32a), where a large peak in the spectral density of the cospectrum is evident at 24 hours and a small one exists at 12 hours. There were also small peaks at 12 and 24 hours in the imaginary part of the cross-spectrum, termed the quadrature. When water levels rose in the fall (from October 17^{th} at 5pm, until the end of the temperature logs), the cospectrum peak at 12 hours disappears and the 24 hour peak is greatly diminished (Figure 32b). Increasing water levels likely insulated the temperature probes from fluctuations in air temperatures at the un-shaded site. The coherency of the12 and 24 hour peaks in the cospectrum were significant, (coherency > 0.95) as shown by peaks at these wavelengths in coherency, (Figure 32c-d). Further support for shade explaining the cooler temperatures at the St. Andrew's site could be made by comparing sunny vs. overcast days.

A lack of deep groundwater discharge into the Raisin River would be supported by other recent findings (Bustros-Lussier 2008) and so we would tend toward the explanation of shading to explain cooler temperatures at the St. Andrew's site.

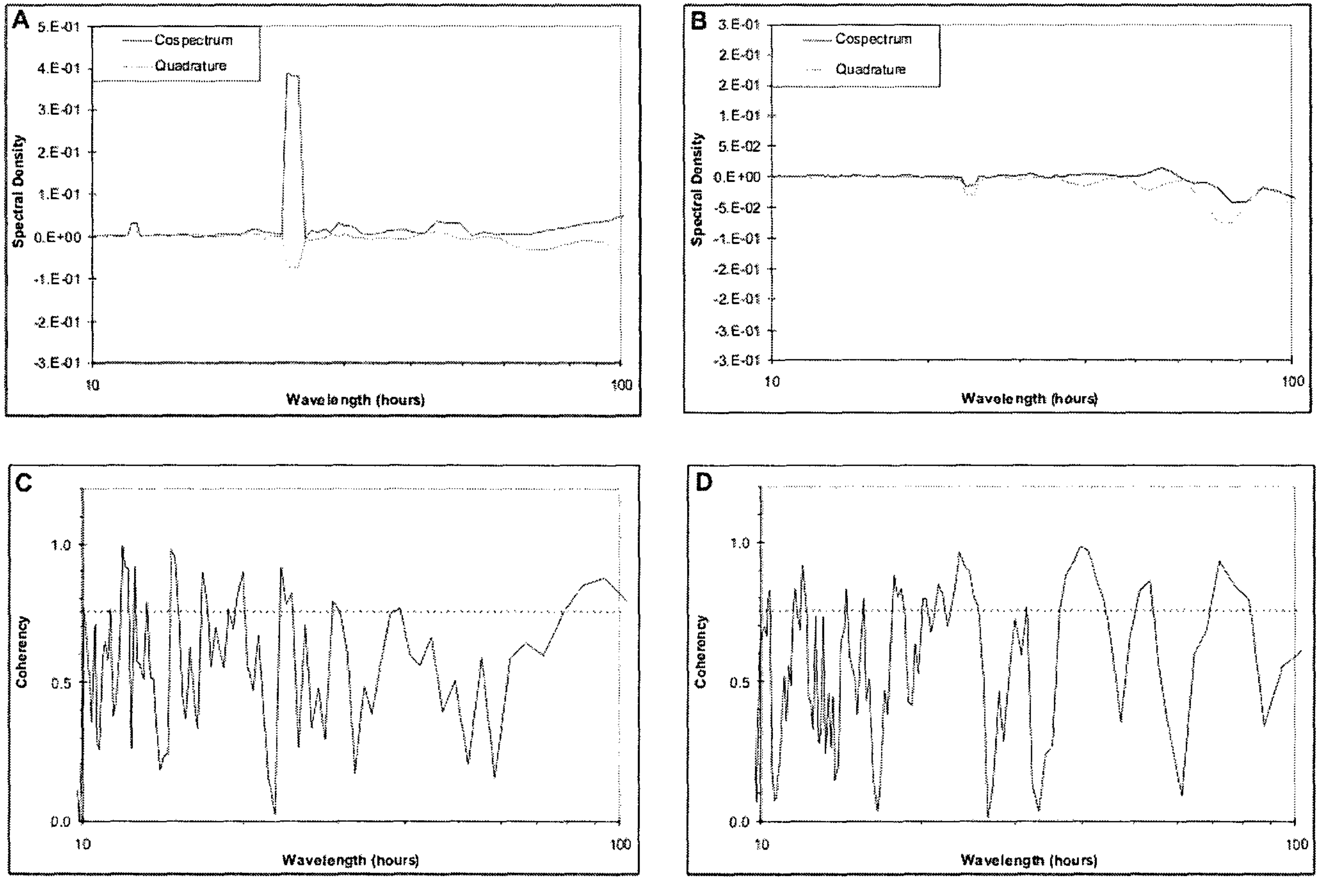


Figure 32.Spectral analysis for surface water temperature differences between the St. Andrew's site and the Macintyre site (delta T) and air temperatures. a) and b) show the cospectrum (real component of the cross-spectrum) and quadrature (imaginary component of the cross-spectrum) comparing delta T with air temperature during base flow and after water levels rise. c) and d) show coherency for the power spectrum of these variables for (among others) 12 and 24 hours where the dotted line shows a level of significance of 95%. Wavelength (i.e. Measurement scales), where the coherency is above the 95% confidence level corresponds to wavelength where the cross-spectrum estimates are significant (at the 95% level).

Reversals of the VHG:

Reversal of the vertical hydraulic gradient such as observed in this study (Figure 25) was previously reported for the Raisin River but with no explanation (Porter 1996). During heavy precipitation events, river levels typically increase more rapidly than the piezometric levels (Abdul 1989, Squillace 1996). Although this behaviour is well known for near-stream piezometers (Abdul 1989), it is not as widely reported for in-stream piezometers (but see Squillace 1996, Storey 2003, Arntszen 2006). One interpretation is that an increase in stream level relative to the in-stream piezometric levels would typically drive channel water into the hyporheic zone and also into the banks (Squillace 1996, Wondzell 1996). The horizontal component of hyporheic flow would thus remain more or less constant (since governed by longitudinal differences in water levels), but the vertical downward flow would now dominate throughout the reach, even in areas of local hyporheic discharge during base flow. The time it would take for piezometric water levels to equilibrate with the higher water level in the river would however depend on the hydraulic conductivity of the sediments, which is low for the fine-textured sediments of the Raisin River watershed. If river levels dropped quickly after a storm event, in-stream piezometric levels would experience a delayed response, showing initially and potentially prolonged higher piezometric water levels as described in Arntszen (2006). This interpretation is supported by the work of Bustros-Lussier (2008), which showed that base flow in the Raisin River appears to be predominantly from shallow groundwater.

Conclusions:

Reversals in the VHG appear to be the result of the combined low hydraulic conductivity of the sediments and the variability in water levels typical for such mid-order rivers. These piezometric responses are consistent with the model for hysteresis proposed by Arntszen (2006) for larger, regulated rivers.

Since previous results indicated that there was no deep groundwater discharge present along the chosen reaches (Bustros-Lussier 2008), we would expect any surface water/ groundwater fluxes to involve shallow groundwater. While hyporheic flow on the Raisin River at the reach scale is supported by our hydraulic gradient and temperature probe data at the Macintyre site, only some evidence for hyporheic flow from temperature probe data exists at the St. Andrew's site but this may be due to its small size. The finding that such low-gradient mid-order rivers show reach-scale hyporheic flow is consistent with some (White 1987), but differ from other watersheds with similar gradient and geology (Wright 2005). This could be due to differences in hydraulic conductivity of the substrate that we did not investigate in full.

Large storm events in the watershed induce great fluctuations in water levels of the Raisin River, such as those in Jul. – Aug. 2006 and the rise in water levels during the fall can be quick (Figure 31). Sudden water level rises can lead to bed movements and a reversal in the hydraulic gradient at sites that usually exhibit recharge which is important in conserving hydraulic conductivity in superficial sediments, preventing colmation or, the retention processes that lead to clogging of the riverbed by fine particles (Schälchli 1992). Variability in water levels of mid-order streams also promotes shallow groundwater-surface water interactions, which have been shown to be important for the transformation and retention of nutrients (Triska et al. 1989). This is likely an important factor in maintaining ecological health in the Raisin River in buffering against environmental stressors such as increased nutrient load and organic matter due to factors such as the local presence of agriculture and increased urbanization within the watershed.

Chapter 5: Evaluation of the influence of groundwater on algal biomass and accrual at the reach scale in a shallow river (Raisin River, South Eastern Ontario, Canada) (Modified from Woods et al. 2009)

Introduction

Factors determining phytoplankton and periphyton biomass are less well understood for rivers and streams than for lake ecosystems. For phytoplankton, researchers are divided as to whether river hydrology is more important in determining biomass (e.g. Jones & Barrington 1985, Round 1988) than nutrient concentrations (e.g. Basu & Pick 1996, Van Nieuwenhuyse & Jones 1996). For periphyton, there is a similar debate as to whether nutrients are the most important factor influencing algal biomass (e.g. Chételat et al. 1999).

Using a multiple river model, Chételat et al. (1999) found a significant correlation between epilithic periphyton and water column total phosphorus (TP) concentrations in Eastern Ontario and Western Quebec rivers. In contrast, this work in a single river system (the Raisin River, Ontario) during base-flow did not find any significant correlations between periphyton biomass and nutrients or flow, and concluded that light limitation was not a significant factor either because of the river's shallow depth. It was hypothesized that the effects of nutrients might be observed through time at the reach scale and furthermore that groundwater may be influencing algal biomass at this scale. In oligotrophic mountain streams, benthic algal growth has been shown to increase in retention pools where interstitial water containing nutrients is released from the sediments (Coleman & Damn 1990). The study objectives were to: 1) investigate the relationship between algal biomass, nutrients and flow over time at the reach scale and 2) characterize groundwater- surface water interactions to determine their effects on periphyton accrual (net growth).

Key Words: accrual, hyporheic, nutrients, periphyton, reach scale, river

Methods

The study sites used in this Chapter are the same as described previously in Chapter 4. Both St. Andrews and Macintyre rapids reaches were sampled twice a week (May to October, 2006) for water chemistry including nitrate (NO_3) and total phosphorous (TP). Temperature, specific conductivity (SpC), oxidation-reduction potential (ORP), dissolved oxygen (DO) and pH (Hydrolab Minisonde 4 Multiprobe Logger) were also measured (Hydrolab Minisonde 4 Multiprobe Logger) at each sampling; nutrients were analysed upon return to the University of Ottawa. River discharge was measured on most samplings at each reach using a current velocity meter (Swoffer model 2100) and then multiplying by cross-sectional area of the reach. These discharge estimates were then compared against the Water Survey of Canada (WSC) gauge "Raisin River near Williamstown" (45°9'19" N, 74°38'16" W), located 2km upstream of the Macintyre reach, so that the WSC station could be used to estimate discharge.

Mini-piezometers were installed and monitored as described in Chapter 4 except that groundwater inputs were estimated from hydraulic gradients calculated based on measured hydraulic heads in piezometers at the Macintyre Rapids reach only (piezometers installed at the St. Andrew's reach were used only for geochemical analyses and not for hydraulic gradient estimates because of time constraints). Darcy's Law was applied to hydraulic head measurements as a surrogate for flux as described in Chapter 4.

Hyporheic groundwater was collected from all piezometers on the same date once all had been installed (Nov 9^{th}, 2006). Piezometers were first pumped two days prior to sampling to allow fresh groundwater to enter the piezometer. NO_3, TP and chloride were analysed from the hyporheic groundwater samples at the University of Ottawa. Deeper groundwater was collected Nov 11th from nearby wells for chloride and NO_3 analysis only. In the statistical analyses, non-detected values were assigned values half-way between zero and the detection limit.

Periphyton standing stock was sampled by collecting rocks at each piezometer for both reaches and at additional locations chosen to cover the reach to represent all microhabitats. Rocks were scrubbed to remove periphyton, which was homogenized and collected on filters for chlorophyll a (Chl-a) analysis (Burnison 1980). Rock surface area was estimated by weighing foil required to cover the rock and then calculating surface area from the foil surface area to weight relationship. Periphyton accrual was estimated on 2-3 dates over a 14-day period at each piezometer at the Macintyre Rapids reach using unglazed tiles (4.8 cm x 4.8 cm). Periphyton (Chl-a) biomass on tiles was analysed following Wintermans & DeMots (1965), except that Chl-a was extracted directly from the tiles.

Results

Nutrients and Geochemistry:

Nutrient concentrations (NO_3, TP), other chemical and physical variables generally did not differ significantly between reaches when compared using a two-way factorial ANOVA to remove the influence of dates sampled ($p > 0.05$ for all parameters except that Macintyre rapids had a higher pH than St. Andrew's reach ($p < 0.001$) (Table 11). River discharge at the downstream Macintyre Rapids reach was on average twice that of St. Andrews' reach based on three dates when discharge was measured at both sites. Measured discharge at the

Macintyre reach correlated well with discharge from the WSC gauge ($r^2 = 0.99$, $p < 0.001$, n = 10 dates). Although a significant correlation was observed at St. Andrew's ($r^2 = 0.96$, $p < 0.001$, n = 7 dates), it was the result of a single outlying value.

Table 11. Means and ranges of variables measured in two reaches of the Raisin River, between May 29th and Nov 9th, 2006. Abbreviations defined in the text.

	St. Andrew's reach			Macintyre Rapids reach		
Variable	Mean	Range	n (days sampled)	Mean	Range	n (days sampled)
Temperature (°C)	17.91	6.78-27.4	11	19.30	7.25-29.45	13
pH	7.60	7.22-8.10	11	7.93	7.38-531.7	13
SpC (µS/sec)	487	339-560	11	471	326-532	13
DO (mg/L)	8.76	7.27-10.5	10	10.40	6.92-13.64	11
ORP (mV)	508	371-615	10	511	342-634	12
TP (µg/L)	56	34-79	10	52	34-52	12
NO_3 (µg/L)	293	7-980	10	378	4-972	12
Max depth (m)	0.63	0.37-0.88	7	0.50	0.28-0.74	12
Discharge (m^3/sec)	1.92	0.39-7.53	7	4.16	0.36-9.02	12

Nitrate and discharge have been shown to be closely tied in Southern Ontario basins (Hill 1978), following a linear relationship; however, the discharge-nitrate relationship at Macintyre rapids was best described by a second order polynomial equation:

NO_3 (mg/L) = -0.0118 (discharge m^3/sec)^2 + 0.2101 (discharge m^3/sec) - 0.0021 ($p < 0.0001$, r^2: 0.874). This type of relationship likely resulted from the coincidence between high discharge, storm events and moderate NO_3 concentrations. (The discharge-nitrate relationship at St. Andrew's was not investigated since a gauge closer to this reach would have been needed).

Hyporheic groundwaters collected from piezometers at Macintyre Rapids showed higher chloride concentrations than surface waters ($p = 0.02$) but not significantly different from groundwater in nearby wells ($p = 0.45$) despite a lower average concentration in piezometers than well water (Table 12). A significant difference in chloride concentration between surface water and well water was observed ($p < 0.001$). Hyporheic groundwater from St. Andrew's piezometers had elevated chloride levels compared to surface water ($p < 0.001$) or well water ($p = 0.001$) and there was no significant difference between surface water and well water ($p = 0.91$).

Table 12. Geochemistry of groundwater, surface water and piezometer water. Average concentrations (ppm ± SE) of chloride, nitrate and total phosphorous from piezometers (Nov. 9th, n = 13), surface water (Nov. 9th, n = 6 locations within the reach) and from wells (Nov. 11th, n = 2).

	Cl		NO_3		TP	
St. Andrew's reach:						
Wells	15.4	(4.2)	<0.5	n/a	n/a	n/a
Piezometers	123.5	(9.6)	0.010	(0.004)	0.067	(0.004)
Surface water	14.3	(1.8)	0.476	(0.004)	0.064	(0.001)
Macintyre Rapids reach:						
Wells	34.2	(1.2)	<0.5	n/a	n/a	n/a
Piezometers	26.7	(4.2)	0.6	(0.2)	0.062	(0.005)
Surface water	15.5	(1.4)	0.7	(0.1)	0.065	(0.002)

Nitrate concentrations from piezometers were lower than those from surface waters at St. Andrew's reach ($p < 0.001$) but not at Macintyre Rapids ($p = 0.67$) (Table 12). Total phosphorous concentrations from piezometers at St. Andrew's or Macintyre Rapids reaches were not significantly different from surface waters ($p = 0.75$ and $p = 0.71$).

Periphyton biomass (natural substrate):

Periphyton biomass correlated with TP in early summer (Figure 33a) at the Macintyre Rapids site before a major storm in August ($p = 0.002$, $r^2 = 0.97$, $n = 5$). No correlation was seen at St. Andrew's likely due to low sample size ($n = 3$) (Figure 33b). Biomass at both sites declined sharply after a large storm event on August 1st, after which it recovered, reflecting nutrient concentrations.

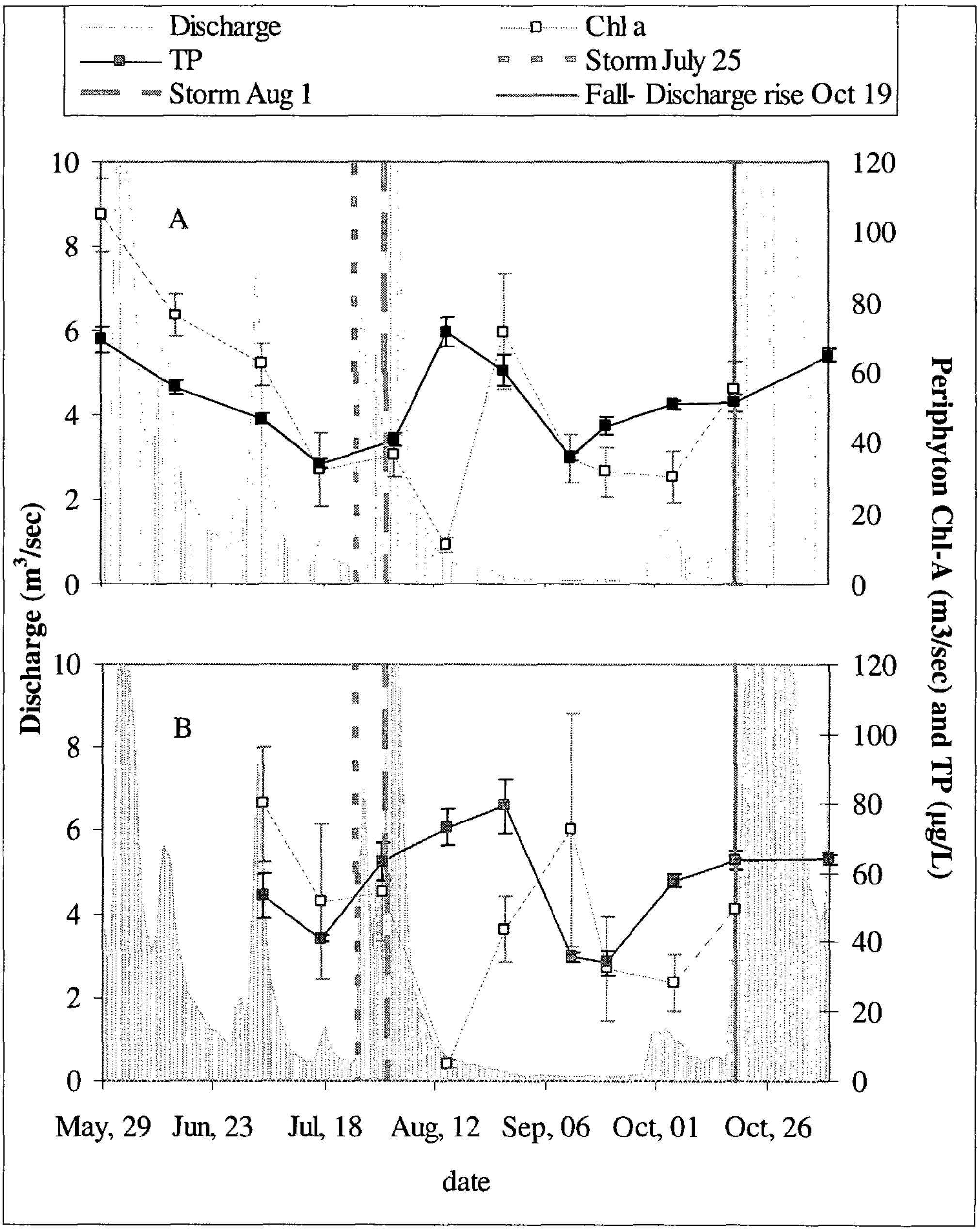


Figure 33. Seasonal variation in TP and periphyton biomass on rocks at downstream Macintyre Rapids (A), and upstream St. Andrew's reaches (B) superimposed on discharge measured upstream of Macintyre Rapids at Williamstown. n = 4-6 for TP, n = 4-19 for Chl-*a* Error bars ± SE.

Periphyton accrual (tiles):

There was a greater difference in periphyton accrual on tiles between piezometer clusters than within each cluster, collected on the same date ($p = 0.001$). Similarly, with respect to hydraulic gradients, there was a greater difference between clusters than within clusters ($p < 0.001$). Generally, hydraulic gradients during low flow indicated a greater downward flux in the upstream part of the reach compared to downstream (locations A-D, Figure 34) [range; average, n]: A) [-0.28 to 0.00; -0.09, 35], B) [-0.16 to 0.09; -0.01, 6], C) [-0.13 to 0.30; 0.08, 12], D) [0.00 to 0.07; 0.04, 4]. The direction of vertical flow reversed at some point throughout the study in 4 of the 17 piezometers. Most piezometers in the reach showed downward flow most of the time (representing a loss of river water into the underlying aquifer) and showed a trend of increasing algal accrual with increasing flux (Figure 35, $p = 0.048$, $r^2 = 0.29$, $n = 14$). Values exhibiting upward groundwater flow into the river also showed a positive trend with algal accrual but were not numerous enough to ascertain the significance of the trend (Figure 35, $p = 0.07$, $r^2 = 0.86$, $n = 4$).

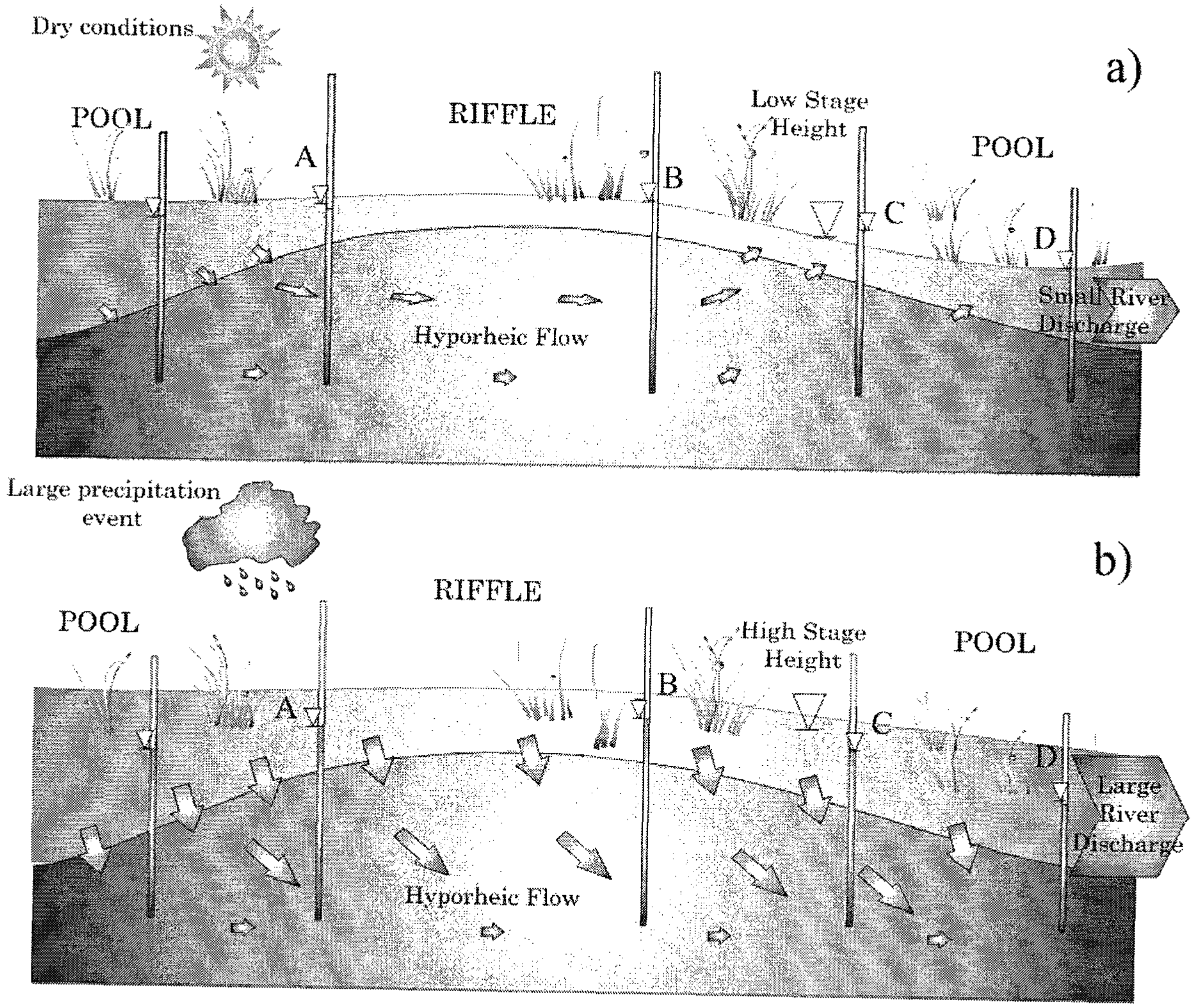


Figure 34. Effect of a large precipitation event on reach-scale hyporheic flow. (a) Base flow conditions showing predominantly horizontal hyporheic flow throughout the reach, with local recharge and discharge. (b) Storm flow conditions over the same reach showing predominantly downward hyporheic flow with a minor horizontal component. The increased downward force (hydraulic gradient) due to the higher stage immediately after a large precipitation event forces more water into the sediment, contributing to vertical hyporheic water and surface water exchange. Measured hydraulic gradients A-D at Macintyre rapids during low flow periods are given in the text.

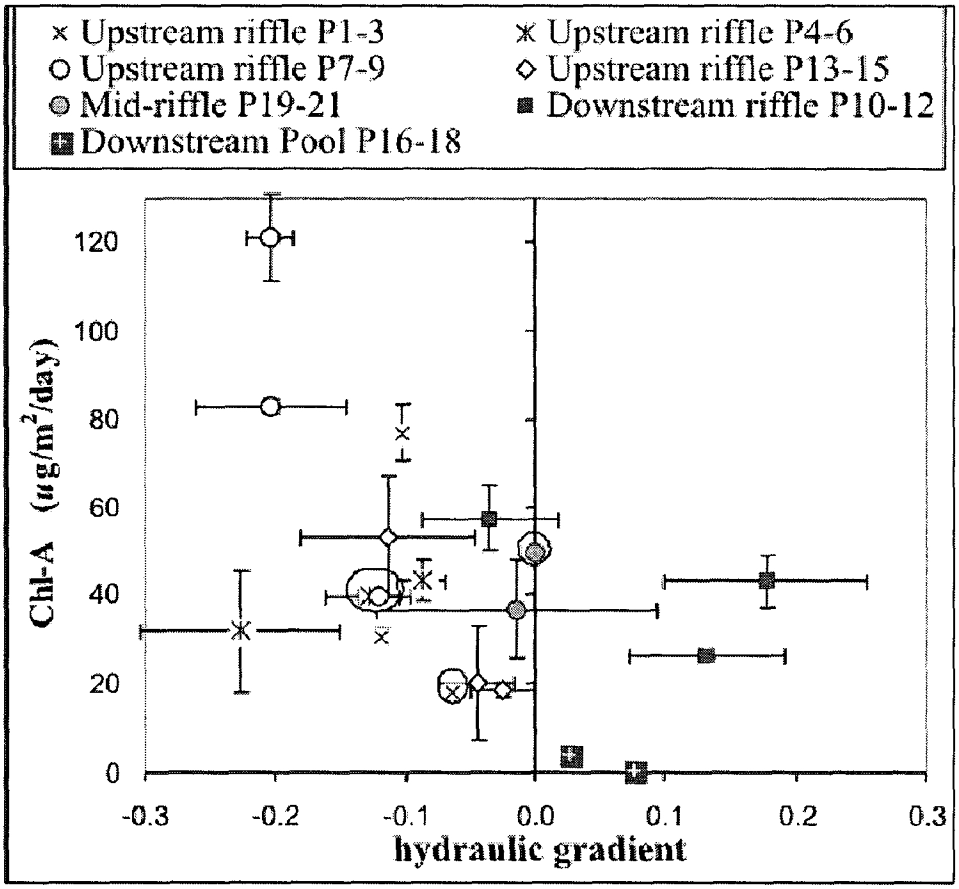


Figure 35. Algal biomass accrual in relation to hydraulic gradients. Algal accrual was an average of n = 2 tiles ± SE, except points circled where n =1. P1-P21 refer to individual piezometres. Hydraulic gradients were averages of 2-3 piezometers per cluster (± SE) except for: P1-3 and P16-18 (n=1).

Though a positive correlation was observed between algal accrual and stream current velocity ($p = 0.012$, data not shown), the relationship was dependent on two data points and fell apart when they were removed ($p = 0.17$). Depth within the stream was not a significant predictor of algal accrual.

Discussion

Algal-nutrient relationships and extreme discharge events:

Although these results indicated that algal biomass was positively related to total phosphorus concentration, the relationship was disrupted during episodes of high river discharge after storm events (Figure 33). Sloughing from high river discharge and sediment scour appear to be a major mechanism for the removal of periphyton (Lohman et al. 1992, Francoeur & Biggs 2006) and should be considered when attempting to determine factors controlling algal biomass. For mid-order channels, care should always be given to sample well after large precipitation events. Calculation of the storm hydrograph for the river would help determine the expected response to precipitation events, although spatial variability in periphyton removal is high for a given set of flows (Hicks et al. 2000) and sediment loads (Peterson 1996).

Hyporheic flow and geochemistry of hyporheic waters:

Typical hyporheic flow during base flow as described in Brunke & Gosner (1997) is illustrated in Figure 34a. Longitudinal differences in water levels in the river cause hyporheic flow in the same direction, with a dominantly horizontal component. The presence of pools and riffle zones typically causes small vertical gradients that result in local recharge and discharge from the hyporheic zone.

Reversal of the vertical hydraulic gradient such as observed in this study (Figure 35) was previously reported for the Raisin River but with no explanation (Porter 1996) and has been discussed in Chapter 4. During heavy precipitation events, river levels typically increase more rapidly than the piezometric levels (Abdul 1989). Although this behaviour is well known for near-stream piezometers (Abdul 1989), it is not as widely reported for in-stream piezometers (but see Arntszen 2006). The interpretation applied here is that an increase in stream level relative to the hyporheic piezometric levels would typically drive channel water into the hyporheic zone (Wondzell 1996). The horizontal component of hyporheic flow would thus remain more or less constant (since governed by longitudinal differences in water levels), but the vertical downward flow would now dominate throughout the reach (Figure 34b), even in areas of local hyporheic discharge during base flow.

The hyporheic response to precipitation is also driven by the groundwater flow system transverse to the river. For the fine-textured soils in the Raisin basin, the extensive capillary fringe and the relatively subdued topography likely contribute to produce relatively rapid piezometric responses and increased shallow groundwater discharge into the river near the banks (Abdul 1989). This interpretation is supported by the work of Bustros-Lussier (2008), which showed that base flow in the Raisin River appears to be predominantly from shallow groundwater.

The geochemistry of the hyporheic waters (piezometer water in Table 12) indicated that there may be mixing of surface and groundwater although there were some inconsistencies. It was anticipated that there would be intermediate concentrations of chloride in piezometers, with higher concentrations in well water and lower concentrations in surface waters as groundwater tends to have higher concentrations of ions such as chloride, due to dissolution of minerals within the aquifer. Data from Macintyre Rapids supported this although there were no significant differences between piezometer water and surface water. Data from St. Andrew's did not support this: piezometer water showed higher Cl^- concentrations than either surface water or groundwater which might be the result of contribution by road salt residues or septic effluent (from homes and the adjacent school). With respect to nitrate, concentrations should theoretically be higher in surface waters and lower in groundwaters with piezometer waters containing intermediate values. While data from St. Andrew's supported this, the data from Macintyre Rapids did not (Table 12). With high discharge events, nitrate concentration in the river water did not increase linearly with discharge, suggesting a greater amount of low-nutrient groundwater was discharged to the river at these times.

Hydraulic gradients and algal accrual

Surface water flow through the sediment-surface water boundary appeared to stimulate algal growth given the observed trend of increased algal accrual with increasing downward flux (Figure 35). The results for upward fluxes are inconclusive because of lack of data. During base flow, this increased flow through the boundary layer (where river current velocity is typically low) may stimulate algal growth by changing the boundary layer thickness (which changes nutrient supply and the removal of metabolic waste). The positive trend found between algal accrual and current velocity supports the idea that flow through the periphyton mat stimulates growth in general whether this flow comes from surface water or groundwater. However, stream velocities were moderate during these measurements; high velocities, such as during high rainfall events, can cause sloughing of benthic algae. During storm flow, the increased river current may have an over-riding deleterious impact on algal growth.

Conclusion

The results presented in this study show that the relationship between periphyton biomass and TP can be linked temporally but disrupted by extreme discharge events. Algal biomass accrual increased linearly with both a positive or negative hydraulic gradient in the

Macintyre Rapids reach. Groundwater may thus influence periphyton dynamics even in lowland nutrient enriched river systems.

Chapter 6: Conclusion

This thesis provided both a spatial study of algal biomass (benthic and suspended) and its relationship with environmental variables at the watershed scale, as well as a temporal study of algal biomass at the reach scale in the Raisin River. Different variables were found to be important at spatial vs. temporal scales in predicting algal biomass, which allowed for a better overall understanding of the factors affecting algal biomass in the watershed. Groundwater/ surface water interactions were also investigated at the reach scale using hydrogeological field methods to determine whether hyporheic flow could explain some of the variability in benthic algal biomass.

Contrary to previous studies in western Quebec and eastern Ontario watersheds where nutrients were the most important variables predicting benthic and suspended algal biomass, light and temperature or stream order were the most important predictors of suspended algal biomass and turbidity and temperature best explained benthic algal biomass. Light was not an important predictor of benthic algal biomass because of shallow river depth and was not likely limiting.

Suspended algal biomass in the Raisin River was less than predicted by a published model based on total phosphorus concentrations possibly due to grazing and an increased rate of sedimentation from contact with the substrate due to the shallow nature of the river. Periphyton biomass was also less than predicted by a published model based on total phosphorus possibly due to losses from grazers, but perhaps also due to losses from sloughing during spates, as found in the temporal study of benthic algae.

Periphyton and suspended algal biomass were negatively correlated. This likely reflects the competition between these two communities for common resources. Overall, periphyton was the dominant algal community, as expected for low-order streams and rivers.

Because we found no correlation of benthic algal biomass with nutrients at the watershed scale, we hypothesized that the effects of nutrients on periphyton biomass might be observed through time at the reach scale and furthermore that groundwater may be influencing algal biomass at this scale. In the hydrogeological field investigation of surface water/ groundwater interactions, it was found that one of the reaches studied showed evidence for shallow hyporheic flow, while results from the other remained inconclusive due to lack of data. Neither reach showed evidence of deep groundwater discharge, which supports previous work on the Raisin River (Bustros-Lussier 2008). Hydrology and especially the flashy nature of the Raisin River (where water levels in a water course respond suddenly to precipitation events) may be the primary driver for surface water/ groundwater interactions, as water may be forced into the substrate when water levels rise quickly and discharge into the river when water levels subsequently drop. This response to water level fluctuations was evident even in areas where downward or upward flow gradients were dominant, creating a temporary reversal in the vertical hydraulic gradient. The response of the shallow hyporheic to water level changes in the river was found to be delayed, likely due to the low hydraulic conductivity of the substrate, which followed the hysteresis model proposed by Arntszen (2006) for larger, regulated rivers.

Contrary to the watershed scale study, the reach-scale temporal study did indeed show a link between total phosphorus and periphyton biomass, although this relationship was temporarily disrupted by extreme discharge events when periphyton would be sloughed off. Benthic algal biomass accrual increased linearly with both a positive or negative hydraulic gradient in the Macintyre Rapids reach, indicating that surface water/ groundwater interactions were important despite the lack of deep groundwater discharge mentioned previously. Groundwater may thus influence periphyton dynamics even in these lowland nutrient enriched river systems.

References

Abdul, A.S. & R.W. Gillham, 1989. Field studies of the effects of the capillary fringe on streamflow generation. Journal of Hydrology 112: 1-18.

Allander, K.K., 2003. Trout Creek- Evaluating ground-water and surface water exchange along an alpine stream, Lake Tahoe, California. In *Heat as a Tool for Studying the Movement of Ground Water Near Streams*, ed. D.A. Stonsestrom and J. Constantz, 35-45. USGS Circular 1260. Reston, Virginia: USGS.

Allen, J.D. & M.M. Castillo, 2007 Stream Ecology, 2nd edn. Springer, The Netherlands. 436 pp.

Anderson, J.K., S. M. Wondzell, M. N. Gooseff & R. Haggerty, 2005. Patterns in stream longitudinal profiles and implications for hyporheic exchange flow at the H.J. Andrews Experimental Forest, Oregon, USA. Hydrological Processes 19: 2931–2949.

Anderson, M.P., 2005. Heat as a groundwater tracer. Ground Water 43: 951-968.

Arntszen, E.V., 2006. Effects of Fluctuating River Flow on Groundwater/Surface water mixing in the hyporheic zone of a regulated, large cobble bed river. River Research and Applications 22: 937-946.

Basu, B.K. & F.R. Pick, 1996. Factors regulating phytoplankton and zooplankton biomass in temperate rivers. Limnology and Oceanography 41: 1572-1577.

Biggs, B.J.F., R.J. Stevenson & R.L. Lowe,1998 A habitat matrix conceptual model for stream periphyton. Archiv für Hydrobiologie 143: 21-56.

Brunke, M. & T. Gosner, 1997. The ecological significance of exchange processes between rivers and groundwater. Freshwater Biology 37: 1-33.

Bukaveckas, P.A. & A.S. Crain, 2002. Inter-annual, seasonal and spatial variability in nutrient limitation of phytoplankton production in a river impoundment. Hydrobiologia 481: 19-31.

Burnison, B.K., 1980. Modified dimethyl sulfoxide (DMSO) extraction for chlorophyll analysis of phytoplankton. Canadian Journal of Fisheries and Aquatic Sciences 37: 729-733.

Bustros-Lussier, E., 2008. Drag-probe electrical conductivity survey of the Castor and Raisin River used to determine deep groundwater discharge. (M. Sc. Thesis, University of Ottawa).

Caraco, N.F., J.J. Cole, D.L. Strayer, 2006. Top-down control from the bottom: Regulation of eutrophication in a large river by benthic grazing. Limnology and Oceanography 51: 664-670.

Carpenter, S.R., N.F. Carado, D.L. Correll, R.W. Howarth, A.N. Sharpley & V.H. Smith, 1998. Nonpoint pollution of surface waters with phosphorus and nitrogen, Ecological Applications 8: 559–568.

Carr, G.M., P.A. Chambers & A. Morin, 2005. Bacteria and algae in stream periphyton along a nutrient gradient. Canadian Journal of Fisheries and Aquatic Science 62: 1309-1319.

Chase, C. I. Elementary statistical procedures. New York: McGraw-Hill, 1976. 277 pp.

Chételat, J., F.R. Pick, A. Morin & P.B. Hamilton, 1999._Periphyton biomass and community composition in rivers of different nutrient status. Canadian Journal of Fisheries and Aquatic Science 56: 560-569.

Coleman, R.L. & C.N. Dahm, 1990. Stream geomorphology: effects on periphyton standing crop and primary production. Journal of the North American Benthological Society 9: 293-302.

Conant, B., 2004. Delineating and quantifying ground water discharge zones using streambed temperatures. Ground Water 42: 243-257.

David, M., D.M. Pepin & F.R. Hauer, 2002. Benthic Responses to Groundwater-Surface Water Exchange in 2 Alluvial Rivers in Northwestern Montana. The Journal of the North American Benthological Society 21: 370-383.

Dingman, S.L., 2002. Physical Hydrology. Second Edition. Prentice Hall, New Jersey. 646 pp.

Dodds, W.K., 2003. The misuse of inorganic N and soluble reactive P to indicated nutrient status of surface waters. Journal of American Benthological Society 22: 171-81.

Dodds, W.K., 2006. Eutrophication and trophic state in rivers and streams. Limnology and Oceanography 51: 671-680.

Dodds, W.K. & E.B. Welch, 2000. Establishing nutrient criteria in streams. Journal of North American Benthological Society 19:186-196.

Edwards R.W. & M. Owens, 1962. The effect of plants on River Conditions IV. The Oxygen Balance of a Chalk Stream. The Journal of Ecology 50: 207-220.

Environment Canada 2007 Water Survey of Canada http://www.on.ec.gc.ca/water/raps/cornwall/intro_e.html, accessed November 3, 2007

Environment Canada 2008 Climate centre http://climate.weatheroffice.ec.gc.ca/climate_normals/results_e.html?Province=ALL&StationName=cornwall&SearchType=BeginsWith&LocateBy=Province&Proximity=25&ProximityFrom=City&StationNumber=&IDType=MSC&CityName=&ParkName=&LatitudeDegrees=&LatitudeMinutes=&LongitudeDegrees=&LongitudeMinutes=&NormalsClass=A&SelNormals=&StnId=4255&&autofwd=0&START=1&END=13&VIEWBUTTON=View&b_templatePrint=true, accessed on January 5, 2008

EPA online 2004 http://www.epa.gov/glnpo/lakeont/2004update/LO2004.pdf accessed on January 5, 2008

Evans, C. & T.D. Davies, 1998. Causes of concentration/discharge hysteresis and its potential as a tool for analysis of episode hydrochemistry. Water Resources Research 34: 129-137.

Figueroa-Nieves, D., T.V. Royer & M.B. David, 2006. Controls on chlorophyll-a in nutrient-rich agricultural streams in Illinois, USA Hydrobiologia 568: 287–298.

Flecker, A.S., B.W. Taylor, E.S. Bernhardt, J.M. Hood, W.K. Cornwell, S.R. Cassatt, M.J. Vanni, & N.S. Altman, 2002. Interactions between herbivorous fishes and limiting nutrients in a tropical stream ecosystem. Ecology 83: 1831-1844.

Forsythe, W.C., E.J.Jr. Rykiel, R.S. Stahl, H. Wu, & R.M. Schoolfield, 1995. A model comparison for daylength as a function of latitude and day of year. Ecological Modelling 80: 87-95.

Francoeur, S.N., 2001. Meta-analysis of lotic nutrient amendment experiments: detecting and quantifying subtile response. Journal of North American Benthological Society 20: 358-368 .

Francoeur, S.N. & B.J.F. Biggs, 2006. Short-term effects of elevated velocity and sediment abrasion on benthic algal communities. Hydrobiologia 561: 59–69.

Freeze, R.A. & J.A Cherry, 1979. Groundwater. Englewood Cliffs, N.J.: Prentice-Hall. 604 pp.

Golterman, H.L., 1975. Chemistry. In: Whitton BA (ed) River ecology. University of Calofornia Press, Berkeley, CA, pp 39-80.

Gooseff, M.N., J.K. Anderson, S.M. Wondzell, J. Lanier & R. Haggerty, 2005. A modelling study of hyporheic exchange pattern and the sequence, size, and spacing of stream bedforms in mountain stream networks, Oregon, USA. Hydrological Processes 19: 2915-2929.

Hagerthey S.E., W.C. Kerfoot, 1998. Groundwater Flow Influences the Biomass and Nutrient Ratios of Epibenthic Algae in a North Temperate Seepage Lake. Limnology and Oceanography 43: 1227-1242.

Hannah D.M., P.J. Wood & J.P. Sadler, 2004. Ecohydrology and hydroecology: A 'new paradigm'? Hydrologicial Processes 18: 3439-3445.

Hicks, D.M., B. Gomez & A. Trustrum, 2000. Erosion thresholds and suspended sediment yields, Waipaoa River basin, New Zealand. Water Resources Research 36: 1129–1142.

Hill, A.R., 1978. Factors affecting the export of nitrate-nitrogen from drainage basins in Southern Ontario. Water Research 12: 1045-1057.

Hillebrand H., 2002. Top-down vs. bottom-up control of autotrophic biomass – a meta-analysis on experiments with periphyton. Journal of the North American Benthological Society 21: 349-369.

Holsworth, J., 2005. Solar Radiation Attenuation in the Raisin River. (Undergraduate Environmental Science Honours Thesis, University of Ottawa). 79 pp.

Jeffrey, S.W. & G.F. Humphrey, 1975. New spectrophotometric equations for determining chlorophylls a, b, c1 and c2 in higher plants, algae, and natural phytoplankton. Biochemistry and Physiology Planzen., 167: 191-194.

Johnson, S.L., 2004. Factors influencing stream temperatures in small streams: substrate effects and a shading experiment. Canadian Journal of Fisheries and Aquatic Science 61: 913–923.

Jones, R.I. & R.J. Barrington, 1985. A study of the suspended algae in the River Derwent, Derbyshire, U.K. Hydrobiologia 128: 255-264.

Klausmeier, C.A., E. Litchman, T. Dausfresne, & S.A. Levin, 2004. Optimal nitrogen-to-phosphorus stoichiometry of phytoplankton. Nature: 429:171-174.

Köhler, J., M. Bahnwart & K. Ockenfeld, 2002. Growth and Loss Processes of Riverine Phytoplankton in Relation to Water Depth. International Review of Hydrobiology 87: 241-254.

Kramer, D., 1987. Dissolved oxygen and fish behaviour. Enviornmental Biology of Fishes 18: 81-92.

Kutner, M.H., C.J. Nachtsheim, J. Neter & W. Li, 2005. Applied Linear Statistical Models Fifth Edition. McGraw-Hill Irwin, Boston. 1396 pp.

Lee, D.R., 1972. Septic tank nutrients in groundwater entering Lake Sallie, Minnesota. M.S. Thesis, North Dakota University, Grand Forks, 96 pp.

Lee, D.R. & J.A. Cherry, 1979. A field exercise on groundwater flow using seepage meters and mini-piezometers, Journal of Geological Education 27: 6-10.

Lohman, K., J.R. Jones & B.D. Perkins, 1992. Effects of nutrient enrichment and flood frequency on periphyton biomass in northern Ozark streams. Canadian Journal of Fisheries and Aquatic Science 49: 1198-1205.

Lowe, R.L., S.W. Golliday & J.R. Webster, 1986. Periphyton response to nutrient manipulation instreams draining clearcut and forested watersheds. Journal of the Northa American Benthological Society 5: 221-229.

Maharaj, S.P.A., 2007. Factors Contributing to Elevated Levels of Methyl and Total Mercury Related to alteration in Wetland Conditions and Agricultural Activity (The Raisin River Watershed). M.S. thesis, University of Ottawa.

Mellina, E., R.D. Moore, S.G. Hinch, J. Stevenson Macdonald & G. Pearson, 2002. Stream temperature responses to clearcut logging in British Columbia: the moderating influences of groundwater and headwater lakes. Canadian Journal of Fisheries and Aquatic Sciences 59: 1886-1900.

Mosisch, T.D., S.E. Bunn, P.M. Davis, S.J. Marshal, 1999. Effects of shade and nutrient manipulation on periphyton growth in a subtropical stream. Aquatic Botany 64: 167-177.

Peterson, C.G., 1996. Response of benthic algal communities to natural physical disturbance. In Stevenson, R. J., M. L. Bothwell, & R. L. Lowe [ed.], Algal Ecology: Freshwater Benthic Ecosystems. Academic. p. 375-402.

Pichard, V., F.R. Pick & A.L. Martel (2009) Diversity, distribution and abundance of freshwater mussels in the Raisin River drainage basin, Eastern Ontario, Canada. Verh. Internat. Verein. Limnol.

Porter, S., 1996. Groundwater/surface water interaction in the Raisin River Watershed, near Cornwall, Ontario. M.S. thesis, University of Ottawa.

Pringle, C.M. & J.A. Bowers, 1984. An in situ substratum fertilization technique: diatom colonization on nutrient-enriched, sand substrata. Canadian Journal of Fisheries and Aquatic Sciences 41: 1247-1251.

Quinn, J.M., Cooper, A.B., Davies-Colley, R.J., Rutherford, J.C., Williamson, R.B., 1997. Land use effects on habitat, water quality, periphyton and benthic invertebraes in Waikato, New Zealand, hill-country streams. New Zealand Journal of Marine and Freshwater Research 31: 579-597.

Raisin - South Nation Source Water Protection Region (SWPR). Water Budget: Conceptual Understanding (Draft). February, 2007.

Redfield, A.C., 1958. The biological control on chemical factors in the environment. American Scientist 46: 205-221.

Reynolds, C.S., 1988. Round, F.E. (ed.), Algae and the Aquatic Environment. Biopress, Bristol, pp. 285-311.

Rose, S., 2003. Comparative solute-discharge hysteresis analysis for an urbanized and a 'control basin' in the Georgia (USA) Piedmont. Journal of Hydrology 284: 45-56.

Round, F.E., 1988. Algae and the Aquatic Environment. Biopress. 460 pp.

Rutherford, J.C., S. Blackett, C. Blackett, L. Saito & R.J. Davies-Colley, 1997. Predicting the effects of shade on water temperature in small streams. New Zealand Journal of Marin and Freshwater Research 31: 707-721.

Schälchli, U., 1992. The clogging of coarse gravel river beds by fine sediment. Hydrobiologia 235/236: 189-197.

Schmidt, C., B. Conant Jr., M. Bayer-Raich, & M. Schirmer, 2007. Evaluation and field-scale application of an analytical method to quantify groundwater discharge using mapped streambed temperatures, Journal of Hydrology 347: 292-307.

Singer, S.N., C.K. Cheng & M.G. Scafe, 2003. The Hydrogeology of Southern Ontario Second Edition. *Environmental Monitoring and Reporting Branch, Ministry of the Environment.* Toronto, ON. 395 pp.

Slavik K., B.J. Peterson, L.A. Deegan, W.B. Bowden, A.E. Hershey & J.E. Hobgie, 2004. Long-term reponses of the Kuparuk River ecosystem to phosphorus fertilization. Ecology 85: 939-954.

Sophocleous. M., 2002. Interactions between groundwater and surface water: the state of the science. Hydrogeology Journal 10: 52-67.

Squillace, P.J., 1996. Observed and simulated movement of bank-storage water. Ground Water 34: 121-134.

Stanford, J.A. & J.V. Ward, 1993. An ecosystem perspective of alluvial rivers: connectivity and the hyporheic corridor. Journal of the North American Benthological Society 12:48-60.

Statistics Canada Census for Cornwall, 2001. http://www12.statcan.ca/english/profil01/CP01/details/Page.cfm?Lang=E&Geo1=CSD&Code1=3501012&Geo2=PR&Code2=35&Data=Count&SearchText=CORNWALL&SearchType=Begins&SearchPR=01&B1=All&Custom=, accessed on January 5, 2008

Steinman, A.D. & G.A. Lamberti, 1996. Biomass and pigments of benthic algae. In Hauer FR & Lamberti GA (Eds), Methods in Stream Ecology. Academic Press, San Diego, California, 295-313 pp.

Stockner, J.F. & K.R.S. Shortreed, 1978. Enhancement of autotrophic production by nutrient addition in a coastal rainforest stream on Vancouver Island. Canadian Journal of Fisheries and Aquatic Science 35: 28-34.

Storey, R.G., K.W.F. Howard, & D.D. Williams, 2003. Factors controlling riffle-scale hyporheic exchange flows and their seasonal changes in a gaining stream: A three-dimensional groundwater flow model. Water Resources Research 39: 1034-1051.

Suchy, M., (in progress). Seasonality and Fate of Agricultural Groundwater in Eastern Ontario: An Aqueous and Isotope Geochemistry Approach. M.Sc Thesis, University of Ottawa, Canada.

Sullivan, B.E., F.G. Prahl, L.F. Small & A. Covert, 2001. Seasonality of phytoplankton production in the Columbia River: a natural or anthropogenic pattern? Geochimica et Cosmochimica Acta 65: 1125-1139.

Szilagyi, J., & M.B. Parlange, 1998. Baseflow separation based on analytical solutions of the Boussinesq equation. Journal of Hydrology 204: 251-260.

Terasmae, J. & R.J. Mott, 1965. Map 1175A, Surficial Geology, Cornwall, Ontario-Quebec. Geological Survey of Canada.

Thomas, M.P., 1966. Effect of glacial geology upon the time distribution of streamflow in eastern and southern Connecticut. *US Geological Survey Professional Paper 550-B*: 209-212.

Triska F.J., V.C. Kennedy & R.J. Avanzino, 1989. Retention and transport of nutrients in a third-order stream in northwestern California: hyporheic process. Ecology 70: 1893–1905.

Turcotte, D.L. & G. Schubert, 1982. Geodynamics. John Wiley and Sons, New York: 450 pp.

Van Nieuwenhuyse, E.E. & J.R. Jones, 1996. Phosphorus–chlorophyll relationship in temperate streams and its variation with stream catchment area. Canadian Journal of Fisheries and Aquatic Science 53: 99-105.

Vannote, R.L., G.W. Minshall, K.W. Cummins, J.R. Sedell & C.E. Cushing, 1980. The river continuum concept. Canadiaen Journal of Fisheries and Aquatic Sciences 37: 130-137.

Walling, D.E. & B.W. Webb, 1992. Water Quality. I. Physical charictaristics. In P. Calow and G.E. Petts, eds. The rivers Handbook. I. Hydrological and ecological principles. Blackwell Scientific Pubications, Oxford: 48-72.

Ward, J.V. & K. Tockner, 2001. Biodiversity: towards a unifying theme of river ecology. Freshwater Biology 46:807-819.

Water Survey of Canada, 2008.
http://www.wsc.ec.gc.ca/products/main_e.cfm?cname=products_e.cfm, Accessed January 15, 2008.

Wetzel, R.G., 2001. Limnology. Third Ed. Elsevere, San Diego.1006 pp.

Wetzel, R.G. & G.E. Likens, 1979. Limnological Analyses. W.B. Saunders Co., Philadelphia. 357 pp.

White, D.S., C.H. Elzinga & S.P. Hendricks, 1987. Temperature Patterns within the Hyporheic Zone of a Northern Michigan River. Journal of the North American Benthological Society 6: 85-91.

Whiting, P.J., 2002. Streamflow Necessary for Environmental Maintenance. Annual Review Of Earth And Planetary Sciences 30: 181-206.

Wintermans, J.F.G. & A. DeMots, 1965. Spectrophotometric characteristics of chlorophylls a and b and the phaeophytins in ethanol. Biochim. Biophys. Acta. 109: 448-453.

Wolfe, J.E. & O.T. Lind, 2008. Influence of suspended clay on phosphorus uptake by periphyton. Hydrobiologia 610: 1-12.

Woods, L.M.E., F.R. Pick & M.J.L. Robin, 2009. Evaluation of the influence of groundwater on algal biomass and accrual at the reach scale in a shallow river (Raisin River, South Eastern Ontario, Canada).Verh. Internat. Verein. Limnol. 30: 662-668.

Wright, K.K., C.V. Baxter & J.L. Li, 2005. Restricted hyporheic exchange in an alluvial river system: Implications for theory and management Journal of the North American Benthological Society 24: 447-460.

Zalewski, M., G.A. Janauer & G. Jolankaj, 1997. Ecohydrology: a new paradigm for the sustainable use of aquatic resources. UNESCO IHP Technical Documents in Hydrology no. 7, IHP-V Projects 2·32·4, UNESCO, Paris, France.

Appendices

Appendix A: Cornwall Climate data

Table A1. Data obtained from Environment Canada's online climate record database for the Cornwall Ontario climate station.

CORNWALL *
ONTARIO

Latitude: 45° 1.200' N | Longitude: 74° 45.000' W | Elevation: 64.00 m

Climate ID: 6101874 | WMO ID: | TC ID:

* This station meets WMO standards **for temperature and precipitation.**

Created : 2002-06-21

Modified : 2004-02-25

Reviewed : 2004-02-25

Environment Canada | Environnement Canada

Canadian Climate Normals 1971-2000

Url of this page : http://www.climate.weatheroffice.ec.gc.ca/climate_normals/results_e.html

Temperatur	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec	Year	Code
Daily Average (°C)	-8.8	-7.3	-1.3	6.5	13.9	18.7	21.6	20.4	15.5	9.1	2.6	-4.9	7.2	A
Standard Deviation	3	2.9	2.4	1.8	1.7	1.2	1	1.1	1.2	1.6	1.6	3.2	0.9	A
Daily Maximum (°C	-4.6	-3.1	3	11.3	19.3	23.9	26.7	25.3	20.2	13.2	5.9	-1.3	11.7	A
Daily Minimum (°C	-12.9	-11.4	-5.5	1.7	8.5	13.4	16.4	15.4	10.8	4.9	-0.8	-8.5	2.7	A
Extreme Maximum	18	18	26	30.5	33	35	35.6	36.5	34.5	28.5	23.9	20		
Date (yyyy/dd)	1996/19	2000/27	1998/31	1990/27	1999/31	1964/30	1955/22+	2001/08	1999/04	1991/05	1961/03+	2001/05		
Extreme Minimum	-43.3	-35.6	-25	-15	-5.6	0.6	3.3	1.7	-6.1	-8.3	-20	-33.3		
Date (yyyy/dd)	1957/15	1951/10	1989/07	1954/03+	1952/05+	1953/04	1952/30	1957/22+	1951/30	1956/26	1951/06	1951/28+		
Precipitation:														
Rainfall (mm)	27.3	20	34.3	69.6	82.8	88	92.6	93.2	102.4	81	69.5	34.1	794.8	A
Snowfall (cm)	54.5	43	33.1	11.3	0	0	0	0	0	1.4	17	46.7	207.1	A
Precipitation (mm)	81.7	63	67.5	81	82.9	88	92.6	93.2	102.4	82.4	86.5	80.8	1002	A

Extreme Daily Rain	37.5	37.6	39.1	33.8	46.8	58.9	73	70.9	**110.4**	60	46.4	35		
Date (yyyy/dd)	1978/08	1961/25	1973/17	1990/03	2000/08	1969/23	1977/17	1962/20	**1979/14**	1995/05	2000/26	1982/15		
Extreme Daily Snow	**53.3**	33.8	38.1	35.6	8.9	0	0	0	0	14.4	36.1	35.5		
Date (yyyy/dd)	**1966/30**	1972/03	1971/04	1975/03	1963/10	1951/01+	1951/01+	1951/01+	1951/01+	1988/22	1975/14	1993/21		
Extreme Daily Preci	53.3	37.6	42.4	41.7	46.8	58.9	73	70.9	**110.4**	60	46.4	35.5		
Date (yyyy/dd)	1966/30	1961/25	1973/17	2000/08	2000/08	1969/23	1977/17	1962/20	**1979/14**	1995/05	2000/26	1993/21		
Extreme Snow Dept	48	**56**	34	30	0	0	0	0	0	0	19	40		
Date (yyyy/dd)	1999/16	**2000/16+**	1998/24	2001/01	1983/01+	1983/01+	1983/01+	1983/01+	1983/01+	1983/01+	1997/17+	1995/21+		

Days with Maximum Temperature:														
<= 0 °C	21.9	19.6	10.2	0.83	0	0	0	0	0	0	4.8	17.8	75.1	A
> 0 °C	9.1	8.7	20.8	29.2	31	30	31	31	30	31	25.2	13.2	290.2	A
> 10 °C	0.48	0.43	4.3	15.9	29.5	30	31	31	29.5	20.9	6.1	1.2	200.5	A
> 20 °C	0	0	0.38	2.7	13.2	23.9	29.9	28.2	14.2	3.2	0.27	0	115.9	A
> 30 °C	0	0	0	0.03	0.47	2.1	4	2.4	0.43	0	0	0	9.4	A
> 35 °C	0	0	0	0	0	0	0	0.03	0	0	0	0	0.03	A

Days with Minimum Temperature:														
> 0 °C	1.5	1.4	5.9	18.2	30.5	30	31	31	29.7	25.5	13.1	2.8	220.6	A
<= 2 °C	30.6	27.8	28.7	17	1.9	0.1	0	0	0.83	9.8	22.4	29.9	169.1	A
<= 0 °C	29.6	26.9	25.1	11.8	0.47	0	0	0	0.33	5.5	16.9	28.2	144.7	A
< -2 °C	27.9	24.7	19.9	5.8	0.03	0	0	0	0	1.5	10.6	24	114.3	A
< -10 °C	18.8	16.6	7.2	0.33	0	0	0	0	0	0	1.1	11.9	55.9	A
< -20 °C	5.5	3.5	0.34	0	0	0	0	0	0	0	0	1.8	11.2	A
< - 30 °C	0.14	0.07	0	0	0	0	0	0	0	0	0	0	0.21	A

Days with Rainfall:														
>= 0.2 mm	3.6	3.5	6	10.6	12.9	11.9	10.9	11.2	12.2	11.8	11.3	5.1	111	A
>= 5 mm	1.6	1.6	2.3	5	5.4	5.6	5.7	5.4	6	5	4.4	2.3	50.6	A
>= 10 mm	0.87	0.6	1.1	2.5	2.8	3.2	3.1	3.3	3.6	2.6	2.4	1.2	27.4	A
>= 25 mm	0.3	0.07	0.1	0.27	0.33	0.53	0.73	0.83	0.67	0.43	0.4	0.17	4.8	A

Days With Snowfall:														
>= 0.2 cm	12	9.1	6.7	2	0.03	0	0	0	0	0.4	4	10.4	44.5	A
>= 5 cm	4	2.8	2.4	0.63	0	0	0	0	0	0.1	1.3	3.1	14.3	A
>= 10 cm	1.4	1.4	0.77	0.37	0	0	0	0	0	0.03	0.63	1.4	6	A
>= 25 cm	0.07	0.1	0.1	0.07	0	0	0	0	0	0	0.03	0.13	0.5	A

Days with Precipitation:														
>= 0.2 mm	14.6	11.5	11.6	11.8	12.9	11.9	10.9	11.2	12.2	12.1	14.2	14.4	149.5	A
>= 5 mm	5.8	4.5	4.7	5.7	5.4	5.6	5.7	5.4	6	5	5.7	5.5	65.1	A
>= 10 mm	2.2	2.1	1.8	2.9	2.9	3.2	3.1	3.3	3.6	2.6	3	2.7	33.6	A
>= 25 mm	0.37	0.17	0.27	0.4	0.33	0.53	0.73	0.83	0.67	0.43	0.5	0.33	5.6	A
Degree Days:														
Above 24 °C	0	0	0	0	0.2	3.3	9.5	5.8	0.9	0	0	0	19.8	A
Above 18 °C	0	0	0	1.3	14.8	57.3	115.3	87.7	22.9	1.5	0	0	300.7	A
Above 15 °C	0	0	0.2	4.8	41.4	121.7	203.3	168.3	59.1	7.5	0.3	0	606.5	A
Above 10 °C	0	0.1	2.3	23.6	134.4	261.2	357.8	321.3	170.3	45.9	5.4	0.1	1322.3	A
Above 5 °C	1	1.1	15.1	84.6	276.8	410.7	512.8	476.3	315.5	139.6	31.5	2.9	2267.8	A
Above 0 °C	10.4	11.1	58.9	201.6	431	560.7	667.8	631.3	465.5	282.4	107.4	22.7	3450.7	A
Below 0 °C	275.5	224.5	96.4	6.8	0	0	0	0	0	0.2	30.2	174.8	808.4	A
Below 5 °C	421.2	355.9	207.6	39.7	0.8	0	0	0	0.1	12.4	104.2	310.1	1451.9	A
Below 10 °C	575.2	496.3	349.9	128.8	13.4	0.5	0	0	4.8	73.7	228.1	462.3	2332.8	A
Below 15 °C	730.2	637.6	502.7	259.9	75.4	11	0.5	2	43.6	190.3	373	617.1	3443.5	A
Below 18 °C	823.2	722.5	595.5	346.5	141.9	36.6	5.5	14.4	97.5	277.3	462.7	710.1	4233.5	A

Appendix B – Discharge diversion data from Long Sault

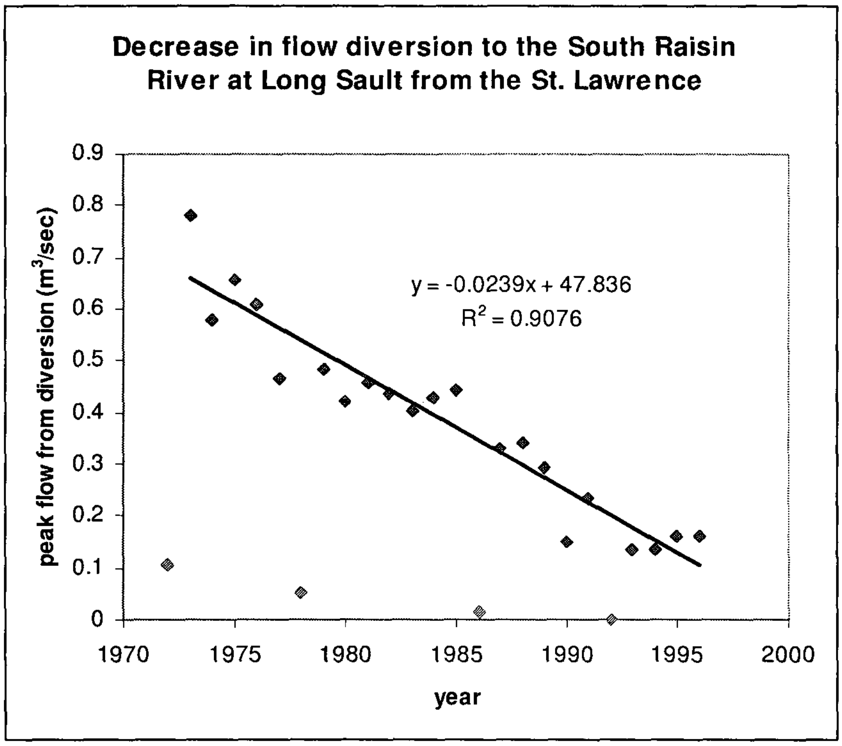


Figure B1. Decrease in flow diversion to the South Raisin River at Long Sault from the St. Lawrence River.

Appendix C – Hydrology Methods

The method used to determine the base-flow portion of the hydrograph was that of Dunne and Leopold, 1978 where a line was drawn from the initial rise of the hydrograph at a slope of 0.05 ft^3/sec*A (mi^2) until it intercepted the hydrograph. Though this method was only intended for watersheds less than 20 mi^2, it was found that this method gave the most realistic result while remaining the most objective. It was assumed that the contributing area in the watershed was equal to the entire area of the watershed since we were not able to obtain data to determine the actual contributing area.

The North and Middle branches of the Raisin contain the contributing area, A_c that has been estimated at 405.6 km^2 (Figure 9), data provided by the Raisin River Conservation Authority. Four constructed hydrographs were analysed using discharge estimates from the Williamstown gauge station (raw data obtained from Environment Canada).

Note that the axis scale varies by graph.

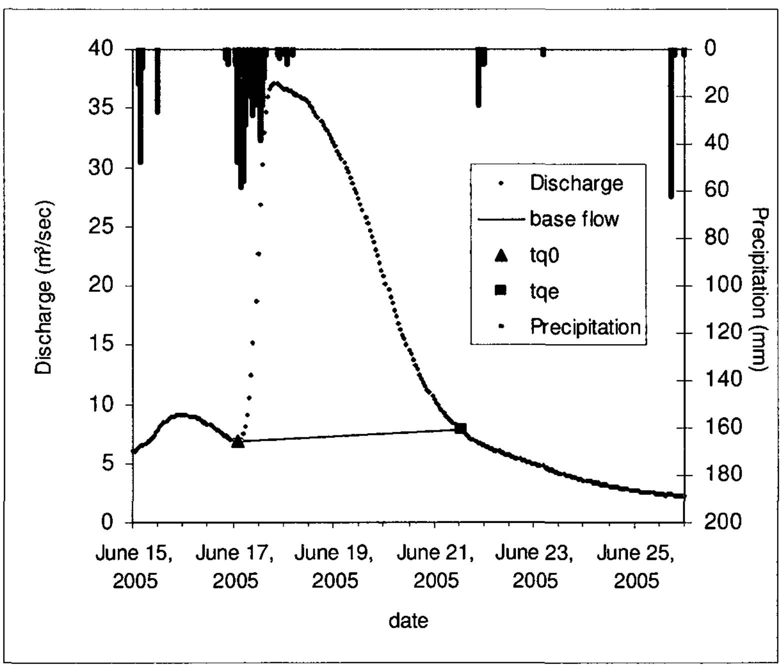


Figure C1. June 16th, 2005 storm hydrograph

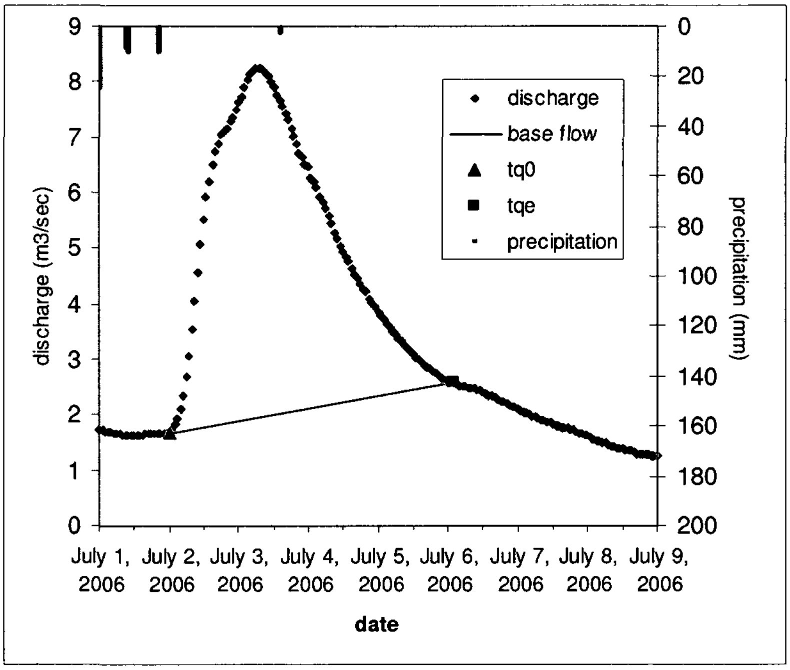


Figure C2. July 2nd, 2006 storm hydrograph

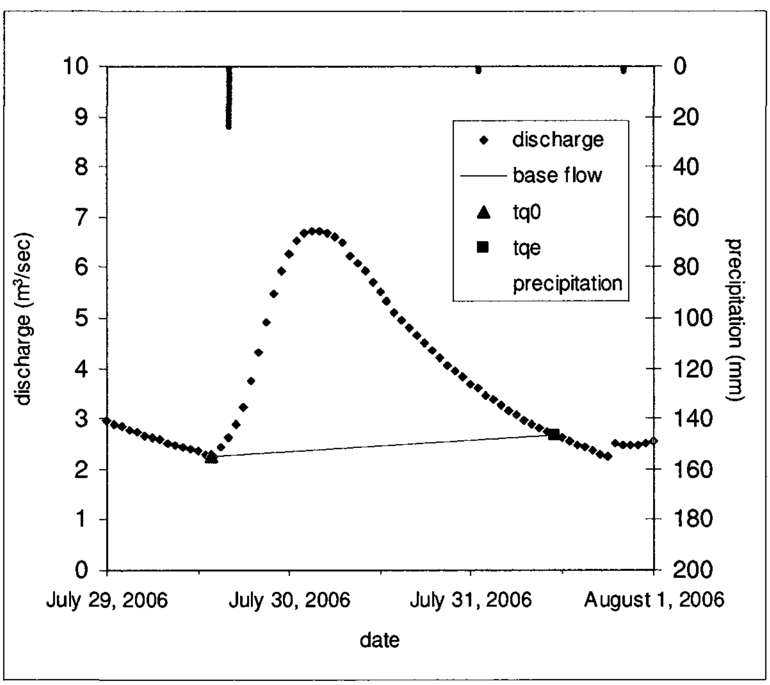


Figure C3. July 29th, 2006 storm hydrograph

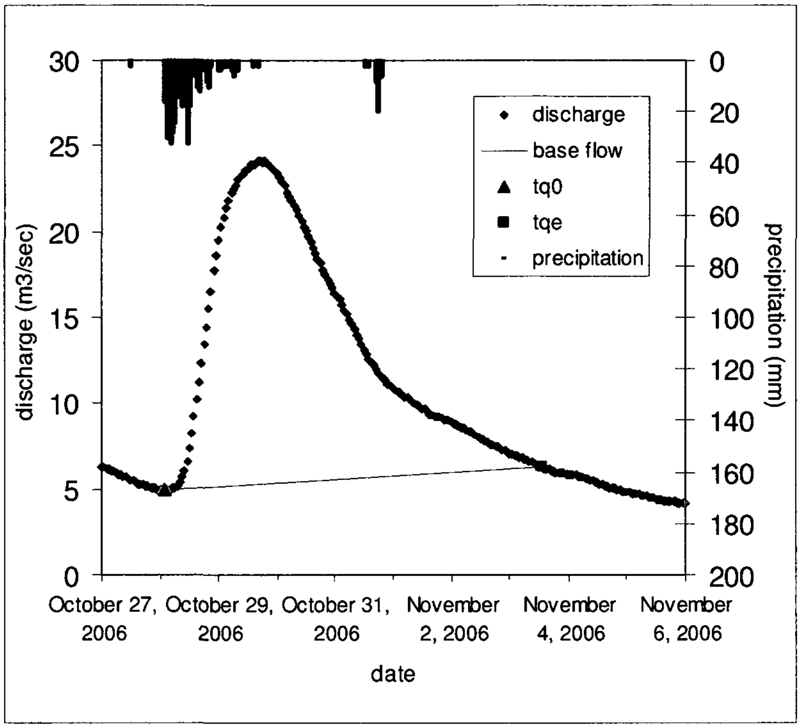


Figure C4. October 27th, 2006 storm hydrograph

Appendix D- Watershed-scale algal-environmental interactions

Table D1. List of abbreviations for variables after transformations were applied to ensure normal distributions.

STRAHLER	Strahler Stream order [ ]
DISTOMTHKM	Distance to mouth of Raisin River [km]
MATSITEDSCH	River discharge at each site [m^3/s], transformed using McCall's area transformation
MATWILLDAYDI	River discharge at the "near Williamstown" Env. Canada gauge [m^3/s], transformed using McCall's area transformation
PERIVEL	Average velocity at each site measured above all periphyton samples taken [m/s]
MATPOOLDEPTH	Depth of the pool section near the riffle site [m], transformed using the McCall's area transformation
TEMP	Temperature [ºC]
DOC	Dissolved organic carbon (calculated from absorbance units measured at 440nm)
SQRTCOLOUR	Colour (absorbance units measured at 440nm), transformed by square root
EXTCOEFF	Extinction Coefficient
TURBIDITY	Turbidity [nephelometric turbidity units]
SPC	Specific Conductivity [µS/cm]
LOGNO3	Nitrate [µg/L], log-transformed
LOGNH4	Ammonium [µg/L], log-transformed
TP	Total Phosphorous [µg/L]
SRP	Soluble reactive phosphate [µg/L]
LOGPERI	Periphyton biomass (measured as chlorophyll-a in mg/m^2), log-transformed
LOGSUSPUGL	Suspended algal biomass (measured as chlorophyll-a in µg/L), log-transformed
LOGSUSPMGM2	Suspended algal biomass (measured as chlorophyll-a in mg/m^2), log-transformed

Table D2. Descriptive statistics for all variables fitted to a normal distribution. Shapiro-Wilk (SW) P-Value indicates goodness of fit to normal distribution where $p < 0.01$ indicates values are not normally distributed.

	N of cases	Min.	Max.	Median	Mean	95% CI Upper	95% CI Lower	Std. Error	Std. Dev	C.V.	SW P-value
STRAHLER	40	3	6	4.5	4.6	4.8	4.3	0.1	0.8	0.18	0.00
DISTOMTHKM	40	6.92	66.27	33.22	33.42	38.88	27.96	2.70	17.08	0.51	0.19
MATSITEDSCH	38	17.59	62.41	40.00	40.21	43.48	36.94	1.61	9.94	0.25	1.00
MATWILLDAYDI	40	26.41	60.03	43.50	42.34	45.42	39.25	1.53	9.65	0.23	0.23
PERIVEL	40	0.00	0.61	0.27	0.26	0.31	0.22	0.02	0.14	0.52	0.85
POOLDEPTH	40	0.10	0.50	0.30	0.30	0.33	0.26	0.02	0.10	0.34	0.07
TEMP	39	17.99	29.71	23.92	24.18	24.98	23.39	0.39	2.46	0.10	0.52
DOC	40	1.5	40.6	18.8	17.8	21.2	14.4	1.7	10.6	0.60	0.03
SQRTCOLOUR	40	0.12	0.84	0.37	0.38	0.44	0.32	0.03	0.18	0.48	0.04
EXTCOEFF	39	0.67	7.94	3.36	3.14	3.69	2.59	0.27	1.70	0.54	0.04
TURBIDITY	40	0.9	14.5	6.3	6.4	7.5	5.3	0.5	3.4	0.54	0.17
SPC	39	265.1	766.7	477.1	490.0	524.5	455.5	17.0	106.4	0.22	0.81
LOGNO3	35	0.70	2.48	1.78	1.59	1.79	1.40	0.10	0.57	0.36	0.01
LOGNH4	27	0.70	1.85	1.30	1.35	1.44	1.25	0.05	0.24	0.17	0.01
TP	40	10	130	50	54	59	48	3	18	0.33	0.00
SRP	35	5	50	30	28	33	24	2	13	0.46	0.01
LOGPERI	40	1.15	2.23	1.69	1.70	1.77	1.63	0.04	0.23	0.13	0.89
LOGSUSPUGL	40	0.28	1.11	0.69	0.69	0.76	0.63	0.03	0.20	0.29	0.82
LOGSUSPMGM2	40	0.0012	0.0197	0.0053	0.0069	0.0082	0.0056	0.0006	0.0041	0.59	0.00

Table D3. Pair-wise Frequency Matrix showing n observations used in the corresponding Pearson Correlation Matrix of independent variables.

	MATDAYLIGHTH	DISTOMTHKM	RT4SITEDSCH	MATWILLDAYDI	PERIVEL	MATPOOLDEPTH	SQRTPERIDEPT	TEMP	INVDO	SQRTDOC	SQRTCOLOUR	EXTCOEFF	TURBIDITY	SPC	LOGNO3NOND	LOGNH4	TPNOOUT	SRP	LOGPERI	LOGSUSPUGL	LOGSUSPUGM2
MATDAYLIGHTH	50																				
DISTOMTHKM	50	50																			
RT4SITEDSCH	41	41	41																		
MATWILLDAYDI	50	50	41	50																	
PERIVEL	43	43	41	43	43																
MATPOOLDEPTH	50	50	41	50	43	50															
SQRTPERIDEPT	43	43	41	43	43	43	43														
TEMP	42	42	40	42	42	42	42	42													
INVDO	42	42	40	42	42	42	42	42	42												
SQRTDOC	50	50	41	50	43	50	43	42	42	50											
SQRTCOLOUR	50	50	41	50	43	50	43	42	42	50	50										
EXTCOEFF	49	49	40	49	42	49	42	41	41	49	49	49									
TURBIDITY	50	50	41	50	43	50	43	42	42	50	50	49	50								
SPC	42	42	40	42	42	42	42	42	42	42	42	41	42	42							
LOGNO3NOND	34	34	30	34	31	34	31	31	31	34	34	33	34	31	34						
LOGNH4	36	36	27	36	29	36	29	28	28	36	36	35	36	28	28	36					
TPNOOUT	41	41	39	41	41	41	41	40	40	41	41	40	41	40	29	27	41				
SRP	45	45	36	45	38	45	38	37	37	45	45	44	45	37	34	36	36	45			
LOGPERI	43	43	41	43	43	43	43	42	42	43	43	42	43	42	31	29	41	38	43		
LOGSUSPUGL	49	49	40	49	42	49	42	41	41	49	49	48	49	41	34	36	40	44	42	49	
LOGSUSPUGM2	49	49	40	49	42	49	42	41	41	49	49	48	49	41	34	36	40	44	42	49	49

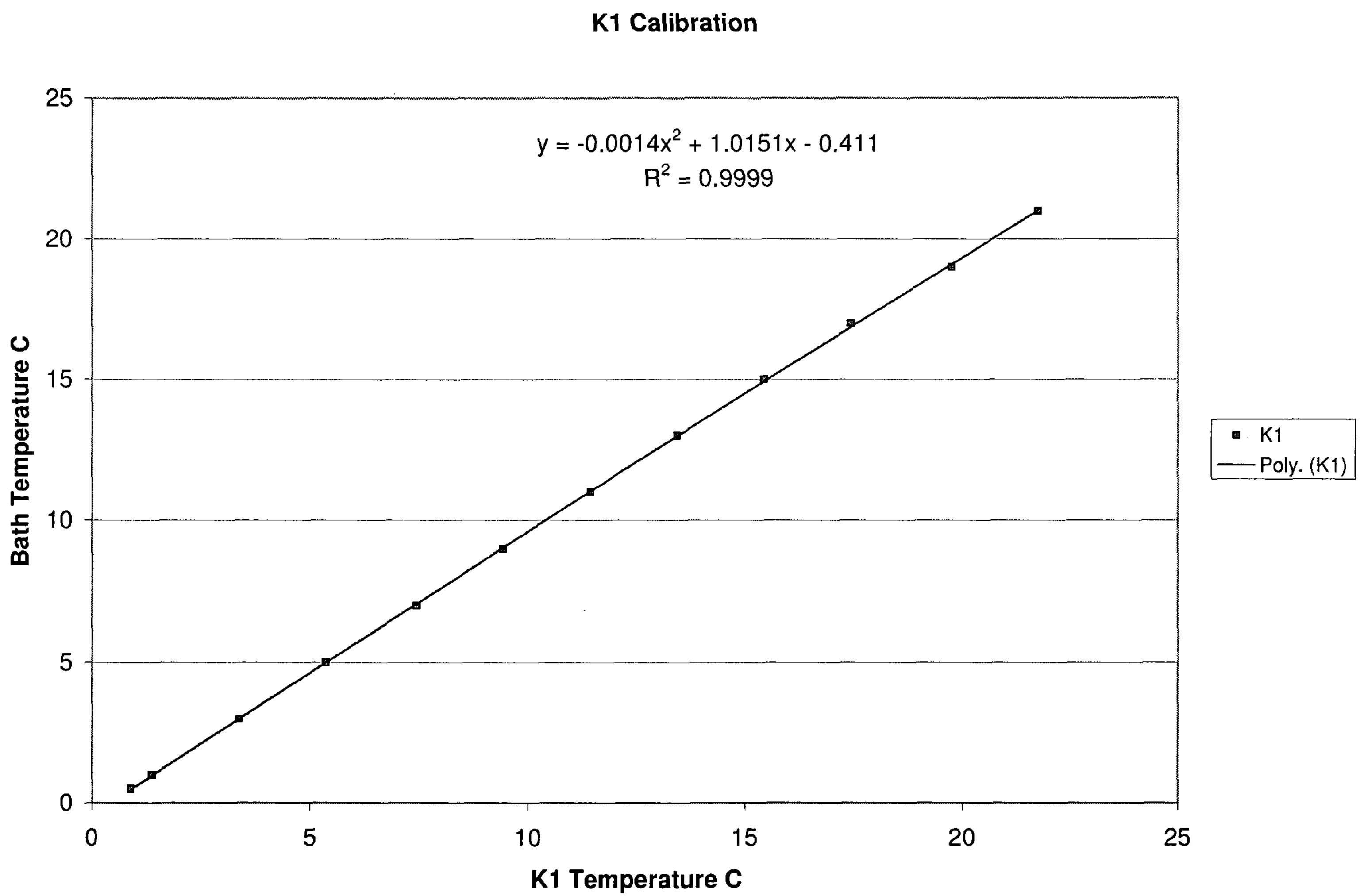
Table D4. Simple regressions of independent variables predicting log suspended algal biomass as Chl a (µg/L).

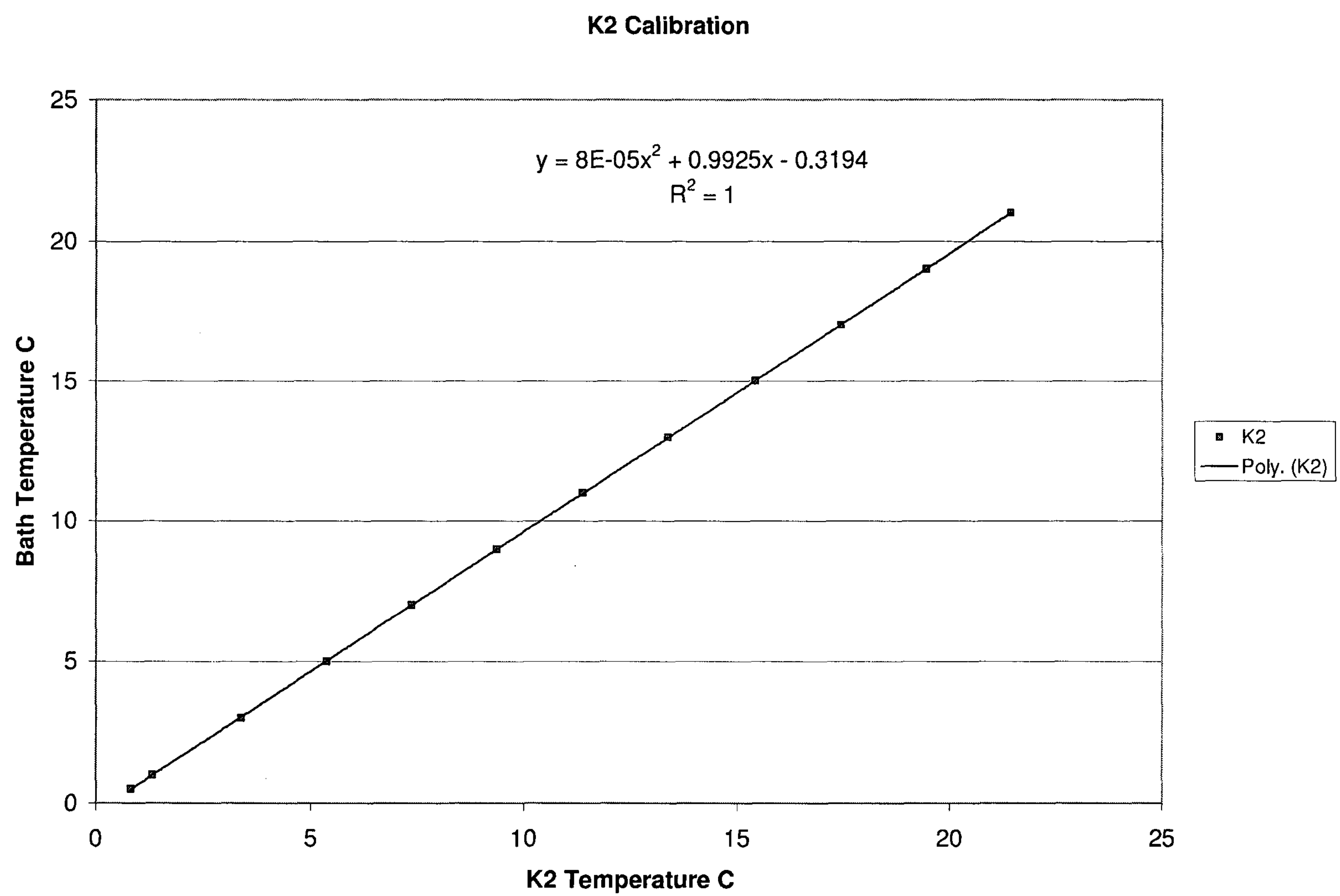
Variable	N	Const.	Coeff.	R^2	Std err	p
STRAHLER	40	0.135	0.122	0.260	0.176	0.001
SPC	39	1.068	-0.001	0.161	0.189	0.011
EXTCOEFF	39	0.551	0.047	0.157	0.186	0.013
LOGPERI	40	1.274	-0.342	0.147	0.188	0.015
TEMP	39	-0.019	0.029	0.125	0.193	0.027
DOC	40	0.593	0.006	0.085	0.195	0.068
SRP	35	0.584	0.004	0.083	0.186	0.094
TP	40	0.531	0.003	0.069	0.197	0.101
MATWILLDAYDI	40	0.464	0.005	0.066	0.197	0.110
TURBIDITY	40	0.596	0.015	0.065	0.197	0.111
SQRTCOLOUR	40	0.591	0.265	0.058	0.198	0.134
DISTOMTHKM	40	0.710	-0.001	0.002	0.204	0.770
MATSITEDSCH	38	0.613	0.002	0.009	0.208	0.564
PERIVEL	40	0.672	0.075	0.003	0.204	0.756
POOLDEPTH	40	0.747	-0.188	0.009	0.203	0.556
LOGNO3	35	0.679	0.016	0.002	0.194	0.786
LOGNH4	27	0.680	0.042	0.002	0.205	0.810

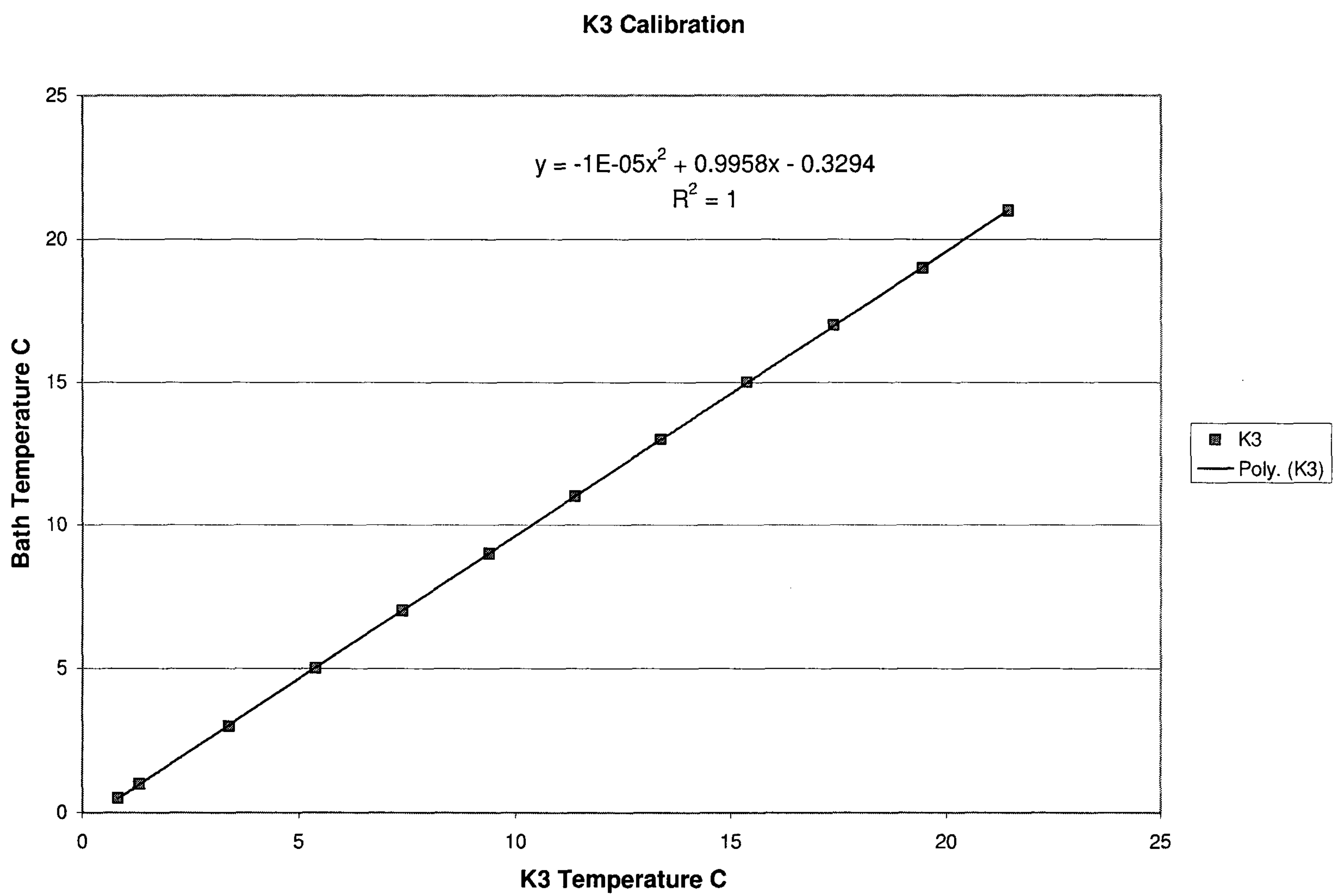
Table D5. Simple regressions of independent variables predicting periphyton biomass as Chl a (mg/m2).

Variable	N	Const.	Coeff.	R^2	SE	p
TURBIDITY	40	1.885	-0.029	0.196	0.205	0.004
LOGSUSPUG/L	40	1.998	-0.429	0.147	0.211	0.015
LOGSUSPMGM2	40	1.829	-18.421	0.111	0.215	0.036
TP	40	1.876	-0.003	0.065	0.221	0.111
DOC	40	1.789	-0.005	0.053	0.222	0.151
SQRTCOLOUR	40	1.806	-0.275	0.050	0.223	0.166
EXTCOEFF	39	1.794	-0.029	0.048	0.226	0.181
LOGNO3	35	1.831	-0.090	0.051	0.225	0.193
SPC	39	1.482	0.000	0.043	0.226	0.205
STRAHLER	40	1.896	-0.043	0.025	0.225	0.325
DISTOMTHKM	40	1.759	-0.002	0.017	0.226	0.421
MATSITEDSCH	38	1.663	0.001	0.003	0.208	0.750
MATWILLDAYDI	40	1.707	0.000	0.000	0.228	0.970
PERIVEL	40	1.728	-0.099	0.004	0.228	0.711
POOLDEPTH	40	1.704	-0.010	0.000	0.228	0.978
TEMP	39	1.426	0.011	0.015	0.229	0.460
LOGNH4	27	1.836	-0.127	0.017	0.229	0.512
SRP	35	1.732	-0.002	0.008	0.230	0.604

Appendix E: Calibration curves for iButtons® (University of Waterloo)




K2 Calibration
y = 8E-05x2 + 0.9925x - 0.3194
R2 = 1
Bath Temperature C
K2 Temperature C
0
5
10
15
20
25
K2
Poly. (K2)


K3 Calibration
y = -1E-05x2 + 0.9958x - 0.3294
R2 = 1
Bath Temperature C
K3 Temperature C
0
5
10
15
20
25
K3
Poly. (K3)

Appendix F: iButton -recorded surface water temperatures vary with air temperature

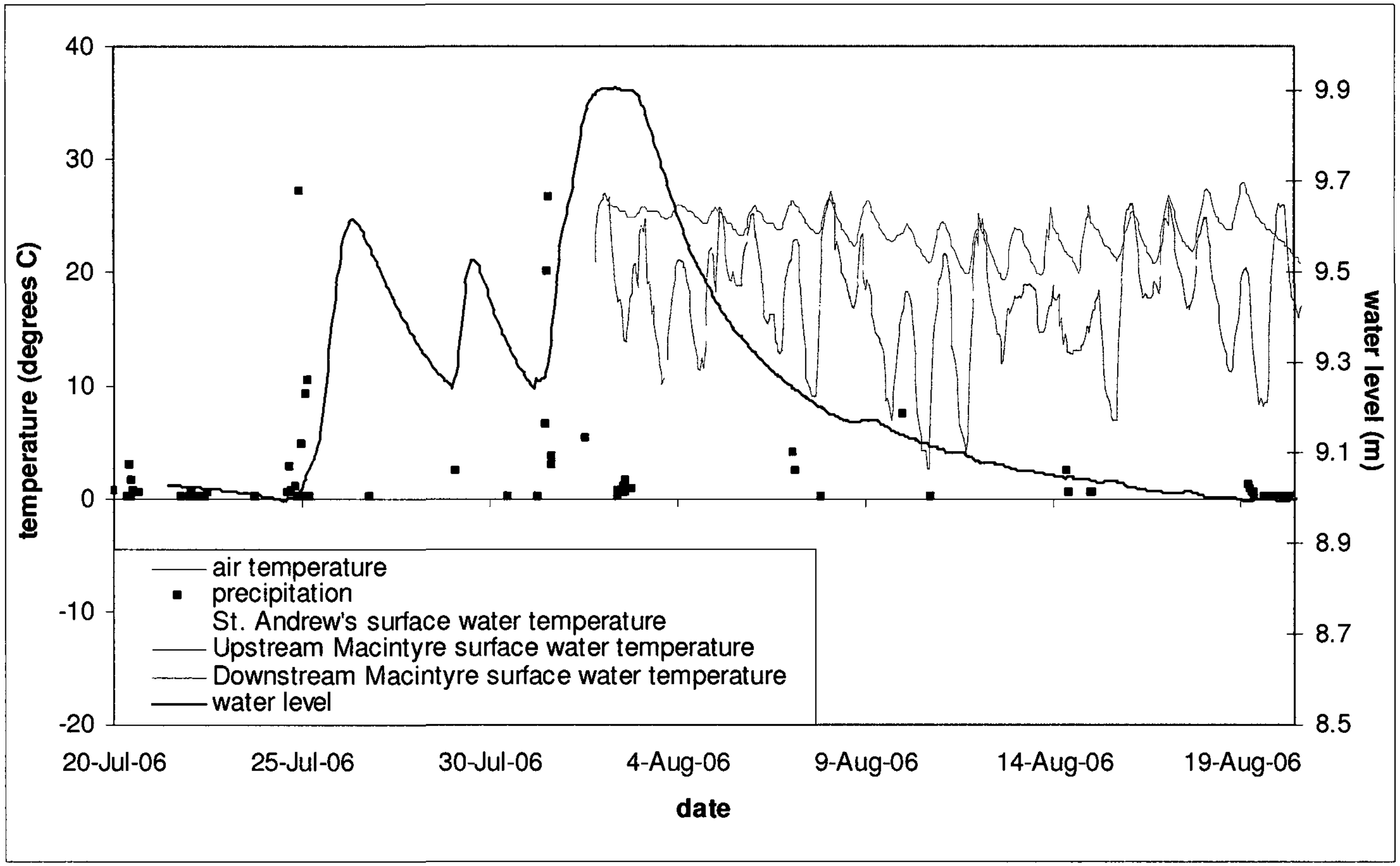


Figure F1. Surface water temperatures recorded by iButton loggers vary with air temperature (2pm, July 20th – 12am, Aug. 21st, 2006).

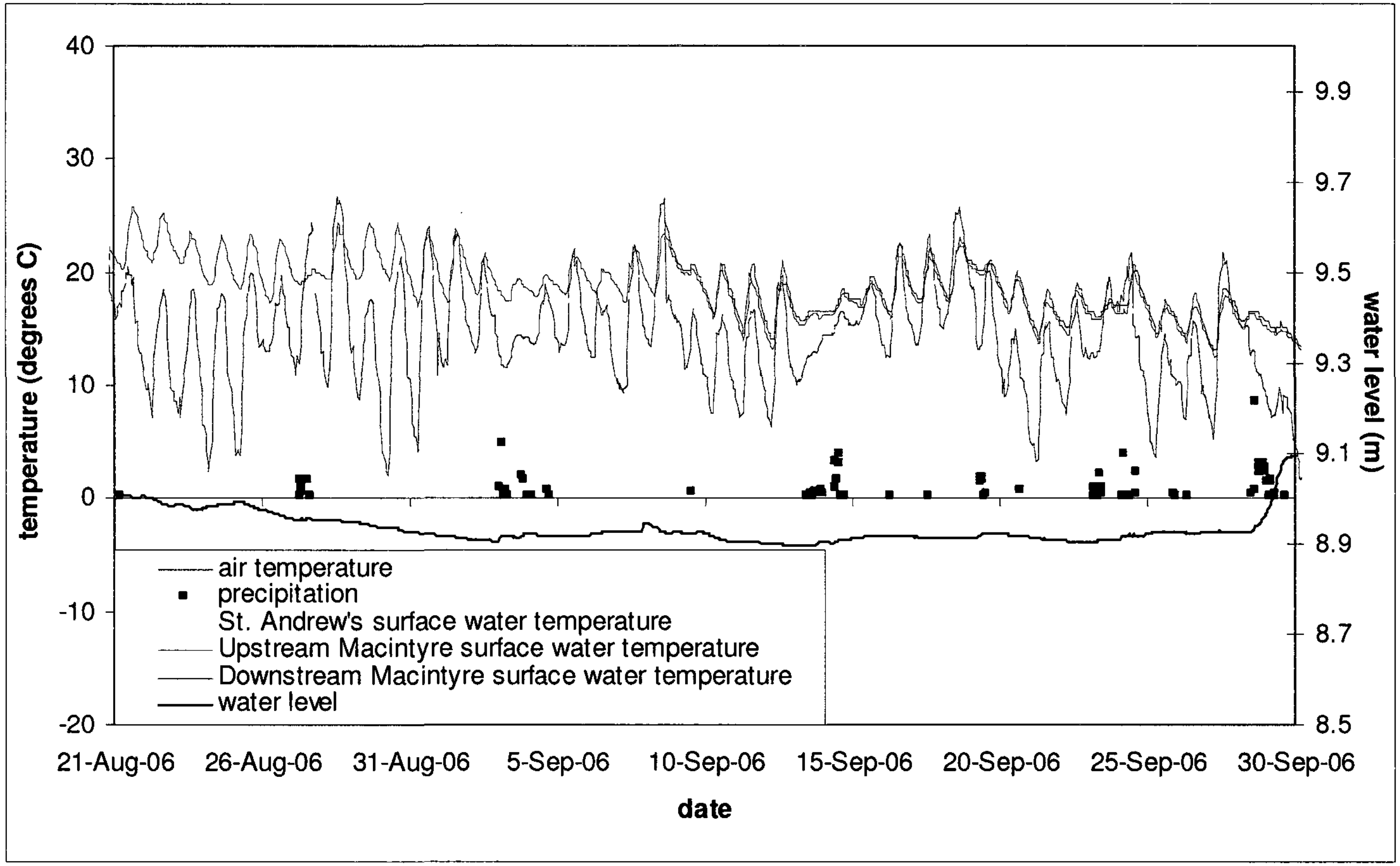


Figure F2. Surface water temperatures recorded by iButton loggers vary with air temperature (12am, Aug. 21st – 12am Sept. 30th, 2006).

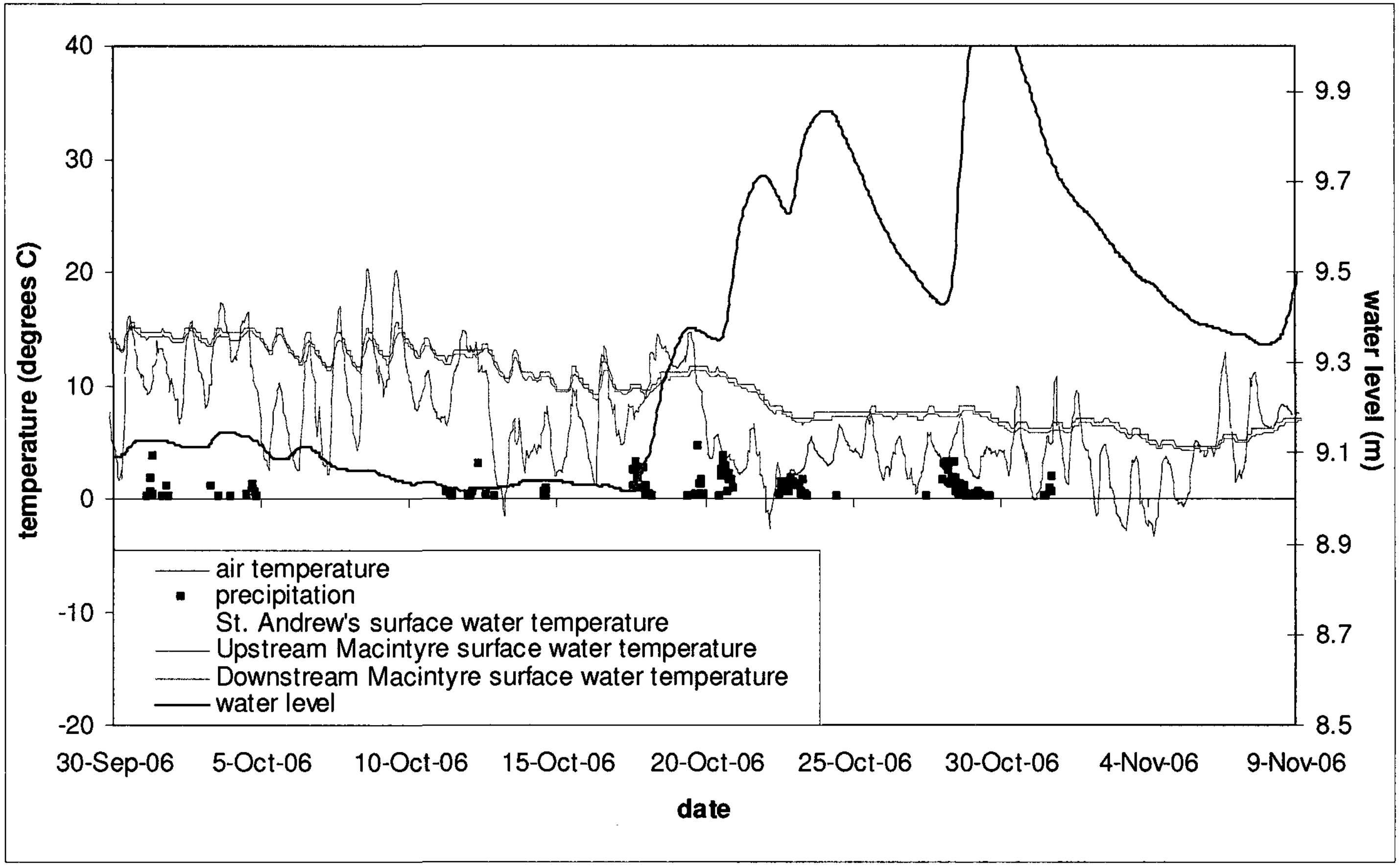


Figure F3. Surface water temperatures recorded by iButton loggers vary with air temperature (12am, Sept. 30th – 12am Nov. 9th, 2006).

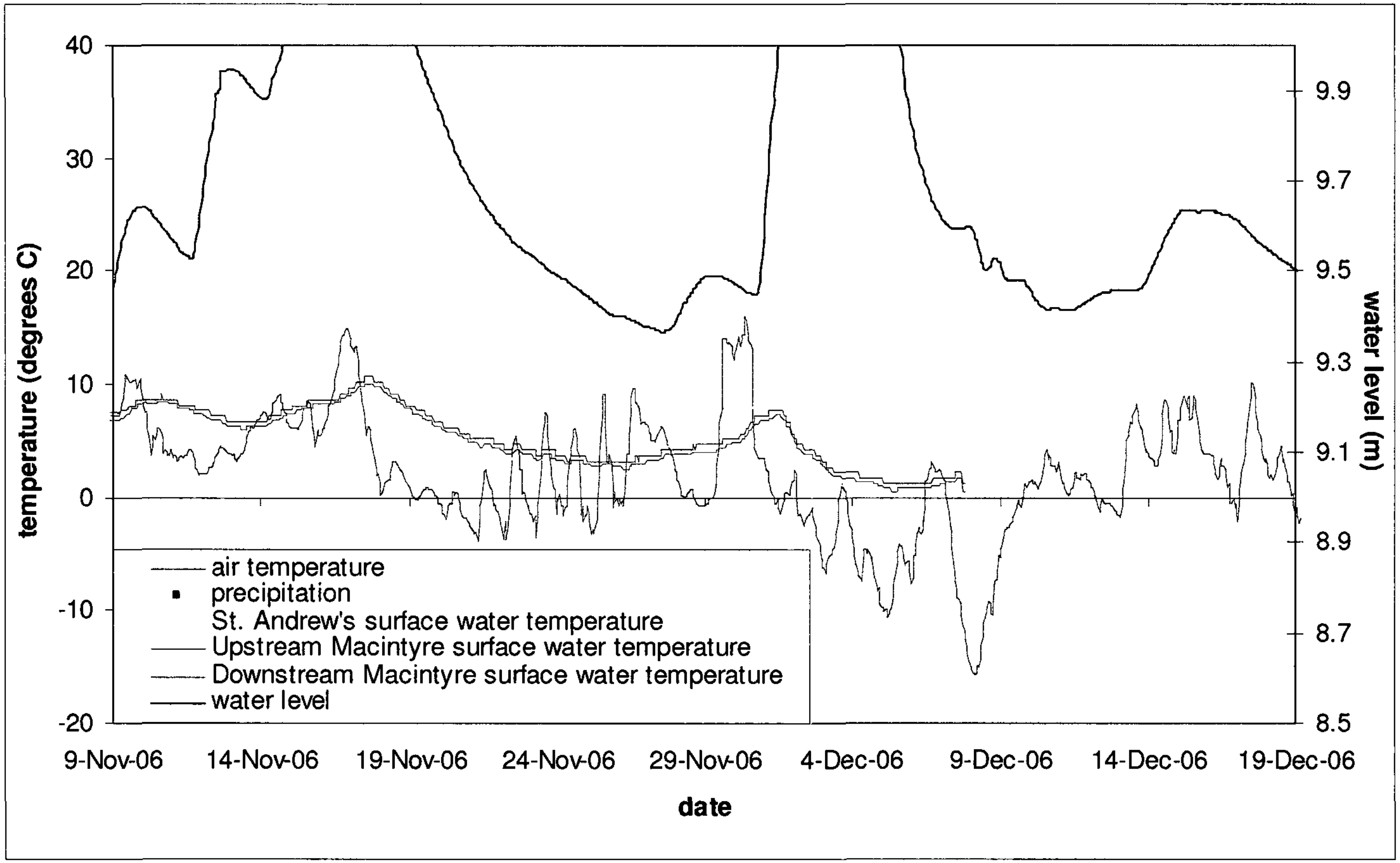


Figure F4. Surface water temperatures recorded by iButton loggers vary with air temperature (12am, Nov. 9th – 12am, Dec. 12th 2006).

Appendix G: Spectral analysis.

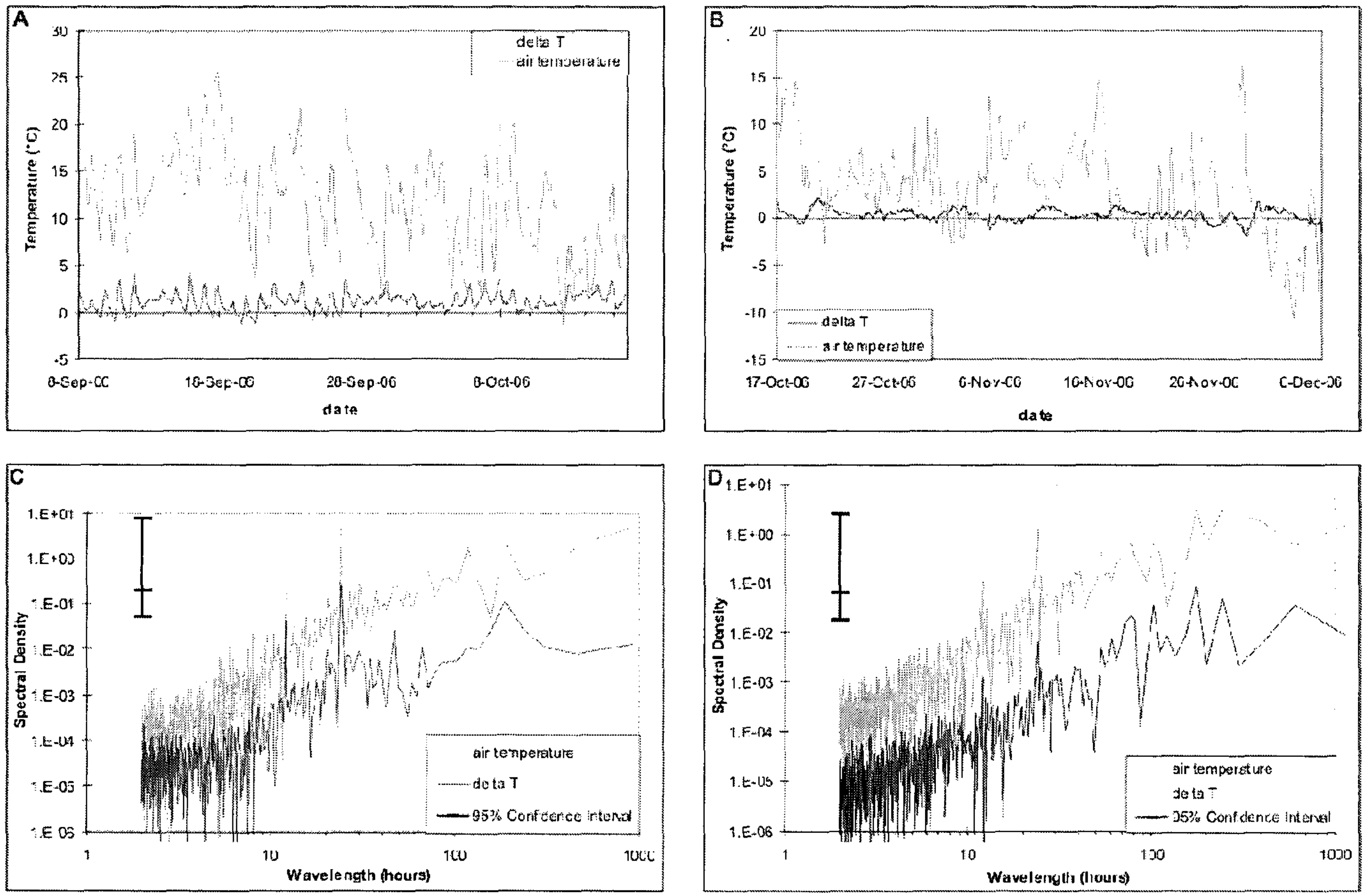


Figure G1. Spectral analysis for surface water temperature differences between the St. Andrew's site and the Macintyre site (delta T) and air temperatures. a) and b) are raw data, c) and d) are periodograms with peaks at 12 and 24 hours for each data set.