Effects of Hydroelectric Dams on Downstream Oxygen and Nitrogen in the Grand River and Conestogo River

by

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Author's Declaration

I hereby declare that I am the sole author of this thesis. This is a true copy of the thesis, including any required final revisions, as accepted by my examiners.

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Abstract

As the global population has increased, so has the need for water supply and energy. Dams have been a point of contention for many years. In some regions they are increasing in importance, but new constructions are becoming more massive and more geographically remote. In other regions, dams have been called into question, as their overall impacts are not fully understood and what is known may not be universally applicable, particularly as the structures age.

The Grand River, Ontario, watershed has a tendency towards high nitrogen concentrations and low nighttime dissolved oxygen (DO) concentrations. Two large dams and their associated reservoirs were studied; Belwood Lake on the Grand River, and Conestogo Lake on the Conestogo River. The focus of this study was to investigate how these bottom-draw dams alter the downstream oxygen cycling and nitrogen cycling. This was done by direct, in-field measurements taken above and within the reservoirs, as well as directly below the dams and downstream. The study was conducted over three years (2008-2010). Seasonal, vertical profile and diurnal measurements were taken for temperature, DO, $\delta^{18}O$-O_2, NO_3^-, NH_4^+, total dissolved nitrogen (TDN) and N_2O.

The DO cycle is altered downstream of the reservoirs, particularly at Belwood Lake where, despite reaeration measures to mediate the low DO concentrations released from the hypolimnion of the stratified lake, downstream respiration still causes the DO to fall below minimum standards. This effect was shown to be seasonal as a result of management practices. Full recovery of the DO downstream of Belwood Lake to above reservoir conditions was not observed before the next impoundment. Recovery was seen downstream of Conestogo Lake.

The reservoirs behaved as a sink for NO_3^- in the spring, as they filled from the winter runoff. As the water was released through the summer, the stored NO_3^- was then released downstream, thus acting as a source. There was evidence that the reservoirs were a source of

NH_4^+ in both spring and summer. High N_2O was measured directly below the dams on occasion; but not further downstream.

This study has provided insight into the behavior of the area directly below bottom-draw dams, which is often ignored or overlooked. It has also shown that there can be differences between reservoirs and dams of similar morphology, depth, and retention in the same basin. Further research should be conducted at locations further downstream to see the true nature of the downstream environment.

Acknowledgements

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1.0 Introduction

1.1 General Understanding of Rivers, Reservoirs and Dams

1.1.1 Reservoir Significance

Anthropogenic disturbances to aquatic ecosystems have been commonplace throughout much of human history. The primary motivations for these alterations are varied and complex. The purpose for alteration of the natural flow of water ranges from irrigation and drinking water in the past to include flood control, flow augmentation and recreation in more recent history (Morris and Fan, 1998). Within the past century, however, the focus has shifted toward the production of hydroelectricity. Though these uses of water are important to accommodate the expanding human population, the effects on the natural system can be devastating (Kennedy *et al.*, 2003).

The most current estimate provided by the World Commission on Dams (2000) is that 60% or more of the world's rivers have been impounded in some way. This number is likely low since more than a decade has passed since these data were tabulated. Worldwide, about 40% of water used for irrigation has its source at a reservoir. Power generated from hydroelectric sources makes about 20% of the total electricity produced (World Commission on Dams, 2000). Societal needs require reservoirs to regulate the irregular flow of surface water, and humans are becoming increasingly reliant on dams (Morris and Fan, 1998). It has also been suggested that dams have even contributed to changes in sea level (Tranvik *et al.,* 2009), as the amount of water withheld from natural flow has increased.

Though the major peak of construction occurred in the 1970s, dams are still being built. Many suitable sites for dams in developed countries have already been dammed. Most of the new dams are found in countries that are less developed or have lower economic wealth. Dams there are often in more geographically extreme areas and are likely to be much larger and have significantly larger impacts (Malmqvist and Rundle, 2002). Some projects are now in areas which require the relocation of villages, towns or cities. An excellent example of this is the Three Gorges Dam on the Yangtze River which, in its construction, required the relocation of almost 2 million people (Gleick, 2008).

Though there are benefits anticipated and realized from dams, many dam projects fail to reach initial expectations of production. For example, electricity production is often lower than initially predicted, with higher financial costs; as well, the environmental costs may be higher than anticipated (World Commission on Dams, 2000). A greater understanding of the responses of aquatic ecosystems to dams is crucial to making wise development decisions (Wetzel, 2001).

1.1.2 Brief History of Impoundments

Anthropogenic manipulation of water began thousands of years ago. The oldest dam still in operation is the Kofini diversion dam used to protect the village of Tiryns in Greece; built c. 1260 BCE (Morris and Fan, 1998). The oldest reservoir still in operation is the Afengtang reservoir located west of Shanghai; built between 589 and 581 BCE (Morris and Fan, 1998). Some of the first reservoirs in the Americas have been found in Central America as archaeological remains from Mayans living there c. 50 BCE (Matheny, 1976). The longest span for a functioning reservoir is held by the Mala'a reservoir in Egypt. Under King Amenemhet III, the reservoir's construction began c. 1842 BCE and served until the eighteenth century, about 3600 years. The remains of many other dams and reservoirs have been found throughout the world, including in the Mediterranean, Asia and Central America (Morris and Fan, 1998).

1.1.3 Three River Concepts

The River Continuum Concept

A basic model of river ecosystem function was proposed by Vannote *et al.* (1980). Their River Continuum Concept (RCC) describes the physical and biotic conditions for a river system from headwaters to mouth.

In the headwaters, the river is strongly influenced by riparian vegetation and allochthonous inputs. The middle reaches are less influenced by the terrestrial inputs of the surrounding area, and autochthonous production plays a larger role. The lower reaches depend mostly on autochthonous production, and have the least dependence on riparian inputs. The biotic community changes as stream order, morphology and energy inputs change.

Though the River Continuum Concept may describe the basic principles of the river ecosystems, it has several shortcomings. The model does not consider the effects of flooding, and the interaction with the shoreline during flooding. It lacks consideration for anthropogenic contributions and modifications such as wastewater inputs and agricultural activity in the riparian zone. A large oversight is the assumption that there is no physical disturbance in the system. Most river systems now have some type of impoundment or other physical modification.

The Serial Discontinuity Concept

The Serial Discontinuity Concept (SDC) was developed later by Ward and Stanford (1983) in response to the River Continuum Concept. It took the main principles of the River Continuum Concept and built from them. The Serial Discontinuity Concept recognizes that there are a limited number of river systems that are free flowing, and attempts to provide a theoretical understanding of regulated systems. The SDC makes four major assumptions:

- The RCC and nutrient spiraling concepts (Newbold *et al.*, 1981) are essentially valid
- The river in question is free of pollution and disturbances other than impoundment

- Remaining system was not harmed by the construction of the dam
- Impoundments are assumed to thermally stratify and release deep water that is not oxygen-deficient

With these assumptions, the Serial Discontinuity Concept goes further into describing changes due to impoundment for theoretical impoundments in the headwaters, mid-reach and lower-reach. Temperature and flow were predicted to change the most in the mid-reach, whereas nutrients were expected to increase in the headwaters and decrease in the mid and lower-reaches.

Though the model does include more details, it too is idealized. Similar to the RCC, it does not consider other anthropogenic disturbances, such as agricultural runoff and wastewater input. Despite being dated and idealized, both the RCC and the SDC are still commonly referred to in both scientific and management literature.

The Cascading Reservoir Continuum Concept

A third concept to be addressed is the Cascading Reservoir Continuum Concept (CRCC) suggested by Barbosa *et al.* (1999). This was formulated in order to account for a common situation, most large rivers have more than one reservoir. The CRCC attempts to build from the RCC and the SDC, assuming that they are essentially valid. It differs though, in that it includes the potential connection between upstream and downstream reservoirs through ecological processes.

1.1.4 Reservoir Fundamentals

Dams modify the river and surrounding landscape in a number of ways. The flux of water and sediments downstream become altered, temperatures and temperature regimes downstream of the dam will be different from those upstream, organism movement upstream will be halted and organism and nutrient movement downstream will change dramatically (Poff and Hart, 2002). Impoundments often have a high watershed area in comparison to reservoir area, resulting in a heavily influenced water body (Thornton *et al.,* 1990; Lind, 1993; and Boyd *et al.,* 2000). In most reservoirs the water retention time is much shorter than that of a natural lake (Baxter, 1974). These changes each add to the modification of biogeochemial cycles, in addition to altering the structure of both aquatic and terrestrial habitats (Poff and Hart, 2002). The possibility of community isolation and fragmentation is an issue when an impoundment is made. There are several upstream factors that influence reservoirs, and several effects that may occur downstream as a result (Table 1.1**).**

Table 1.1: Potential factors regarding dam and reservoir construction that may alter the aquatic environment, adapted from Kalff (2002).

Upstream water quality	Reservoir quality	Operational concerns	Stream and floodplain quality	Downstream water quality
Precipitation chemistry	Depth & Shape	Discharge pattern	Thermal regime	Riparian vegetation
Basin geochemistry	Retention time	Release depth	Flow regime	Wetland size
Terrestrial vegetation	Impoundment age		Groundwater	Ecological diversity
Climatic conditions	Turbidity		Nutrient levels	Biotic productivity
Anthropogenic inputs	Drawdown extent		Substratum type	Migrations
Wetland vegetation and chemistry	Thermal stratification		Water clarity	Species composition
River discharge	Hypolimnetic anoxia		Dissolved salts	Life-cycle phenomena
	Sedimentation		Dissolved gases	Potable water
	Productivity		Organic Detritus	
	Recreation			

Zonation

Reservoirs are artificial systems that have properties of both lakes and rivers thus acting as a river-lake hybrid (Scott *et al.,* 2009). Thornton *et al.* (1990) distinguished three longitudinal zones of a reservoir (Figure 1.1).

- Riverine
- Transition

- Lacustrine

The zone with characteristics most similar to a river system is the riverine. It is relatively narrow, with the potential for a channelized basin (Thornton *et al.,* 1990). The flow is relatively high, and the water is often well mixed (Thornton *et al.,* 1990). There may be high nutrient concentrations, and oxygen concentrations should be high (Kalff, 2002).

The lacustrine zone is the main body of water within the reservoir system. It is usually broad and deep, resembling the basin of a lake (Kalff, 2002). There is little flow in this zone, and flushing is dependent upon release rates and outflow location. Nutrients become limiting and oxygen availability may decline at depth if stratification occurs (Thornton *et al.,* 1990).

The zone dividing the riverine and the lacustrine is the transition. Its morphometry is usually broader than the river with an increase in basin depth (Thornton *et al.,* 1990). The flow is reduced as is the flushing rate. The nutrient and oxygen availabilities will vary depending on seasonal conditions such as flooding or stratification (Thornton *et al.,* 1990). Studies have shown that the transition zone is the area with the highest potential for productivity (Scott *et al.,* 2009).

Though variations among reservoirs exist, each reservoir has all three previously mentioned zones regardless of uniqueness or annual variations. However, these zones may shift as reservoirs are never in a steady state (Thornton *et al*., 1990).

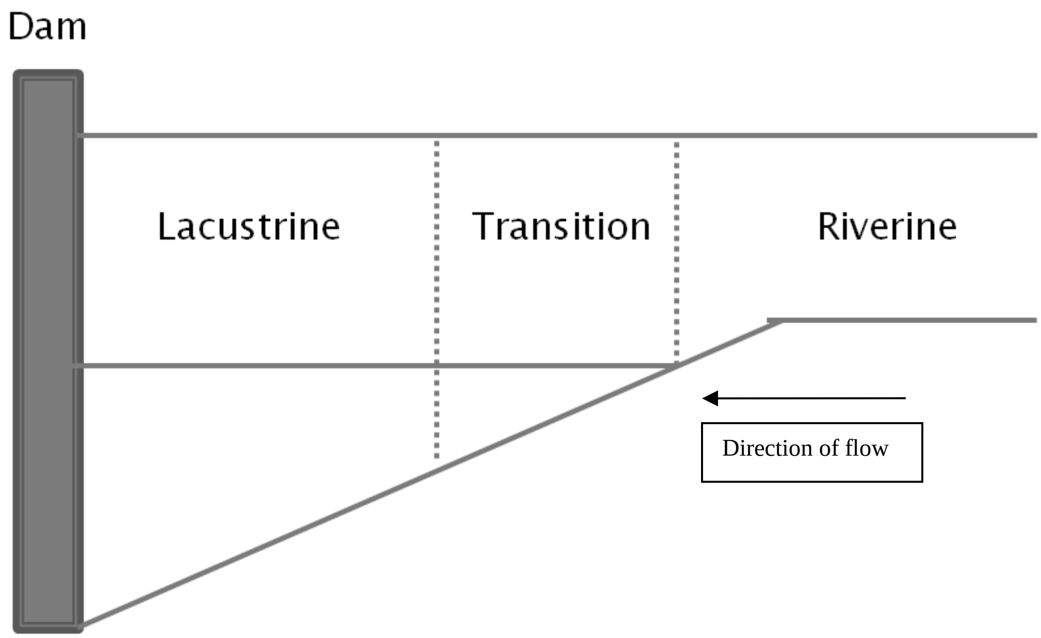

Figure 1.1: Model of Reservoir water regime as proposed by Thornton *et al.*, (1990) indicating the zones of a reservoir.

Dam Construction

Though dams are created for different purposes, a common theme to all is the stabilization of water flow. Most operations store water during wet seasons in order to prevent flooding and use it for flow augmentation or irrigation in dry seasons. Other options for dam use are hydroelectricity production, drinking water and recreation.

The design of a dam is dependent upon intended purpose. Dams can be constructed in different ways: embankment dams (either earth filled or rock filled); gravity dams (usually concrete); arch dams (concrete) and arch-gravity dams (concrete) (Morris and Fan, 1998). There are also different release points that are possible for a dam. A surface release dam will draw epilimnetic water from behind the impoundment, while a deep release will draw from the hypolimnion. Some dams have also incorporated a third method in which water can be drawn from the metalimnion. This last option gives what is known as selective withdrawal, for dams that have been constructed with the ability to draw from all three zones (Stanford and Ward, 2001).

Water Retention

Though most reservoirs have short water residence times (Baxter, 1974), the majority of reservoirs attempt to operate within a range of discharge in order to maintain downstream water levels within acceptable limits (Thornton *et al*., 1990). These minimum releases can increase hypolimnetic mixing and thus may promote the release of nutrients downstream (Thornton *et al*., 1990).

The location of maximum depth in most reservoirs occurs directly in front of the dam, unlike natural lakes where it may be found anywhere in the basin (Baxter, 1974). Many artificial lakes discharge from depth, causing alterations of the physiochemical environment and the downstream environment (Whalen *et al.*, 1982).

Drawdown

Seasonal changes in water level are a common feature of reservoirs, often requiring a manual adjustment in the level of water. Many release a significant amount of water downstream in order to drop the water level. Often this is done in the fall in order to create storage capacity to ameliorate spring flooding (Boyd *et al.*, 2000). This major decrease in water level can severely alter the ecosystem and the chemical composition within the reservoir as well as downstream (Baxter, 1977; Thornton *et al.*, 1990; Shantz *et al.*, 2004).

Sedimentation

In most free-flowing rivers the sediment inputs and outputs for a reach are usually consistent, with higher inputs expected during certain events (e.g. heavy rain); the construction of a dam will significantly change this (Morris and Fan, 1998). The amount of sediment entering a reservoir is generally higher, and the concentration of fine particles is greater than in natural lakes (Thornton *et al*., 1990).

As the reservoir ages there is a buildup of sediment in the reservoir, most often behind the dam. Hydraulic flushing is a method in which excess sediments can be removed. If the water is let out of low-level outlets it is possible to let this flow take accumulated sediments with it (Morris and Fan, 1998). Though this practice will maintain a functioning volume within the reservoir, problems may arise with the release of large amounts of sediment downstream. Heavy metals, chemicals, and other toxins could have accumulated and be stored in the sediments being released, thus further polluting the downstream environment (Bushaw-Newton *et al.,* 2002).

Temperature

Temperature plays a large role in both river and reservoir function, and is a critical parameter for aquatic life. Many reservoirs demonstrate the ability to thermally stratify. This can potentially give rise to temperature changes in the river system. In winter months, deep release dams would provide the downstream environment with warmer water, conversely, in summer the deep release would be sending cooler water downstream (Thornton *et al.,* 1990).

1.1.5 Oxygen Cycling in Reservoir Impacted Ecosystems

Dissolved oxygen is important for the regulation of aquatic biochemical cycles and plays an important part in water quality (Wetzel, 2001). Dissolved oxygen can also be used as an indicator to assess ecosystem health (Odum, 1956). Most aquatic animals require a minimum dissolved oxygen concentration for survival (Venkiteswaran *et al.,* 2008; Barton and Taylor, 1996; Chapman, 1986).

Several processes control the dissolved oxygen concentrations in aquatic systems; principally primary production, respiration, and gas exchange. Primary production releases oxygen into the water, whereas during respiration there is an uptake of oxygen. Gas exchange occurs at the water-air interface (Odum, 1956; Venkiteswaran *et al.,* 2007). Interaction between the surface and ground water is also a possibility, which potentially could be an additional source of oxygen or of anaerobic water (Odum, 1956). The salinity of the water will also change the dissolved oxygen concentration. A decrease in temperature will increase the DO solubility; a decrease in pressure or increase in salinity will lower the solubility (Kalff, 2002).

The process of photosynthesis by macrophytes, algae and phytoplankton adds dissolved oxygen to the aquatic environment. A simplified formula (Kalff, 2002) is:

$$6CO_2 + 6H_2O \rightarrow C_6H_{12}O_6 + 6O_2 \qquad \textbf{Equation 1.1}$$

The reverse equation would be for respiration. The back and forth of oxygen uptake and oxygen release causes a distinct diurnal pattern following the sun (Kalff, 2002). Though this ongoing pattern is natural in its occurrence, if the conditions are such that daytime photosynthetic rates are extraordinarily high, the water can become supersaturated with dissolved oxygen. Conversely, high respiration can cause hypoxic conditions to develop during the night that are unsuitable for some aquatic life.

The presence of a reservoir may change the oxygen cycle, with numerous consequences. If the reservoir is deep enough to thermally stratify, the hypolimnetic water could eventually decrease in oxygen concentration in response to biological oxidation of organic matter (Wetzel, 2001). Depending on the type of dam (as previously discussed) this low oxygen or anoxic water could be released to the detriment of aquatic life downstream.

1.1.6 Nitrogen Dynamics in Reservoir Impacted Ecosystems

The earthly abundance of nitrogen is 0.003%; 97.76% is in rocks, 2.01% is in the atmosphere, and the remaining 0.23% is divided between the hydrosphere and biosphere (Aravena and Mayer, 2009). For plants and other organisms, nitrogen is one of the most important elements for growth (Kalff, 2002).

Nitrogen can occur in nature in many forms. Some of the biologically available forms include ammonium (NH_4^+), nitrite (NO_2^-) and nitrate (NO_3^-). There are a number of sources for nitrogen such as precipitation, fixation, and surface/groundwater inputs (Wetzel, 2001). Loss of nitrogen from the ecosystem will also occur by bacterial reduction of NO_3^- to N_2 (denitrification), system outflow or sedimentation (Wetzel, 2001).

Fixation

Though it is an energy-consuming process (using adenosine triphosphate or ATP and nicotinamide adenine dinucleotide phosphate or NADPH), nitrogen fixation can be carried out by bacteria, including cyanobacteria, and consequently adds available nitrogen to the aquatic system. The process involves the reduction of N_2 into amines, and often occurs in a specialized cell called a heterocyst.

In reservoirs, the N-fixation that occurs along the riverine-transition-lacustrine gradient can vary greatly. Scott *et al.* (2009) found that in the transition zone nitrogen fixation occurred at rates from 25-60 times higher than in the riverine and lacustrine zones as it is considered a biogeochemical hot spot.

Nitrification

Nitrification is the conversion of ammonium (NH_4^+) to nitrate (NO_3^-). For the overall reaction to proceed to completion, 2 moles of O_2 are required for each mole of NH_4^+ (Wetzel, 2001). The first reaction (Equation 1.2) is the oxidation of ammonia to nitrite, followed by (Equation 1.3) the oxidation of nitrite to nitrate (Wetzel, 2001). The overall reaction is shown in Equation1.4:

$$NH_4^+ + 1\tfrac{1}{2}\, O_2 \rightarrow 2H^+ + NO_2^- + H_2O \qquad \textbf{Equation 1.2}$$

$$NO_2^- + \tfrac{1}{2}\, O_2 \rightarrow NO_3^- \qquad \textbf{Equation 1.3}$$

$$NH_4^+ + 2O_2 \rightarrow NO_3^- + H_2O + 2H^+ \qquad \textbf{Equation 1.4}$$

Environmental contamination of aquatic systems by nitrate is a global problem (Xue *et al.*, 2009). NO_3^- is one of the most common contaminants found in ground water (Aravena and Mayer, 2009). NO_3^- is potentially dangerous. Consequently, the World Health Organization has suggested a drinking water limit of 10 mg NO_3^- -N/L (World Health Organization, 2007).

Denitrification

Denitrification is a bacterially mediated process of the reduction of nitrogen oxides (NO_3^- and NO_2^-) to gaseous nitrous oxides (NO & N_2O), then to dinitrogen gas (N_2) (Kalff, 2002). This process is carried out by heterotrophic, facultative anaerobic bacteria in water bodies at the interface between the oxic-anoxic, and in the anoxic zones (Wetzel, 2001). The NO_3^- and the NO_2^- act as the electron acceptors during the oxidation of organic matter (Kalff, 2002). The denitrification process is responsible for the loss of fixed nitrogen to the atmosphere (Kalff, 2002). The reactions follow:

$$NO_3^- + 2H^+ + 2e^- \rightarrow NO_2^- + H_2O \qquad \textbf{Equation 1.5}$$

$$NO_2^- + 2H^+ + e^- \rightarrow NO + H_2O \qquad \textbf{Equation 1.6}$$

$$2NO + 2H^+ + 2e^- \rightarrow N_2O + H_2O \qquad \textbf{Equation 1.7}$$

$$N_2O + 2H{+}\ 2e^- \rightarrow N_2 + H_2O \qquad \textbf{Equation 1.8}$$

1.1.7 The use of Stable Isotopes

There are naturally occurring stable isotopes of hydrogen, carbon, nitrogen, oxygen and sulphur (Clark and Fritz, 1997). These stable isotopes can be used as indicators of many environmental cycles, e.g., water, nutrients, oxygen (Clark and Fritz, 1997).

Oxygen has three stable isotopes that occur naturally: oxygen-16, oxygen-17, and oxygen-18. The natural abundance of oxygen-16, oxygen-17 and oxygen-18 are 99.76%, 0.036%, and 0.204% respectively (Clark and Fritz, 1997).

These isotopes can be used as indicators of aquatic processes. If the dissolved oxygen is shown to be undersaturated, and has a $\delta^{18}O$ greater than 24.4‰, respiration is the dominant contributor (Quay *et al.*, 1995). If the dissolved oxygen is supersaturated and the $\delta^{18}O$ is less than 24.4‰, photosynthesis is dominant (Quay *et al.*, 1995). Finally, if the dissolved oxygen is close to saturation, and the $\delta^{18}O$ is about 24.2‰, gas exchange will be dominant (Quay *et al.,* 1995).

1.2 Study Area

1.2.1 The Grand River

The Grand River is located in Southern Ontario, Canada (Figure 1.2). The area is home to about 975,000 people, with projections of over 1,000,000 within the next decade. In 1994 the Grand was designated a Canadian Heritage River from the Canadian Heritage Rivers System (Department of Canadian Heritage, 1998).

From the headwaters in Dundalk, Ontario, the Grand starts at an elevation of 525 m above sea level and stretches over 300 km to Port Maitland, Ontario, ending at an elevation of 174 m above sea level on Lake Erie. The river basin is 6965 km^2 and includes a number of tributaries: Nith, Conestogo, Speed and Eramosa Rivers (Boyd *et al.,* 2000). The Grand River basin land use is primarily agriculture, at 78%. Natural vegetation accounts for 19%, leaving only 3% for urban settlement. There are 29 waste water treatment plants that discharge effluent to the Grand River and its tributaries.

Historically, the Grand River watershed has been altered by deforestation and wetland draining. This, in combination with floodplain settlement, has led to severe flooding (Boyd *et al.,* 2000). Conversely, during periods of less precipitation, the discharge of the river would be too low to support water abstraction and dilution of sewage (Boyd *et al.,* 2000). As management practices have been developed, there have been a number of anthropogenic changes to the river system. This included channelizing, diking and damming. The Grand River Conservation Authority (GRCA) has 32 dams for which it is responsible, and there are over 100 others that are either municipally or privately owned (Boyd *et al.,* 2000). Of the reservoirs that are currently maintained by the GRCA, 8 are of substantial size (1.5 million m^3 storage or larger).

1.2.2 Concerns in the Grand River, Ontario

The Grand River has a number of large impoundments. These reservoirs have historically thermally stratified, which has resulted in low dissolved oxygen in the hypolimnion. The dams are deep-release and thus the low-oxygenated water is released downstream.

There are a number of largely populated areas in the Grand River basin as well as a high level of agriculture. High concentrations of nitrogen species have been found (Rosamond *et al*., 2012).

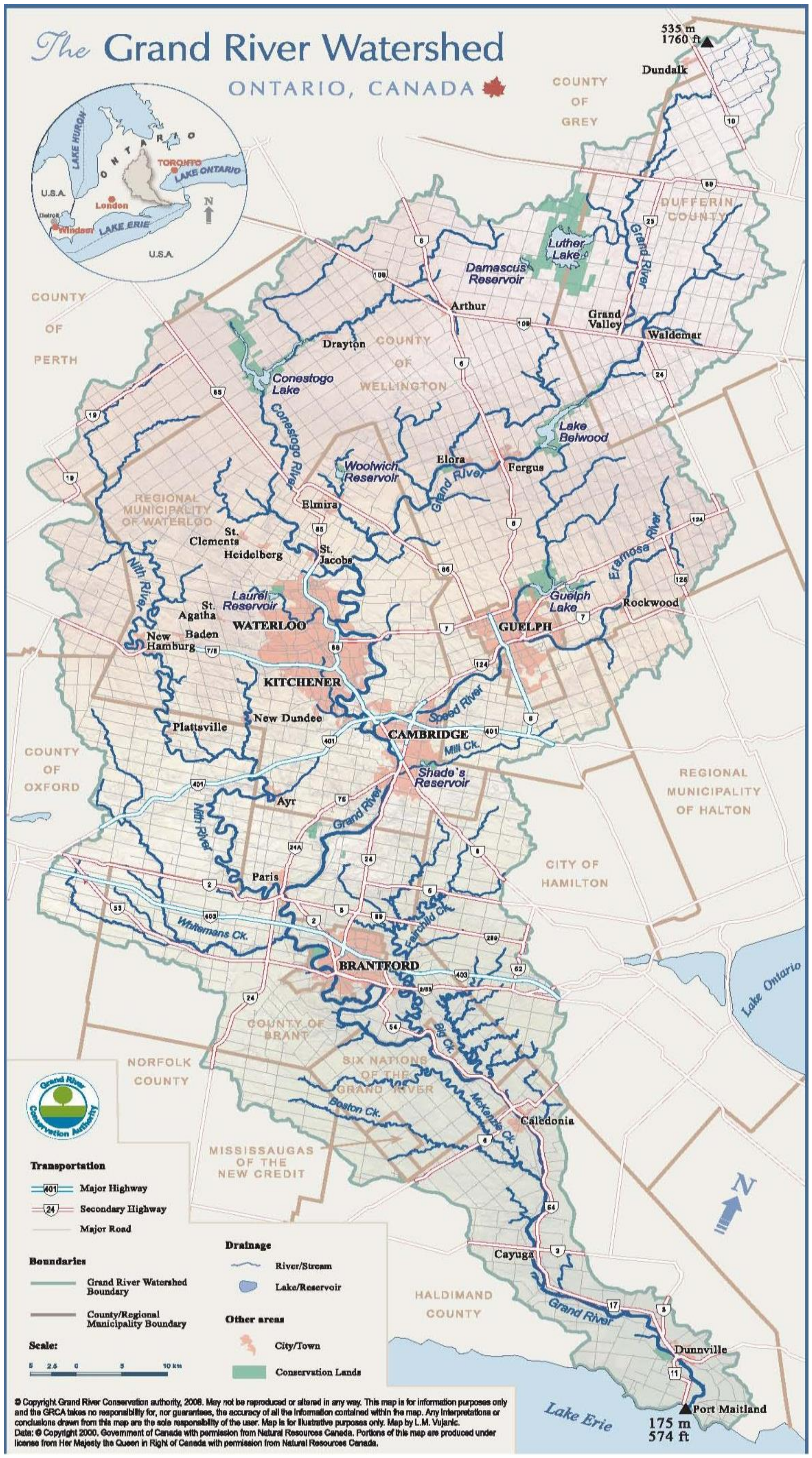

Figure 1.2: Map of the Grand River, Ontario, Canada. Provided by the Grand River Conservation Authority (2008).

1.2.3 Belwood Lake

The Shand Dam was constructed in 1942 as a solution to the flooding and drying of the Grand River. Belwood Lake is a mainstream storage reservoir; the Shand Dam is an earthfill dam with a clay core and concrete gravity spillway (Leach, 1975). Belwood Lake is northeast of Fergus, Ontario, on the Grand River. Retrofitting of the dam for hydroelectricity production occurred later in 1987, and was updated in 2009. The dam is 640 m long and 22.5 m high. At maximum capacity, the reservoir extends for 12 km upstream. The basin area for the reservoir is 802 km^2. The maximum depth occurs in front of the dam at 20.7 m and it can hold a volume of 63,874,000 m^3.

1.2.4 Conestogo Lake

The success of the Shand Dam led to further large dam projects on the Grand River. Another large dam was built in 1958. The Conestogo Dam is located northwest of Elmira, Ontario, on the Conestogo River, a main tributary of the Grand River. Similar to the Shand Dam, the Conestogo is a mainstream storage reservoir with a deep release or bottom draw dam. The dam was designed for flood control and low flow augmentation; it was retrofitted for hydroelectricity production in 1991, and later updated in 2006. The dam is 550 m long and 23.1 m high. The inundation extends for 9.66 km upstream at maximum capacity and has a drainage area of 563 km^2. It has a dendritic 'Y' shape. The maximum depth is 19 m, and is found directly in front of the dam. The maximum volume is 59,457,000 m^3.

Figure 1.3: Sampling Sites at Belwood Lake.

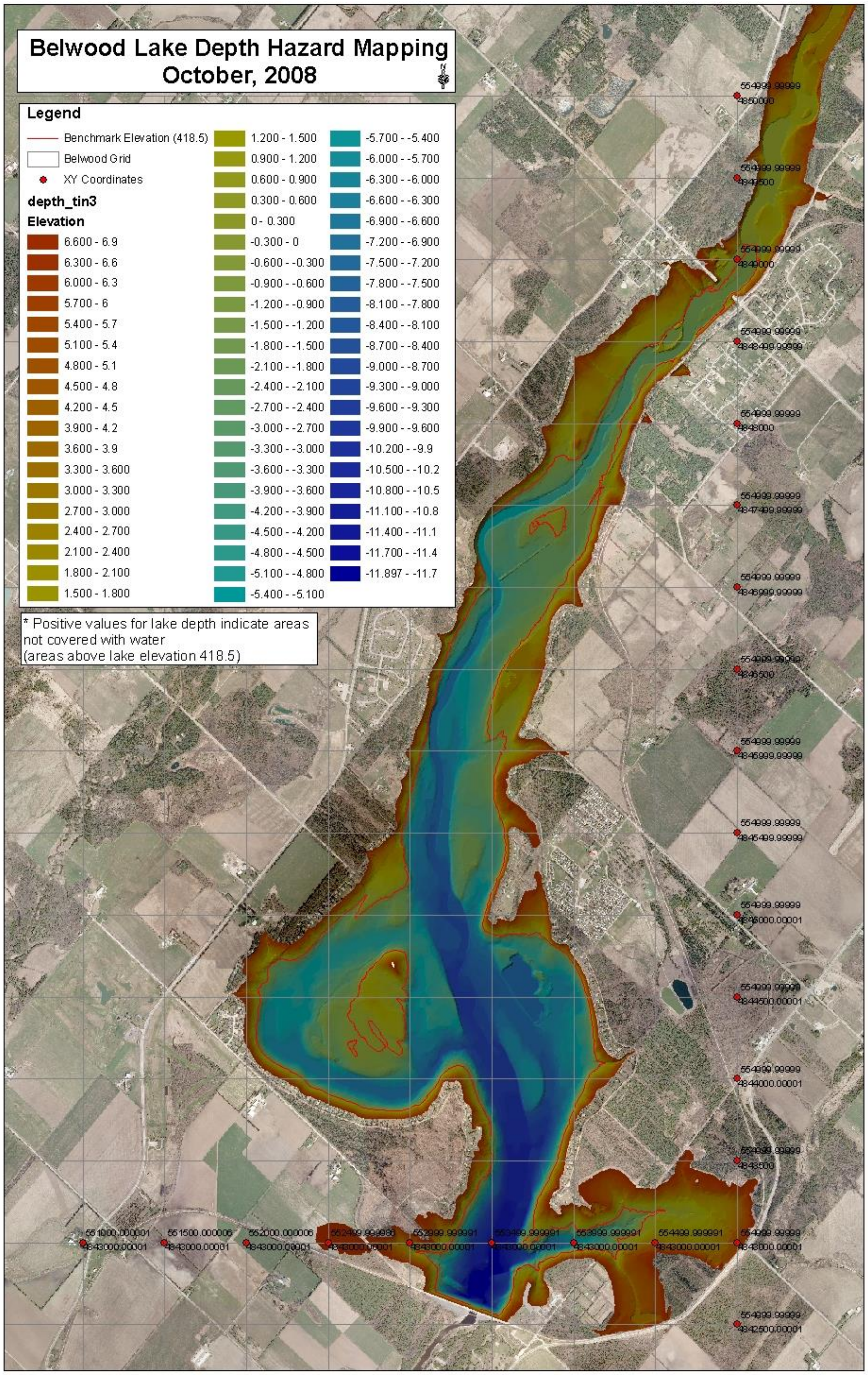

Figure 1.4: Belwood Lake depth hazard map. Provided by the Grand River Conservation Authority (2008).

Figure 1.5: Sampling sites at Conestogo Lake.

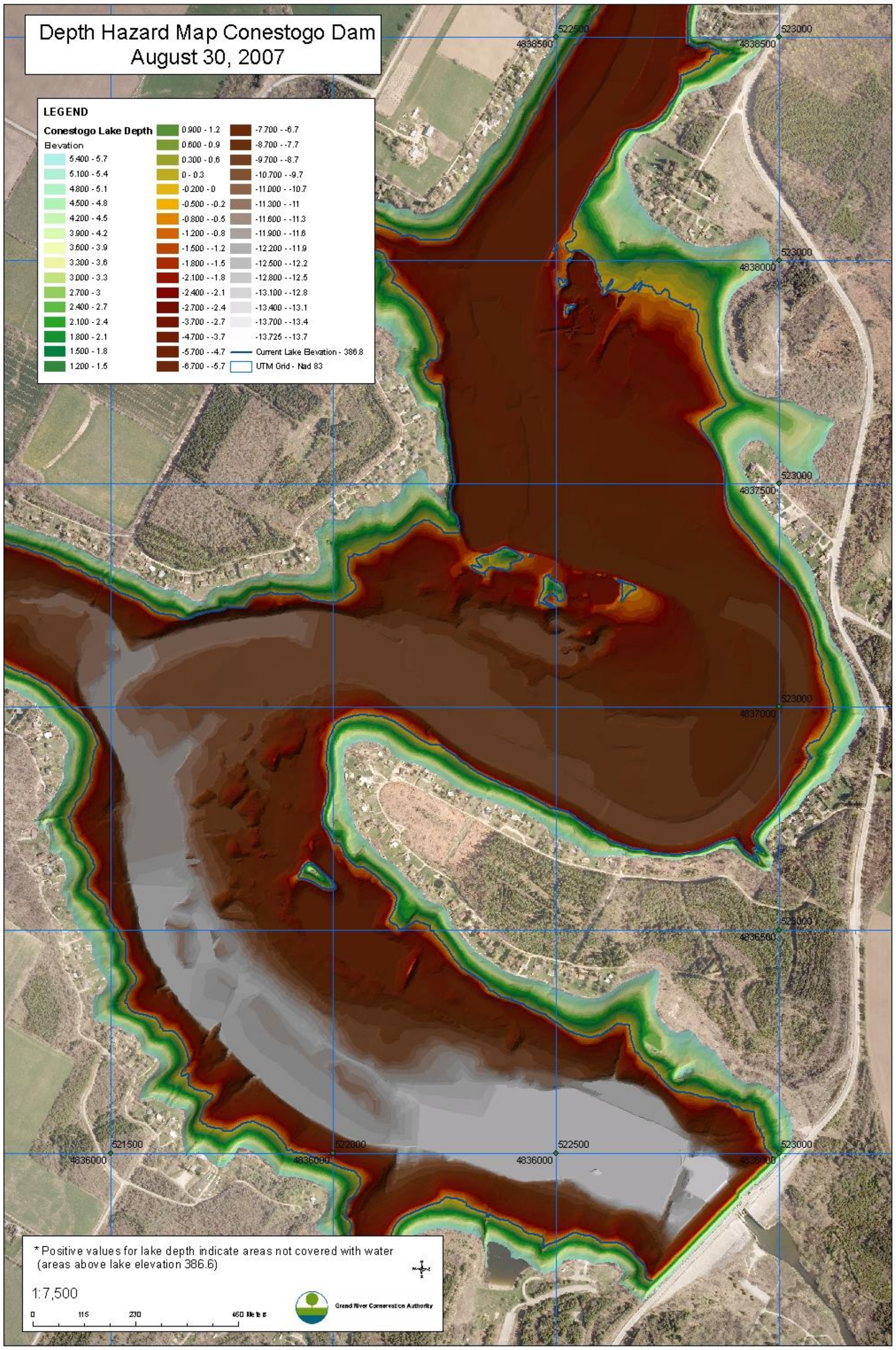

Figure 1.6: Conestogo Lake depth hazard map. Provided by the Grand River Conservation Authority (2008).

1.3 Thesis Structure and Objectives

The goal of this study is to determine the effects of two deep-release dams on dissolved oxygen and nitrogen species in the Grand and Conestogo Rivers.

The second chapter of this thesis considers the cycling of dissolved oxygen, specifically looking at the concentrations of dissolved oxygen and oxygen isotopes along the river transect and how the riverine oxygen cycle is changed by the reservoir, what distance the river takes to recover, and what roles season and gas exchange play. Also considered is the impact that the construction and operation of the dam has on the reservoir and downstream conditions.

The third chapter is concerned with the changes in nitrogen cycling caused by the reservoirs. The goal will be to determine whether the reservoir is storing the incoming nitrogen, utilizing it or transporting it downstream, and the potential causes of those processes.

The fourth chapter will include final conclusions, what relevance the RCC, SDC and CRCC have in the Grand and Conestogo Rivers, issues regarding the current state of reservoirs, and finally, recommendations for future directions in river and reservoir research.

2.0 River oxygen metabolism and aeration as affected by reservoirs

2.1 Introduction

2.1.1 Dissolved Oxygen Preface

Second only to the water itself, dissolved oxygen is an essential parameter in rivers (Wetzel, 2001) affecting many chemical and biological species; it is considered a key measure of water quality (Chomicki and Schiff, 2008).

Dissolved oxygen concentration (DO) in flowing water may change on a daily basis following a cycle established by photosynthetic primary production, which releases oxygen into the water during the day, and respiration, which consumes DO during both day and night (Odum, 1956). DO is affected by gas exchange at the air-water interface, depending on concentration with respect to saturation, and influx of DO with ground water and surface drainage, though these are usually considered negligible (e.g., Odum, 1956).

2.1.2 Seasonal Dissolved Oxygen Changes

Seasonal changes in DO are driven by temperature, light availability for phototrophs, biomass, etc. (Venkiteswaran *et al*., 2007). These parameters can vary greatly on seasonal time scales. In rivers with reservoirs, the seasonal changes become more complex in that there are two types of ecosystems in which change is taking place, and not necessarily in the same way. The river may have periods of low temperature (perhaps ice cover) and high DO, and at the same time the reservoir could be ice-covered with low DO. Conversely, in warmer seasons, the river could be warm with low oxygen concentrations, while the reservoir could be thermally stratified with DO varying greatly with depth. These are a few of the many possible combinations that could occur. All of these seasonal differences have the potential to alter the river downstream of the reservoir.

2.1.3 Dissolved Oxygen at Hydraulic Structures

Hydraulic structures (dams, weirs, etc.) are common in intensively-used rivers. These structures, with their reservoirs, spillways, weirs and gates, can be important when it comes to gas transfer at the air-water interface (Gulliver and Rindels, 1993). In reservoirs with a short residence time, it may be possible to calculate oxygen exchange with the atmosphere as the water passes either through or over the dam. Concentrations of DO measured above and directly below a dam could be used, as photosynthesis and/or respiration will have little to no effect on DO due to the short amount of time it takes to travel through the structure (Gulliver and Rindels, 1993). Then it can be determined whether the dam is providing re-aeration (Gulliver and Rindels, 1993).

In river-reservoir ecosystems, particularly in temperate regions, the function and management of the dam changes with season, thus potentially changing the effect of the dam on the DO regime. An obstacle when calculating gas exchange is the possibility of thermal stratification in the reservoir (Gulliver and Rindels, 1993). It is then necessary to understand the dam withdrawal mechanisms as well as to have an understanding of the thermal profile behind the structure (Gulliver and Rindels, 1993).

2.1.4 Photosynthesis, Respiration, and Ecosystem Metabolism

The primary production to respiration ratio (P:R) is used as an indicator of carbon utilization and trophic status (del Giorgio and Peters, 1994; Tobias *et al*., 2007). If P:R is greater than 1, an ecosystem is considered to be net autotrophic, fixing more carbon than is respired, whereas a P:R that is less than 1 is considered to indicate an ecosystem that is net heterotrophic (Tobias *et al*., 2007) respiring more carbon than it fixes. Unfortunately, aquatic ecosystems with different P and R rates can have the same ratio, making the utility of P:R as an indicator questionable (Venkiteswaran *et al*., 2007). To understand the extent to which ecosystems are carbon sources or sinks, accurate estimates of P and R are required (Mulholland *et al*., 1997).

Though there are many methods available for determining P and R, none are without problems. The stable isotopic composition of the DO has the potential to aid in resolution of some of the uncertainty (Tobias *et al*., 2007). Fractionation between $^{18}O^{16}O$ and $^{16}O^{16}O$ can occur by physical processes, i.e., thermal diffusion or distillation, or by chemical processes, i.e., equilibrium isotopic exchange, decomposition, or oxidation (Lane and Dole, 1956). There is a 0.7‰ equilibrium fractionation during dissolution, so even though the O_2 found in the atmosphere has a $\delta^{18}O$-O_2 of 23.5‰ (Kroopnick and Craig, 1972), when gas exchange is the primary force, the $\delta^{18}O$-O_2 in water is pushed towards +24.2‰ (Quay *et al*., 1995). If respiration is dominant, the $\delta^{18}O$-O_2 is greater than +24.2‰; conversely, if photosynthesis is dominant the $\delta^{18}O$-O_2 is typically less than +24.2‰ (Quay *et al*., 1995). Thus, a heterotrophic system will have a high $\delta^{18}O$-O_2 whereas a system that is autotrophic will have a lower $\delta^{18}O$-O_2 (Tobias *et al*., 2007). The process of photosynthesis causes little to no fractionation, causing either no change or a slightly more negative $\delta^{18}O$-O_2 relative to water (Stevens *et al*., 1975; Quay *et al*., 1995).

Some of the most common methods for measuring metabolism are the chamber methods and the whole-stream, open-water methods (Marzolf *et al*., 1994). Though chamber methods can be replicated and allow for experimentation, the assumption that what occurs in the chamber is comparable to what occurs in the undisturbed water mass is difficult to verify (Welch, 1968).

If upstream and downstream diurnal DO curves are measured, the production over the reach can be calculated (Odum, 1956). The method takes the diel curve from the two sites, then subtracts the upstream curve from the downstream curve compensating for the time required for flow from the first site to reach the second, shifting the upstream curve to the left (Odum, 1956). Gas exchange must be accounted for. An advantage to the open-water method is it does not disturb the environment and includes all parts of the system (Marzolf *et al*., 1994). In a homogeneous reach, metabolism can be measured at a single station. There are several disadvantages of open-water methods. True replication is impossible, as the method cannot be repeated at the same time and reach (Marzolf *et al*., 1994). Reaches need to be selected so that the only changes that occur with the DO are from in-stream processes, i.e., no tributaries or groundwater inputs. Finally, gas exchange can be a difficult measurement or calculation and therefore can lead to error (Marzolf *et al*., 1994). A more problematic assumption that is frequently made for both open water and chamber methods is that day-time respiration is the same as (temperature corrected) night-time respiration (Welch, 1968).

Quay *et al*. (1995) noted that in high productivity waters, DO tends to deviate from steady state. P:R can be calculated for such rivers using simultaneous chemical and isotopic mass balance equations for O_2 and $\delta^{18}O$-O_2 (Parker *et al*., 2005). The photosynthesis-respiration-gas exchange model (PoRGy) was developed to model both DO and $\delta^{18}O$-O_2 in river systems (Venkiteswaran *et al*., 2007). Its intended use was to quantify P, R and G rates in transient field conditions.

2.1.5 Impacts of Dams on River Metabolism

Most open-water P:R calculation methods involve an assumption of steady state. Rivers can have many attributes that would violate that assumption; among those would be the presence of a dam, as it would disrupt the natural downstream flow.

In Brazil, 90% of the electricity consumed is provided by hydroelectric plants associated with reservoirs (de Oliveira Naliato *et al*., 2009). A study conducted by de Oliveira Naliato *et al*. (2009) considered two large reservoirs (in a cascade of 11) on the Paranapanema River to assess the downstream effects of hydroelectric dams. Measurements were done diurnally, at depth, seasonally, and during pulse flow events. They found that the released water varied throughout the year. Downstream flow could either fluctuate rapidly or be unnaturally stable. A difference of up to 9°C was measured between the surface and hypolimnion in the reservoirs, accompanied by a 4 mg/L difference in DO, causing low DO to be measured downstream. They observed stratification and eutrophic conditions in the reservoirs, concluding that a dam with a deep release mechanism altered the downstream environment (de Oliveira Naliato *et al*., 2009). In particular, they showed that the downstream reaches had temperature and DO that were similar to the measurements taken at depth from the reservoir (de Oliveira Naliato *et al*., 2009).

In Tanzania, along the Kihansi River, Ideva *et al*. (2008) had the unique opportunity to take pre-dam construction measurements. From their observations post-construction, the temperatures within the reservoir increased, leading to an increase in temperature downstream. Along with increased variability in DO, the overall trend was of lower concentrations both within the reservoir and downstream (Ideva *et al*., 2008). The lower average DO was attributed to the decomposition of organic matter within the newly flooded area (Ideva *et al*., 2008).

2.1.6 A Day in the Life of the Grand River (DLG)

In the summer of 2007, two longitudinal samplings of the Grand River were done (Venkiteswaran *et al*., 2015) at 23 sites from the headwaters to the river mouth (DLG1, June 14 and DLG2, September 5). Multiple teams were sent out so that all sites could be sampled pre-dawn, mid-morning and early afternoon. At these times, numerous parameters were measured, but for the current purpose only temperature (Figure 2.1A) and oxygen concentrations (Figure 2.1B) will be considered. All samples were taken as close to mid-stream as possible, and all were taken at a depth just below the surface. The results in the area below the Shand dam were of particular interest. There was a large temperature difference above and below the dam in the first excursion, but little difference in the second. The DO were also different in that the first DLG found an increase below the dam, while the second found a decrease. This led to further to study these phenomena (DO and temperature).

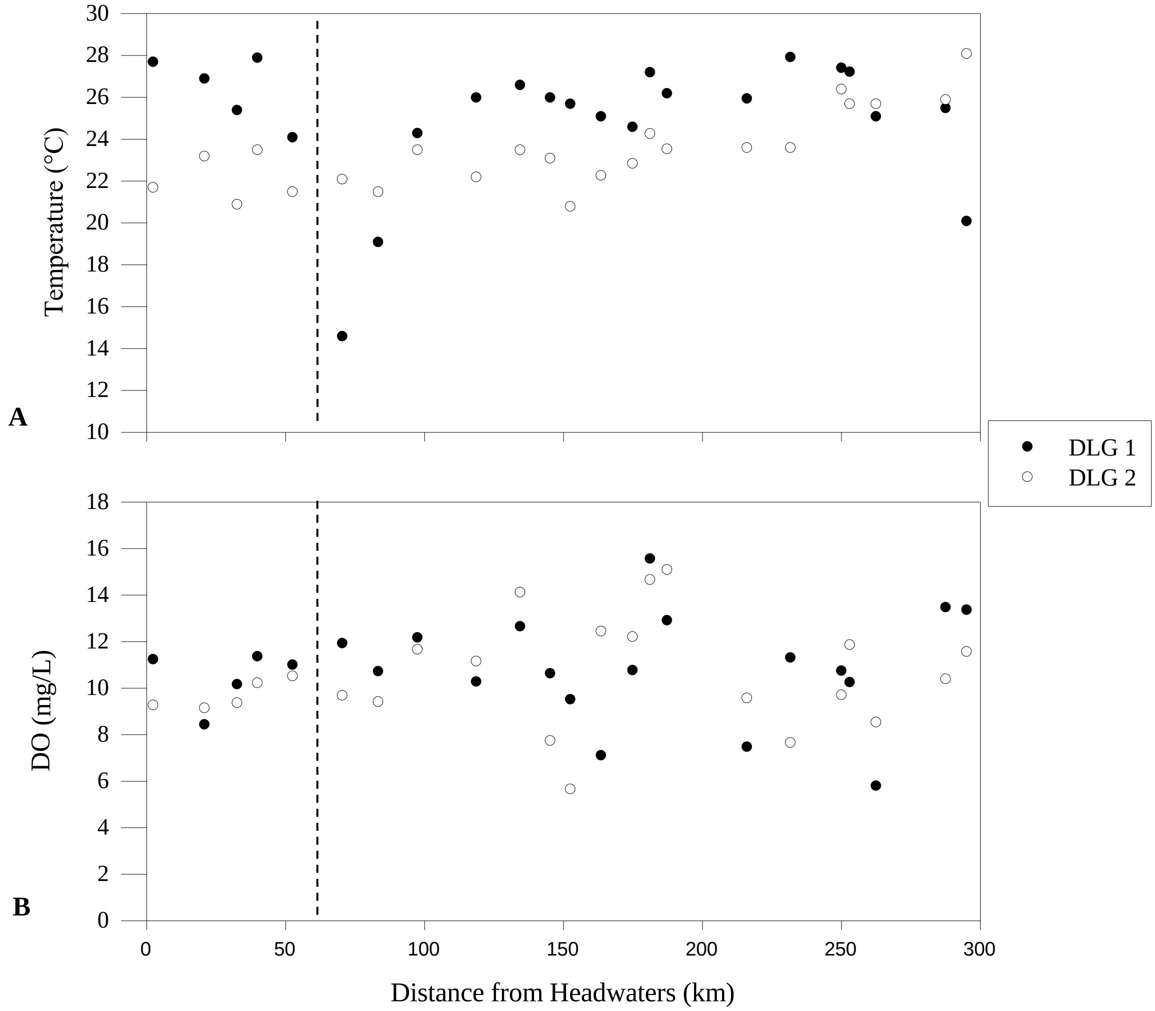

Figure 2.1: Temperature (°C) and dissolved oxygen (mg/L) from the Grand River June 14, 2007 (DLG1) and September 5, 2007 (DLG2) early afternoon samplings (Venkiteswaran *et al*., 2015) The Shand Dam is indicated by the dashed line.

2.1.7 This Study

The Day in the Life of the Grand study has shown that the Grand River has naturally occurring oxygen cycles that are potentially being disrupted by the presence of reservoirs and dams. As previously mentioned, DO distribution in reservoirs is extremely variable (Wetzel, 2001). The Belwood and Conestogo reservoirs are known to thermally stratify and are bottom-draw. When considering reservoirs and their downstream effects, the depth from which water is released has a large impact, especially during stratification periods (de Oliveira Naliato *et al*., 2009).

The following conceptual model (Figure 2.2) attempts to describe the changes that might occur in the daily DO cycle in the context of a river-reservoir-river system. It illustrates the amplitude of the undisturbed upstream riverine cycle, the smaller lake-like cycle within the reservoir, the muted cycle that occurs below the dam, and lastly the recovering system further downstream with a cycle amplitude approaching that of the original upstream reach. The upper and lower constraints for DO in the upstream and downstream river were chosen based on measurements taken during the DLGs. The effects on sites directly below the reservoirs are based on the discontinuity distance concepts (Ward and Stanford, 1983), and downstream field measurements from the 2008 preliminary sampling.

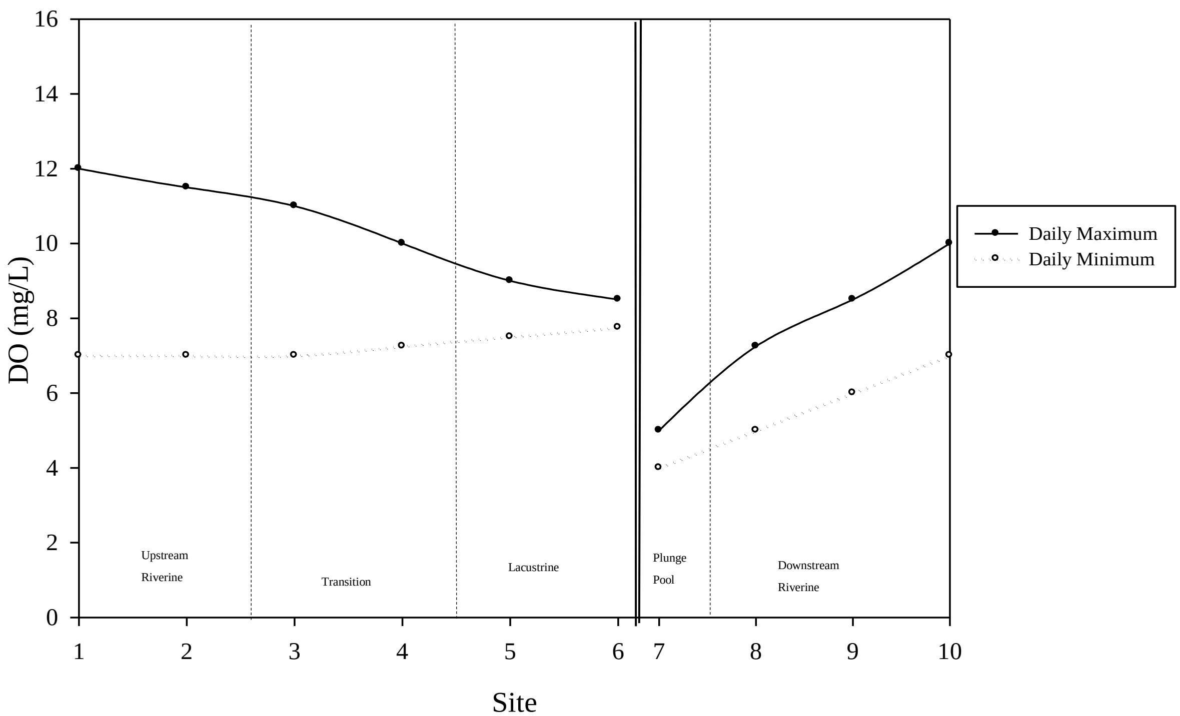

Figure 2.2: A conceptual model designed to illustrate the DO cycle at the surface waters throughout the reservoir ecosystem including the bottom-draw dam structure and the changes it causes in the plunge pool. The difference between the theoretical daily maximum and minimum represents the amplitude of the DO cycle. The graph is further divided into the traditional reservoir zones as described by Thornton et al. (1990), with the lacustrine representing the surface waters.

2.1.8 Chapter Goals

This chapter will examine the changes in DO and $\delta^{18}O$-O_2 between sites upstream and downstream of dams on both daily and seasonal time scales. As well, I will attempt to determine if and how stratification, water residence time and flow affect DO concentrations and cycling. An attempt at determining the implications for P, R, and G will be made to examine dam effects on river metabolism, ecosystem health and system recovery. Finally, I will determine if the dam structure is providing re-aeration, what the efficiency is, and what are the downstream implications.

2.2 Methods

2.2.1 Sampling Procedures

Study sites were selected at Belwood Lake that would best represent the previously mentioned reservoir zones: riverine, transition, and lacustrine. There were ten sites in total: two in the upstream riverine, two in the transition, two in the lacustrine, one directly below the dam and three in the downstream riverine. Sites 1 and 2 are in the upstream riverine at 13 km and 11.5 km from the dam. Sites 3 and 4 are located in the transition zone at 8.5 km and 6.5 km from the dam. The lacustrine zone contains site 5 at 1.5 km from the dam and site 6 is 30m in front of the dam. Located in the plunge pool directly below the dam is site 7. Sites 8, 9, and 10 are further downstream at 250 m, 1.5 km and 4 km below the dam.

Similar to Belwood Lake, site selection at Conestogo Lake was also aimed at the three main reservoir zones. Sites 1 and 2 are located in the upstream riverine section at 22.5 km and 11.5 km from the dam. Sites 3 and 4 are both located in the transition zone, but not consecutively. Site 3 represents the Mallet River which flows into the Conestogo in the transition zone. The distance from the dam at sites 3 and 4 is 8.5 km. Sites 5 and 6 represent the lacustrine zone. Site 5 is 4.5 km from the dam and site 6 is 30 m from the dam. Site 7 is in the plunge pool directly below the dam. Finally, sites 8 and 9 are the downstream riverine sites, located at 250 m and 2.6 km below the dam.

Sampling took place over a three year period. In 2008, preliminary sampling occurred at Belwood Lake in July and Conestogo Lake in August. Both lakes were sampled again in September and October. In 2009, sampling took place at both reservoirs in May, June, August and November; only Belwood Lake was sampled in July. The final sampling was at Belwood Lake in August of 2010.

Fewer sites were sampled in 2008. At Conestogo Lake there were five, and these were sites 3,4,6,7 and 8 of the eventual nine sites described above. At Belwood Lake only sites 6, 7 and 8 were measured in 2008. In 2009, nine sites were sampled at both reservoirs. An additional site (10) was added to Belwood Lake in 2010. Seasonal samples in both 2008 and 2009 were taken at about 20 cm below the surface, as close to main channel flow as possible. The surface water from site 6 (in both of the reservoirs) was obtained either from the boat launch or from the middle of the lake near the dam using a canoe.

Diel Sampling

Sampling trips to collect time series data (diels) were conducted at both Conestogo Lake (May 22, 2009) and Belwood Lake (May 23, 2009). At Conestogo Lake, only 6 of the 9 sites were sampled due to time constraints. The sites were 1, 2, 6, 7, 8, and 9 (Section 2.2.1). At Belwood Lake, 5 of the 10 sites were sampled, again, due to time constraints. The sites were 1, 6, 7, 8, and 9 (Section 2.2.1). The diels in 2009 were done as three time-points; pre-dawn, morning and afternoon. Samples were taken at about 20 cm below the surface and as close to main channel flow as possible with the exception of the samples taken from within the reservoirs, which were taken at the boat launches.

A third diel sampling took place on August 10, 2010. It was 24 h in duration and was done only at Belwood Lake. There were six time points total: morning, afternoon, evening, dusk, midnight, and pre-dawn. For this sampling trip, six sites were sampled (sites 1, 4, 7, 8, 9, and 10). The major difference between the first diel completed at Belwood Lake in May of 2009 and this second sampling was the removal of the sixth site from within the reservoir and the addition of a site further downstream towards the village of Fergus (site 10). For this diel, samples were taken at about 20 cm below the surface and as close to main channel flow as possible.

Lake Profiles

In August and September, 2008, two lake profiles were done at Conestogo Lake. They were both conducted at site 6, within the reservoir about 25 m from the dam. The profiles were completed using pre-marked Tygon® tubing with a weight at the end; this was connected to a flow cell in which the oxygen probe attached. The water was moved by a peristaltic pump. On the first of the sampling trips, samples were taken at 20 cm, 1 m, 2 m, 3 m, 4 m, 5 m, 6 m, 7 m, 8 m, 9 m, 10 m and 15 m. For the second trip, the sampled depths were 20 cm, 0.5 m, 1 m, 1.5 m, 3 m, 5 m, 8 m, 11.5 m, 13 m, and 14 m. Also in September 2008, a small profile was done at Belwood Lake. The surface, 3 m, 8 m, 11 m, and 12 m were sampled.

In June of 2009, both sites 5 and 6 at Belwood Lake were sampled for profiles. Site 5 was sampled at 20 cm and 4 m, and site 6 was sampled at 20 cm, 7 m and 14 m.

2.2.2 Field and Laboratory Analysis

Oxygen concentration

For 2008 the DO and temperature were measured using a YSI multimetre probe; in 2009 and 2010 the DO was measured using duplicate Winkler titrations (Carpenter, 1965). Water for titrations was sampled and analyzed in 500-mL glass BOD bottles with 2 mL each of sodium azide (NaN_3) and manganese sulphate ($Mn(SO_4)_2$). In the laboratory, 2 mL of sulfuric acid was used to acidify the sample. The sample was then titrated with sodium thiosulfate. Starch was used as an indicator for the titration. Precision is estimated at ±0.2 mg/L.

Oxygen isotope measurements

Duplicate $\delta^{18}O\text{-}O_2$ samples were also taken at each sampling. These were collected into 160-mL glass bottles that had been acid washed and rinsed. Each had 0.4 g of NaN_3 and was evacuated through a butyl rubber stopper. The bottles were carefully filled under water using 21-gauge needles in order to ensure that no air entered the bottle. For analysis, a head space was added to the bottles by removing 5 mL of water, and simultaneously injecting of 5 mL of helium. Before analysis the samples were shaken for 90 minutes at 110 rpm; analysis was completed using a Micromass Isochrom μG mass spectrometer (Wassenar and Koehler, 1999). Precision is estimated at ± 0.2‰.

2.2.3 Data Analysis

Metabolism estimates from field measurements

Aeration coefficients $\left(\frac{G}{Z}\right)$ were determined using the night time regression of the measured changes in oxygen over time for the August 2010 sampling, using the methods described by Young *et al*. (2004), Parker *et al.* (2005) and Grace and Imberger (2006). By first calculating the gas exchange coefficient then dividing by the depth the aeration coefficient $\left(\frac{G}{Z}\right)$ can be found and is expressed in per time units (t^{-1}) (Hemond and Fechner, 2015).

For the analysis of sites 1, 4, and 10 methods from Parker *et al*. (2005) were used:

$$\frac{d[O_2]}{dt} = \left(\frac{G}{Z}\right)([O_2]_s - [O_2]_w) - R + P + A \qquad \textbf{Equation 2.1}$$

Where $\frac{d[O_2]}{dt}$ is the rate of oxygen change with time, G is the gas exchange coefficient, Z is the depth, $[O_2]_s$ is the DO at saturation, $[O_2]_w$ is the measured DO of the water, R is respiration, P is gross primary production and A is the accrual of DO from ground and surface water inputs. The night time respiration can be determined assuming P and A are both 0 and that temperature is not changing.

$$R = \left(\frac{G}{Z}\right)([O_2]_s - [O_2]_w) - \frac{d[O_2]}{dt} \qquad \textbf{Equation 2.2}$$

From this, the aeration coefficient $\frac{G}{Z}$ can be found as the slope from plotting $\frac{d[O_2]}{dt}$ vs. $([O_2]_s - [O_2]_w)$. By substituting the value of $\frac{G}{Z}$ back into Equation 2.2 the respiration can be found by assuming R is constant. Then by rearranging Equation 2.1 for P, and still assuming A = 0 (no other inputs), P can be found. To determine P by interval, it is multiplied by *dt*. Addition of these values would give the estimate over 24 h.

Metabolism estimates from model estimates

Gas exchange coefficients (G) for the Grand River have been previously estimated from model calculations (Venkiteswaran *et al*., 2015). These values will be used to calculate P_{net} (Parker, 2005):

$$P - R = P_{net} = \frac{d[O_2]}{dt} - \left(\frac{G}{Z}\right)([O_2]_s - [O_2]_w) \qquad \textbf{Equation 2.3}$$

Dissolved oxygen across a hydraulic structure

The impact of the dam structure on dissolved oxygen was determined by a calculation developed by Gameson (1957) and expanded upon by Baylar *et al*. (2010).

$$r = \frac{(C_s - C_a)}{(C_s - C_b)} \qquad \textbf{Equation 2.4}$$

Where r is the oxygen deficit ratio, C_s is the DO saturation concentration at time t, C_a is the DO above the dam and C_b is the DO below the dam. Using the oxygen deficit ratio the aeration (or transfer) efficiency (E, unitless) for a hydraulic structure can be found (Gulliver *et al*., 1993; Baylar *et al*., 2010).

$$E = 1 - \frac{1}{r} = \frac{(C_b - C_a)}{(C_s - C_a)} \qquad \textbf{Equation 2.5}$$

It is assumed that C_s remains constant on either side to the structure. If E > 1, then the downstream will be supersaturated (e.g. $C_b > C_s$); if E = 1 then the downstream will be at saturation, finally if E = 0 then there is no appreciable amount of gas exchange being provided by the dam structure (Baylar *et al*., 2010).

2.3 Results

2.3.1 Seasonal analysis, 2008

Reservoir Conditions and Precipitation

The 2008 sampling season had a high amount of precipitation, particularly from July to early September (GRCA, 2008). The reservoirs remained at their higher buffer volume or slightly above between June and October; this was similar for both Belwood and Conestogo (GRCA, 2008).

Temperature

During the summer, Belwood Lake (Figure 2.3A) and Conestogo Lake (Figure 2.4A) had higher surface temperatures than what were measured downstream; Conestogo's upstream sites were similar to its downstream. In September, the reservoirs had temperatures similar to each other and downstream, though Conestogo's upstream sites were much lower than all others (~5°C). The late fall sampling followed similar patterns to September, but with overall lower temperatures.

Dissolved Oxygen

The DO at both reservoirs (upstream, in reservoir, downstream) was lower in the summer and early fall than in late fall. All of the sampling trips for both Belwood Lake (Figure 2.3B) and Conestogo Lake (Figure 2.4B) had lower DO in the plunge pool than at the reservoir surface, except for Belwood Lake in September. Conestogo Lake had the largest range in seasonal DO concentrations below and in the reservoir from 4.5 mg/L to 17.6 mg/L compared to

Belwood Lake which ranged from 6.7 mg/L to 15.8 mg/L. The highest DO concentrations were consistently measured in the upstream riverine at Conestogo Lake; though lower concentrations were measured in the reservoir.

$\delta^{18}O\text{-}O_2$

For both summer and early fall, Belwood Lake (Figure 2.3D) showed high $\delta^{18}O\text{-}O_2$ downstream of the reservoir, with lower $\delta^{18}O\text{-}O_2$ at the reservoir surface. Conestogo Lake (Figure 2.4D) had a similar pattern, but with overall lower values.

Upstream of Conestogo Lake differences were observed between site 3 ('M') and site 4. The differences in the $\delta^{18}O\text{-}O_2$ also occurred between samplings. These differences were not observed in either the temperature or DO measurements.

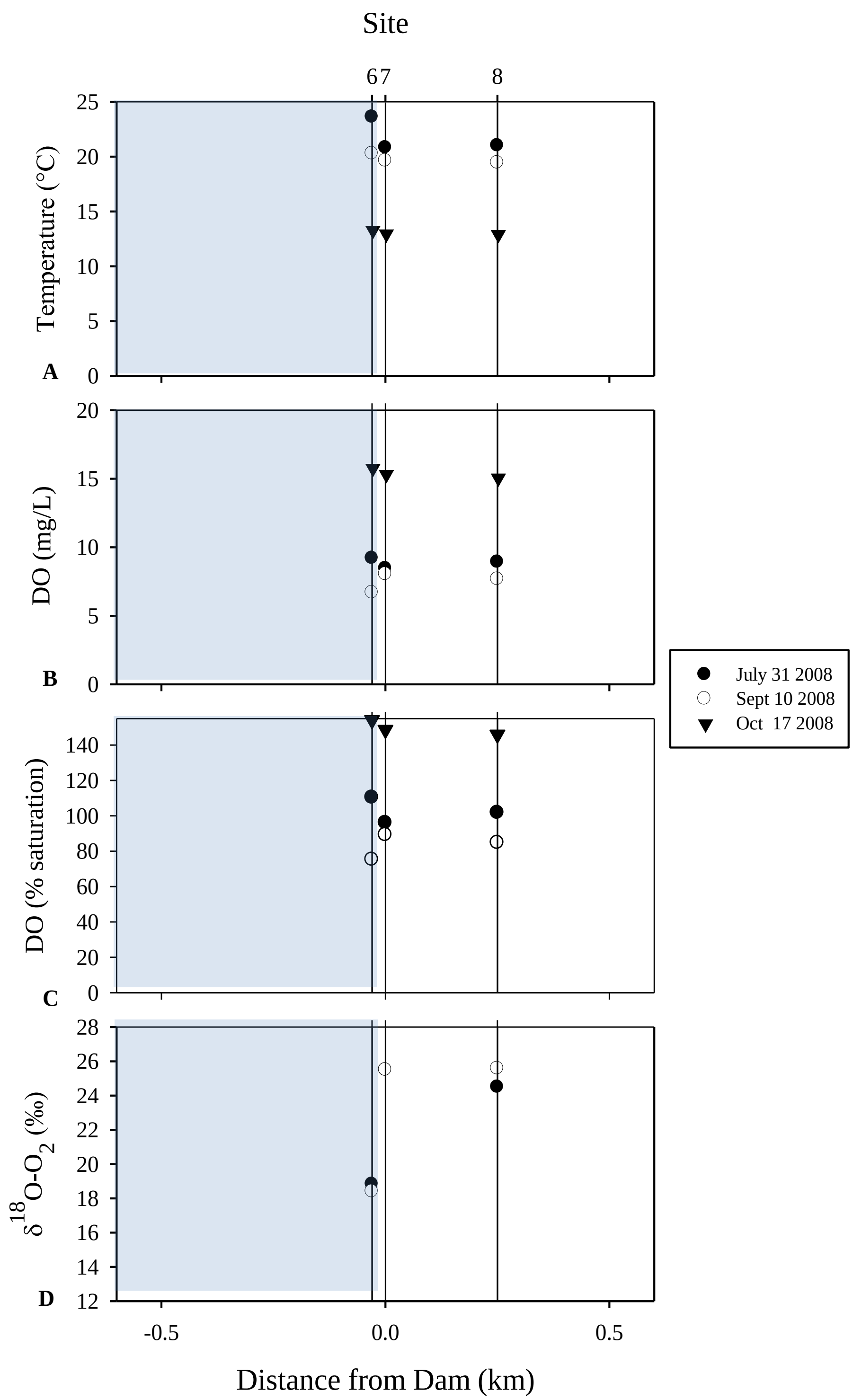

Figure 2.3: A) temperature (°C), B) dissolved oxygen (mg/L), C) dissolved oxygen (% saturation); and D) $\delta^{18}O$-O_2 (‰) from Belwood Lake for the 2008 preliminary sampling season. Sampling times were: July 31, 13:45-14:45; September 10, 17:30 – 19:45; and October 17, 7:45 – 8:45. The Shand Dam is located between sites 6 and 7. The reservoir is shaded.

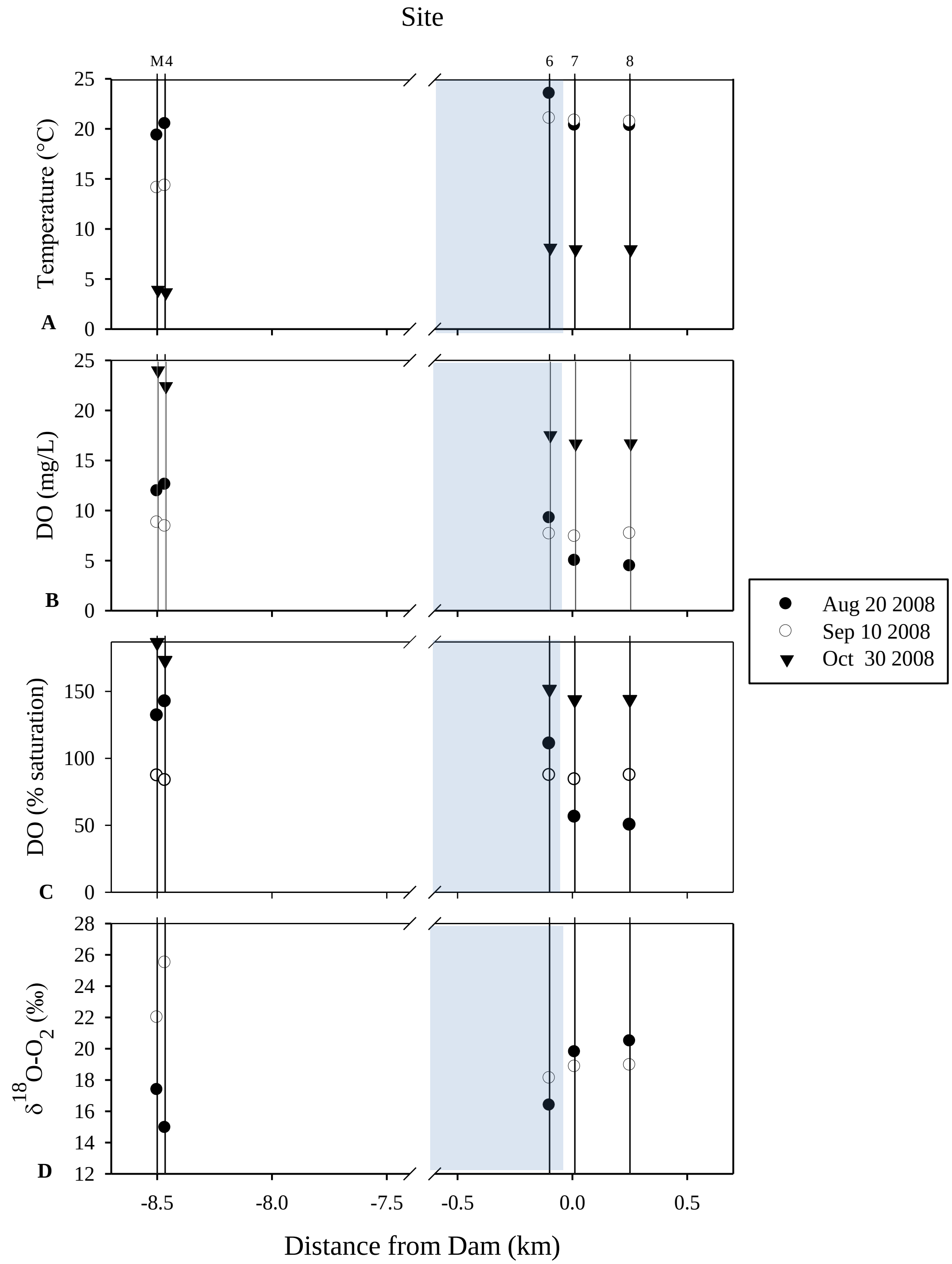

Figure 2.4: A) temperature (°C) B) dissolved oxygen (mg/L), C) dissolved oxygen (% saturation); and D) $\delta^{18}O-O_2$ (‰) from Conestogo Lake for the 2008 preliminary sampling season. Sampling times were: August 20, 11:45 – 13:30; September 10, 9:45 – 12:00; and October 30, 10:00 – 12:15. The Conestogo dam is located between sites 6 and 7. The lacustrine zone of the reservoir is shaded; sites 3 and 4 represent the transition zone.

2.3.2 Vertical profiles in the reservoirs, 2008

Temperature

The profiles taken on September 10 at both Belwood Lake and Conestogo Lake (Figure 2.5A) exhibited little thermal change from surface to depth. On August 20, Conestogo Lake showed a weak temperature gradient around the 9 m depth but the overall surface to depth temperature change was only ~3.5°C.

Dissolved Oxygen

Similar to the temperature profiles in September, the DO profiles at both reservoirs (Figures 2.5B and 2.5C) showed very little change. The summer profile at Conestogo Lake showed two areas of DO decrease: the first between 0-4 m, the second below 8 m, with a total DO change of 5.8 mg/L. In September, both reservoirs had surface saturations under 100%. Conestogo in August was above 100% saturation between 0-1 m.

$\delta^{18}O$-O_2

There was a slight increase in $\delta^{18}O$-O_2 with depth on September 10 at both reservoirs, but only a small overall change (Belwood Lake <1‰, Conestogo Lake <3‰). The August 20 profile at Conestogo Lake had a more pronounced increase with depth, showing a change in $\delta^{18}O$-O_2 of ~ 10‰.

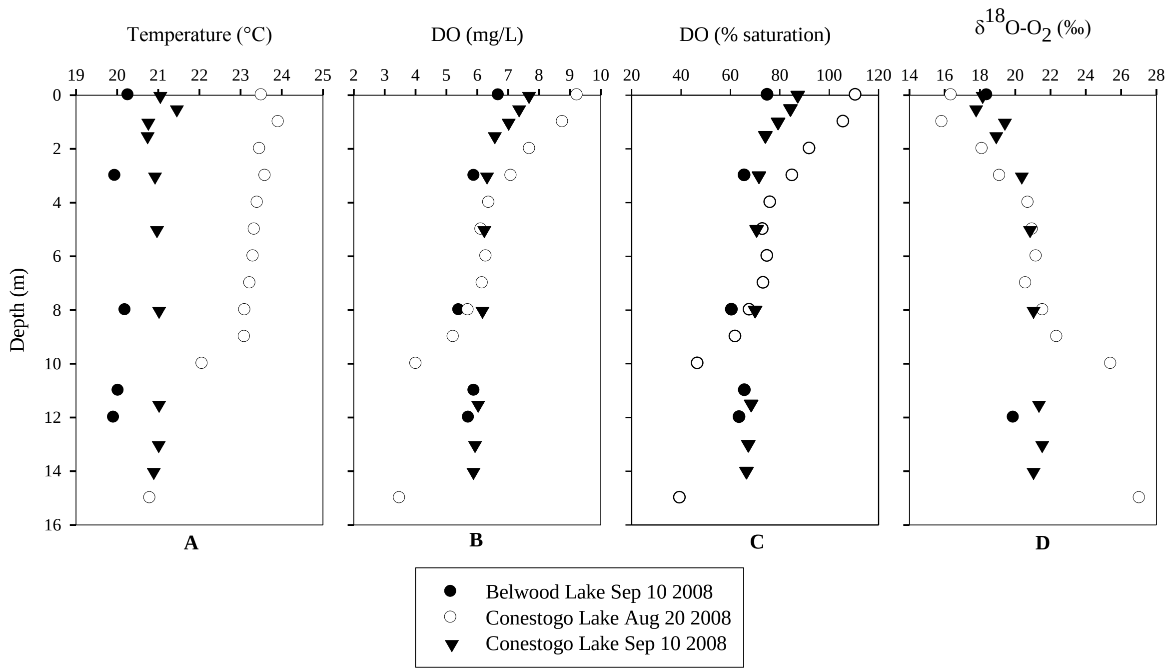

Figure 2.5: Depth profiles from Belwood Lake, September 10 2008, Conestogo Lake, August 20 2008 and Conestogo Lake, September 10 2008. With A) temperature (°C); B) dissolved oxygen (mg/L), C) dissolved oxygen (% saturation); and D) $\delta^{18}O\text{-}O_2$ (‰). Sampling times were 17:30 – 19:00 for Belwood on September 10; 13:30 – 16:15 for Conestogo on August 20; and 12:30 – 16:00 for Conestogo on September 10.

2.3.3 Seasonal Analysis, 2009

Reservoir Conditions and Precipitation

Average rainfall was recorded for the 2009 sampling season, with above average in April, May and August (GRCA, 2009). Both Belwood and Conestogo reservoir volumes remained at or above their upper buffer between May and July, and above from September to November (GRCA, 2009).

Temperature

There was a large temperature range over the sampling season; Belwood Lake from 3.6°C to 28.2°C (Figure 2.6A) and Conestogo Lake from 5.0°C to 23.2°C (Figure 2.7A). In May and June at Conestogo, and additionally in July at Belwood, water was warm upstream and within the reservoirs with cooler plunge pools. The largest change in temperatures seen at either of the reservoirs was in June. From the surface of Belwood Lake to its plunge pool there was a temperature change of 10.3°C (24.7°C - 14.4°C); similarly, at Conestogo Lake there was a change of 5.1°C (19.6°C – 14.5°C). Despite the change from the reservoir to the plunge pool, the temperature of the Conestogo River, by the time it reached site 9, was warming back up (an increase of 3.5°C).

In August, temperatures were generally high for all sites at both reservoirs. This includes the sites downstream from the dams, where only small changes in temperature were observed (~2°C from surface to plunge pool).

The sampling in November for both reservoirs had the lowest overall temperatures, with Belwood Lake slightly lower than Conestogo Lake. Both Lakes had consistent upstream temperatures with increases within the reservoir which carried on through to downstream of the dams.

Dissolved Oxygen

The DO measured at Belwood Lake and Conestogo Lake varied through the 2009 season. Belwood had concentrations ranging from 5.4 mg/L to 17.0 mg/L (Figure 2.6B); Conestogo was anywhere from 6.8 mg/L to 18.4 mg/L (Figure 2.7B).

In May, there was higher DO upstream of both reservoirs than was found within them, but the downstream trends differed. Conestogo had low downstream DO concentrations until recovery to 12.9 mg/L at site 9. Conversely, Belwood had much higher concentrations of DO immediately downstream of the dam, followed by a decrease at site 8 and finally an increase at site 9 (12.3 mg/L, 10.3 mg/L and 11.3 mg/L).

For June 16, Conestogo experienced lower DO (8.5 mg/L – 9.5 mg/L) entering the reservoir, whereas on June 23 Belwood had much higher concentrations in the upstream (17.05 mg/L). Though Belwood reservoir DO concentrations were lower than the inflow, they were still much higher than that found in Conestogo Lake. Downstream of Belwood DO was similar to that found in the reservoir. Conestogo's downstream had a large increase in DO (18.4 mg/L).

Three sampling trips were executed in August; two to Belwood (August 12 and 27) and one to Conestogo (August 25). Both Belwood trips found higher DO entering and within the reservoir, and much lower below the dam and downstream. Conestogo's August sampling was more varied over the longitudinal reach but with a smaller overall range (7.7 mg/L to 11.0 mg/L)

In November, at both lakes and at all sites (upstream, reservoir and downstream), the range of DO concentrations was only 10.2 mg/L to 12.5 mg/L.

$\delta^{18}O$-O_2

Though both lakes saw large ranges of $\delta^{18}O$-O_2 in 2009, Conestogo's was greater, from 11.5‰ to 24.6‰ (Figure 2.7D). Belwood had a range of 14.3‰ to 23.8‰ (Figure 2.6D).

Upstream reaches for both reservoirs in May had lower $\delta^{18}O$-O_2 than was measured in the reservoir. Downstream of Belwood Lake differed from that at Conestogo Lake, as there was a decrease in $\delta^{18}O$-O_2 from the surface to the plunge pool. Downstream of Belwood had increases further downstream, whereas Conestogo was consistent from the reservoir (site 6) to site 8, but had a large decrease (10‰) at site 9. The pattern displayed at Conestogo during the May sampling is similar to that seen at Belwood during the June 23, July 07 and August 27 sampling, with the only difference of a small increase in $\delta^{18}O$-O_2 at site 8 in July.

Conestogo in August showed some small changes between upstream, the reservoir and downstream, the largest at site 9. The $\delta^{18}O$-O_2 at Belwood Lake on August 12 was high and did not show large longitudinal variation; this was similar to what was seen at Conestogo on November 04.

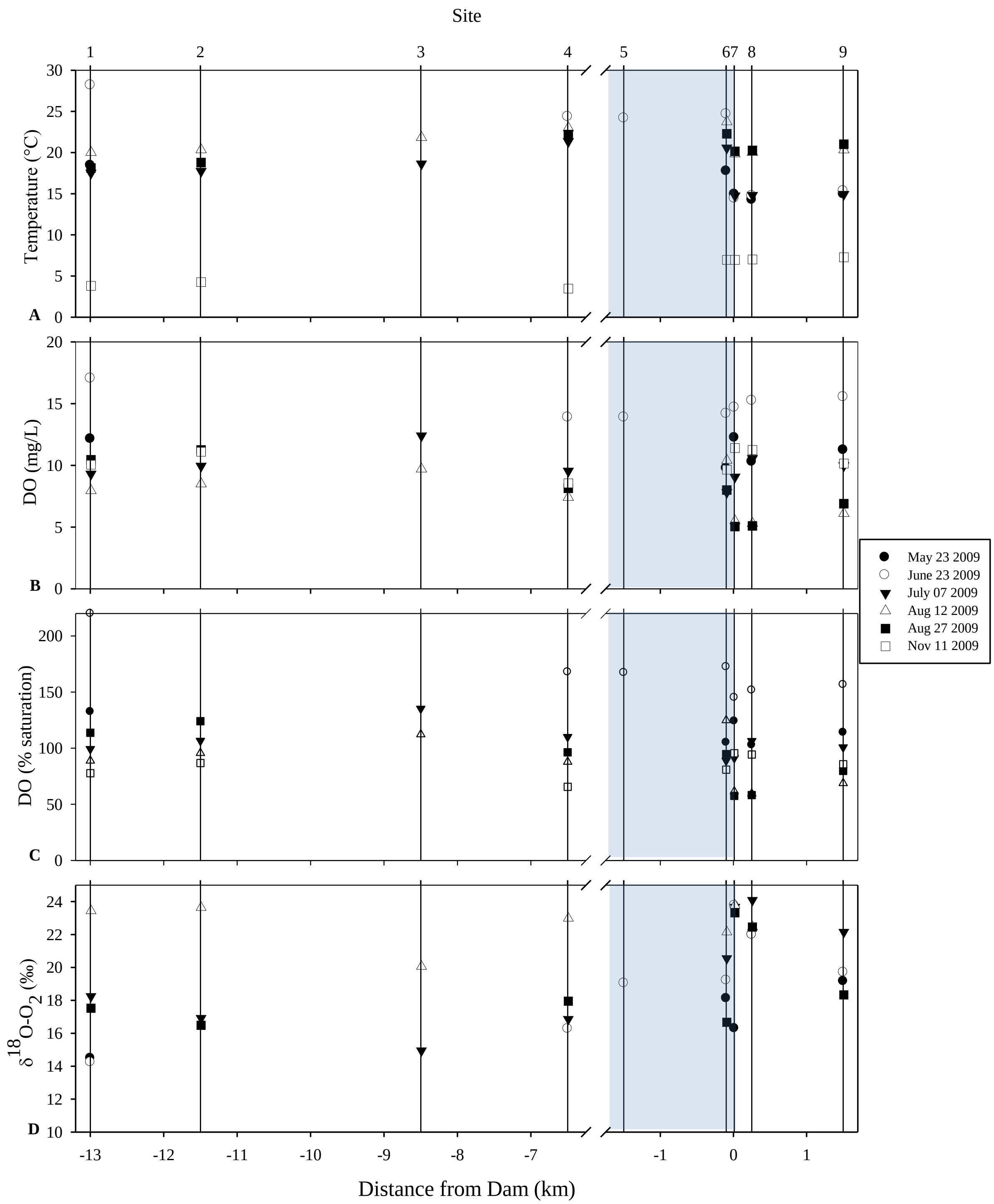

Figure 2.6: A) temperature (°C); B) dissolved oxygen (mg/L); C) dissolved oxygen (% saturation); and D) $\delta^{18}O$-O_2 (‰) from Belwood Lake, 2009. Sampling times were 11:30 – 13:50 (May 23), 10:00 – 16:00 (June 23), 12:00 – 18:30 (July 7), 11:00 – 17:00 (August 12), 11:15 – 13:45 (August 27), and 9:45 – 12:30 (November 11). The Shand dam is located between sites 6 and 7. The lacustrine zone of the reservoir is shaded; sites 3 and 4 represent the transition zone.

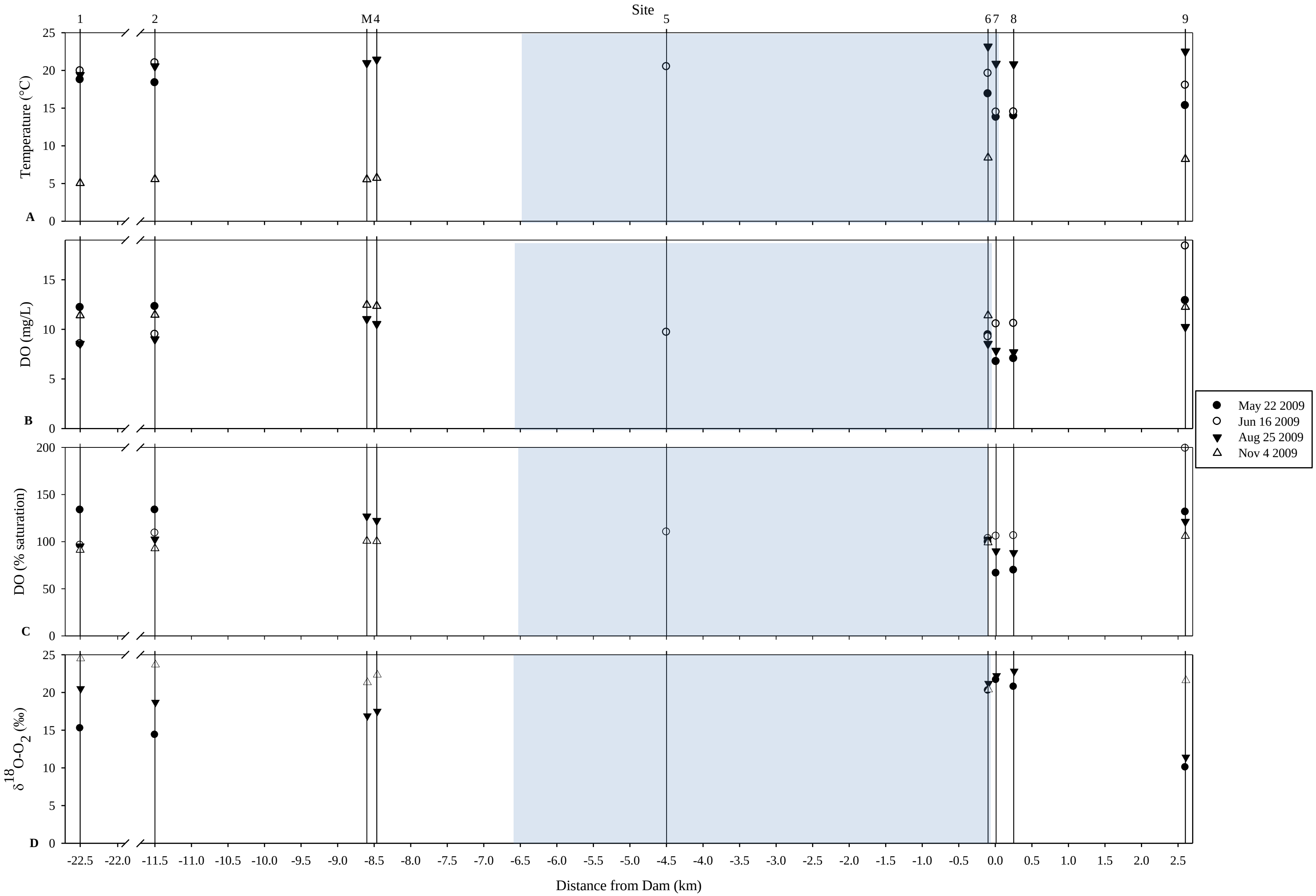

Figure 2.7: A) temperature (°C); B) dissolved oxygen (mg/L), C) dissolved oxygen (% saturation); and D) $\delta^{18}O$-O_2 (‰) from Conestogo Lake, 2009. Sampling times were 11:40 – 13:45 (May 22), 11:00 – 18:00 (June 16), 11:00 – 15:00 (August 25) and 9:45 – 12:30 (November 4). The Conestogo dam is located between sites 6 and 7, the lacustrine zone is shaded.

2.3.4 Vertical profiles in the reservoirs, 2009

Temperature

At Belwood Lake on June 23, 2009 the surface of the reservoir at both sites was warm, at 24°C (Figure 2.8A). The shallower of the sites decreased to 20°C at 4 m. The deeper site indicated weak thermal stratification by decreasing to 17.9°C at 7 m, then to 15.8°C at the bottom.

Dissolved Oxygen

DO measured on the surface at both sites was high ~14 mg/L (Figures 2.8B and 2.8C). There was <0.5 mg/L change at 4m for the shallow site. Site 6 had small changes through the water column, totaling only 4 mg/L from surface to 14 m.

$\delta^{18}O$-O_2

Both sites had similar surface $\delta^{18}O$-O_2 measurements around 19‰ (Figure 2.8D). An increase of only 2‰ occurred at site 5 at the 4 m depth. Values at the bottom of the reservoir (site 6) increased to 26.4‰.

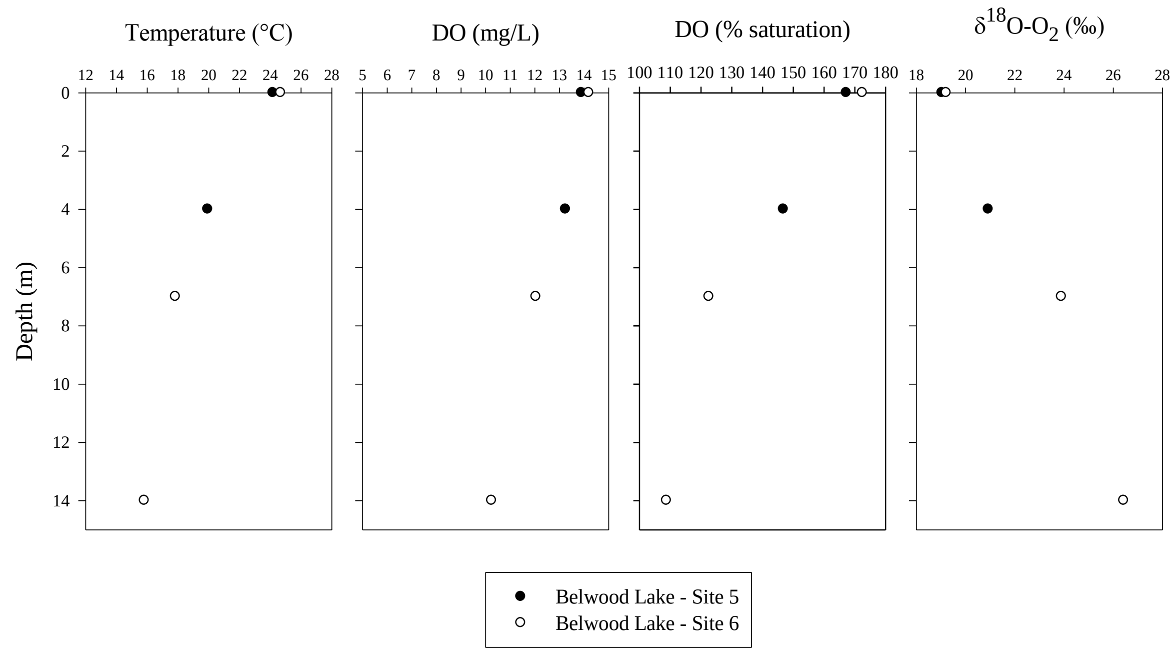

Figure 2.8: Profiles from June 23, 2009 at sites 5 and 6 at Belwood Lake for A) temperature (°C); B) dissolved oxygen (mg/L), C) dissolved oxygen (% saturation); and D) $\delta^{18}O\text{-}O_2$ (‰). Samples were taken between 14:00 – 15:30.

2.3.5 2009 Diel Analysis

Reservoir and Weather Conditions

The month leading up to the diels at both Belwood Lake and Conestogo Lake had higher than average precipitation, in addition to lower than average air temperatures (GRCA, 2009). The volumes of the reservoirs were at or near the summer upper boundary at the time of sampling (GRCA, 2009).

Temperature

The upstream riverine sites for both Belwood Lake (Figure 2.9A) and Conestogo Lake (Figure 2.10A) had temperatures increase throughout the sampling day. Within the reservoirs the temperature was higher in the mornings than upstream, but increased only slightly through the day (~ 2°C in Conestogo Lake, and ~2.5°C in Belwood Lake). The temperature in the plunge pools (site 7) and downstream (site 8) were very similar at both reservoirs. Each was much lower than what was measured at the reservoir surface above (up to 5°C); and both exhibiting almost no change over the day, remaining around 14°C. The furthest site downstream at Belwood was almost the same as its two upstream sites – with little evidence of longitudinal change. Conversely, site 9 at Conestogo had a much greater daily temperature change, though it was still cooler compared to the above reservoir sites.

Dissolved Oxygen

DO changes were different at the different sites during the diels. The upstream sites at both reservoirs had increases over the day (4.5 mg/L), while reaching well over saturation. Within the reservoirs there was very little change observed. Neither reservoir changed more than 1 mg/L. Belwood Lake (Figures 2.9B and 2.9C) did reach and exceed 100% saturation but Conestogo Lake (Figures 2.10B and 2.10C) did not.

Downstream of Conestogo Lake was much lower DO than what was measured at the reservoir surface. Only a small increase in DO occurred throughout the day; this was true for both Site 7 and Site 8. Conversely, site 7 at Belwood Lake was about the same as the above reservoir for the T1 sampling. The DO for T2 and T3 at site 7 were both measurements higher than what was recorded at the reservoir surface. For Site 8 there was a DO decrease (compared to site 7) and almost no overall daily increase.

The furthest downstream sites were also different. At Conestogo Lake, site 9 had both the lowest concentration (5.9 mg/L), but also the largest increase and concentration (T3: 15.2 mg/L) over the diel period. For Belwood Lake, site 9 had a small DO range similar to upstream with a small spike for the T2 sampling.

$\delta^{18}O$-O_2

The upstream sites for both reservoirs had similar $\delta^{18}O$-O_2 measurements and changes; starting high and then decreasing throughout the day. Neither of the reservoirs changed much throughout the day (<2‰), though Belwood Lake (Figure 2.9D) was slightly lower than Conestogo Lake (Figure 2.10D).

Downstream of Belwood showed the $\delta^{18}O-O_2$ decreasing from T1 to T2, yet increasing slightly for T3. This was true for all three sites. At Conestogo, site 7 and site 8 decreased through the day (~5‰), resulting in values similar to the reservoir above. Site 9, similar to its pattern of DO, showed the greatest range: from T1 at 28.1‰ to T3 at 8.6‰.

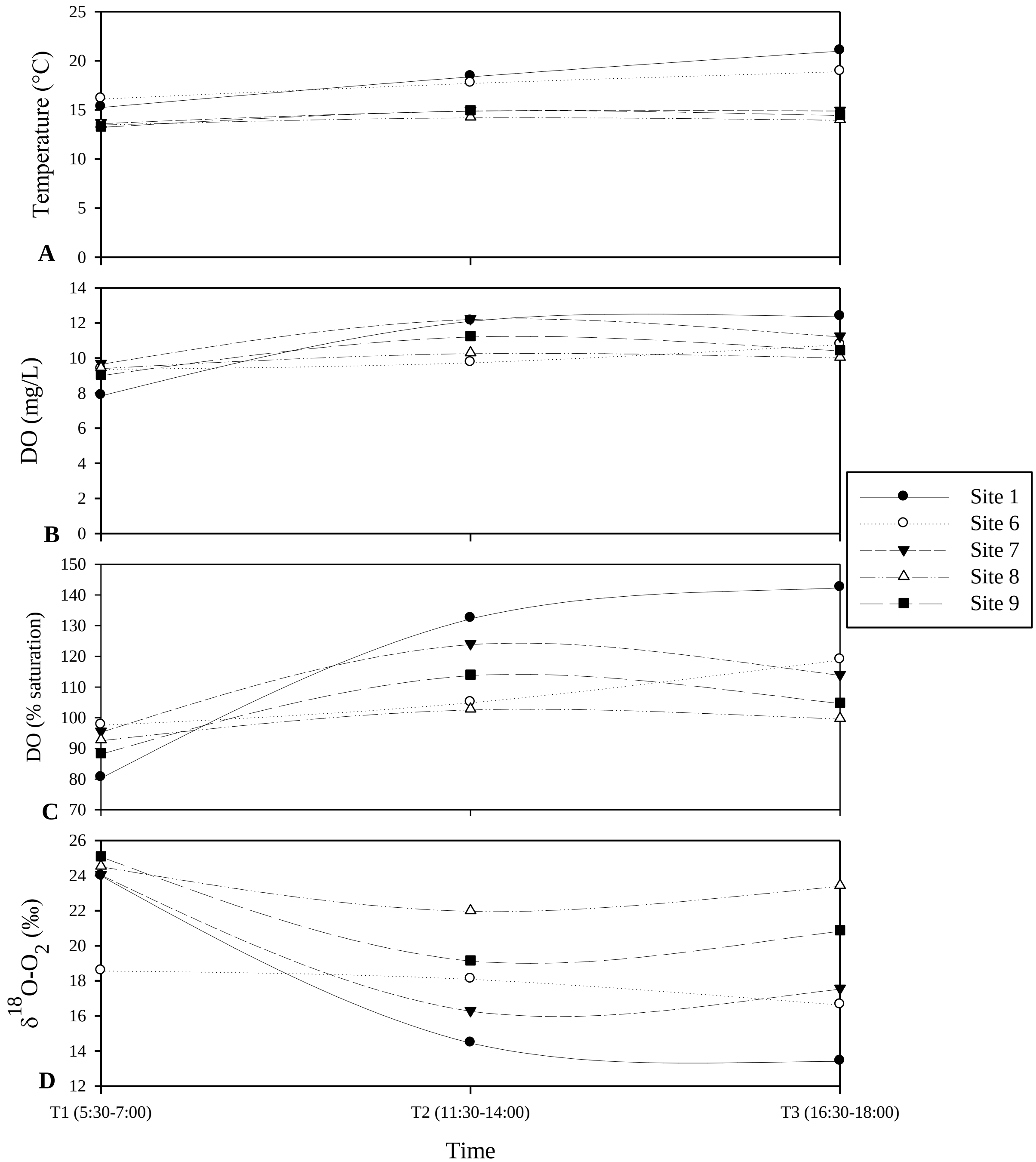

Figure 2.9: A) temperature (°C); B) dissolved oxygen (mg/L), C) dissolved oxygen (% saturation); and D) $\delta^{18}O$-O_2 (‰) from the Belwood Lake diel, May 23 2009. The time is indicated on the x-axis as early morning (T1 5:30-7:00), noon (T2 11:30-14:00), and late afternoon (T3 16:30-18:00). The lines between the points are intended to provide a visual aid for ease in following a particular site's daily changes.

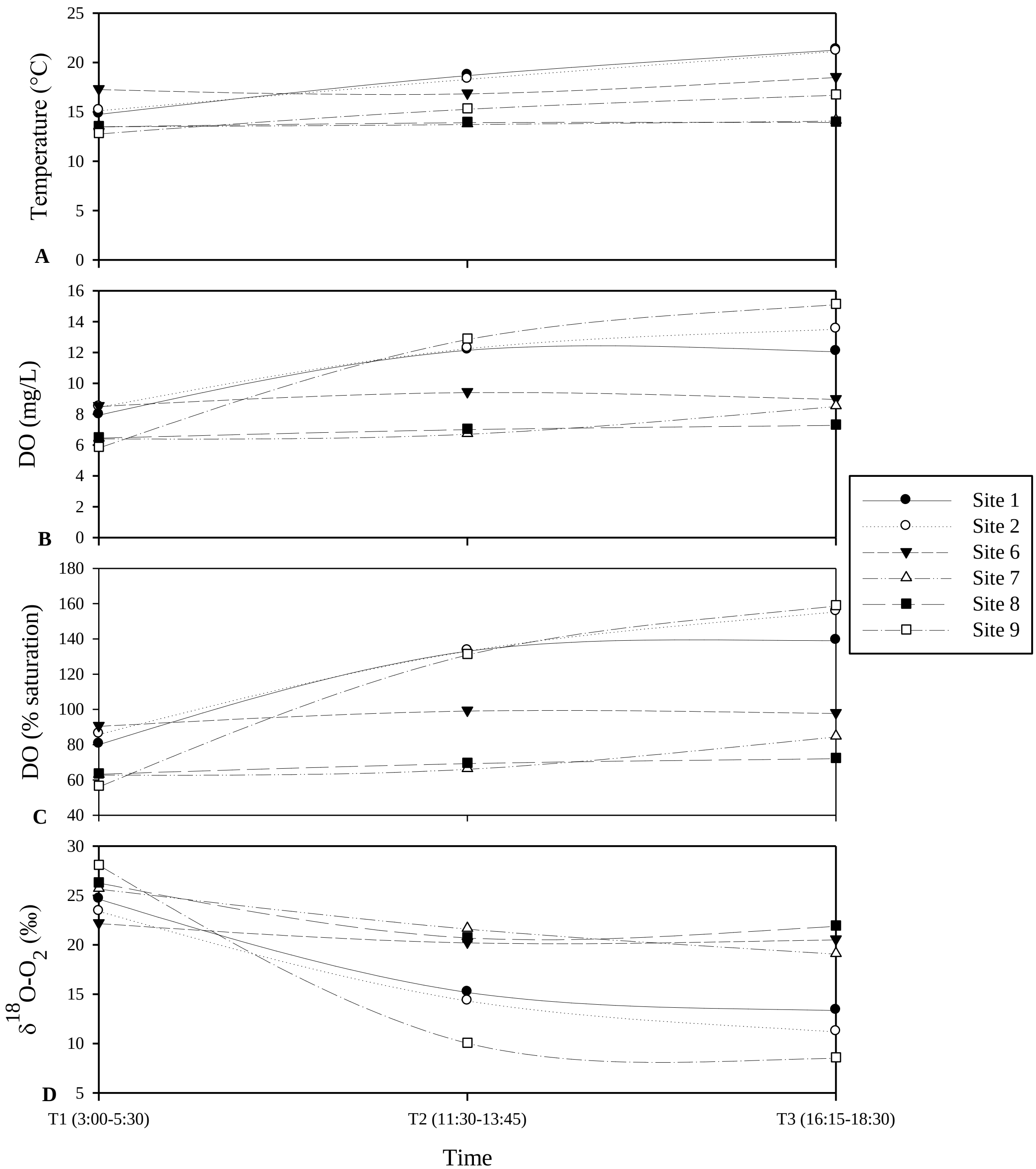

Figure 2.10: A) temperature (°C); B) dissolved oxygen (mg/L), C) dissolved oxygen (% saturation); and D) $\delta^{18}O-O_2$ (‰) from the Conestogo Lake diel, May 22 2009. The time is indicated on the x-axis as early morning (T1 3:00-5:30), noon (T2 11:30-13:45), and late afternoon (T3 16:15-18:30). The lines between the points are intended to provide a visual aid for ease in following a particular site's daily changes.

2.3.6 2010 Diel Analysis

Reservoir Conditions and Precipitation

The 2010 sampling had slightly higher than average rainfall, predominately in June and July; Belwood Lake's volume was higher than the upper buffer from May through October (GRCA, 2010). During the diel, there was an algal bloom at site 4 (personal observation, supplementary data in appendix D).

Temperature

A full diel was conducted at Belwood Lake on August 11, 2010 with the addition of an extra site. The total temperature range (Figure 2.11A) for all sites and times was only 20.7°C to 26.7°C. Sites 7, 8, and 9 did not change by more than 1°C and site 4 remained high throughout. Sites 1 and 10 showed the greatest fluctuation over the diel, a range of 5.6°C and 4.7°C, respectively.

Dissolved Oxygen

The largest range in dissolved oxygen (Figure 2.11B**)** occurred at site 1 between 5.5 mg/L and 10.25 mg/L. Sites 7, 8 and 9 had oxygen cycles that were muted in comparison to site 1. All measurements for the three sites fell between 3.65 mg/L to 6.2 mg/L. Further downstream at site 10, the cycle was more pronounced, though on average still ~2 mg/L lower in amplitude than site 1. The measurements taken at site 4 showed relatively high concentrations, with little overall cycle, ranging only from 11.15 mg/L to 12.95 mg/L.

The unexpected jump in downstream DO at T6 coincided with a release of water from the reservoir and an increase in flow.

$\delta^{18}O$-O_2

The $\delta^{18}O$-O_2 for this diel (Figure 2.11D) showed a number of different cycles. The largest cycles were observed for sites 1 and 10, with site 1 ranging from 26.9‰ at pre-dawn to

16.7‰ in the afternoon. Site 10 had a pre-dawn value of 28.7‰ and an afternoon value of 17.7‰. Almost no change was observed at site 7. There were small cycles measured at sites 8 and 9, with ranges of 21.7‰ to 26.9‰ and 21.2‰ to 28.2‰, respectively. A small cycle was observed at site 4, but with a very different range than any other site, 11.6‰ to 16.8‰.

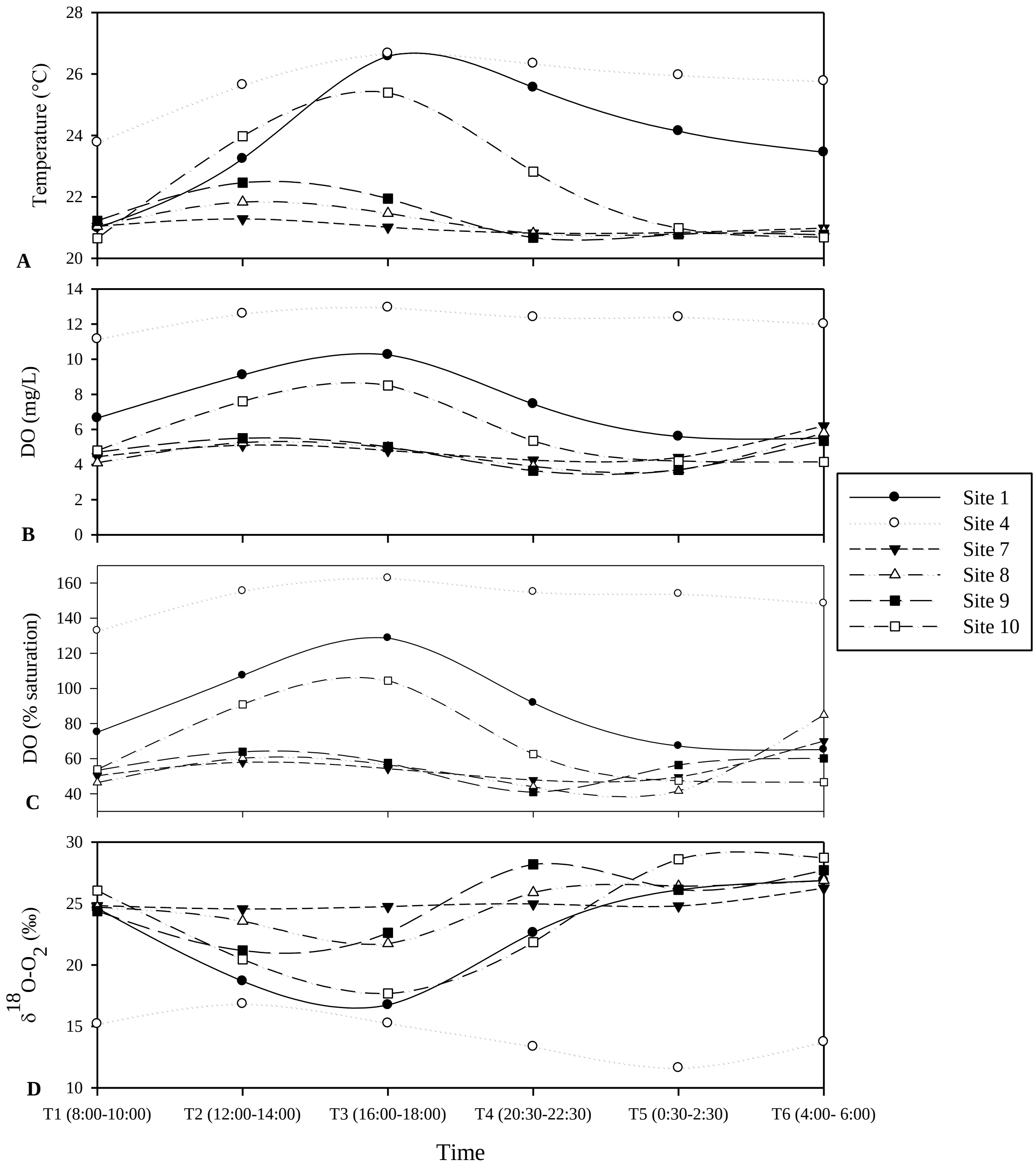

Figure 2.11: A) temperature (°C); B) dissolved oxygen (mg/L), C) dissolved oxygen (% saturation); and D) $\delta^{18}O\text{-}O_2$ (‰) from the Belwood Lake Diel, August 11 2010. The time is indicated on the x-axis as morning (T1 8:00-10:00), noon (T2 12:00-14:00), late afternoon (T3 16:00-18:00), evening (T4 20:30-22:30), night (T5 0:30-2:30), and early morning (T6 4:00-6:00). The lines between the points are intended to provide a visual aid for ease in following a particular site's daily changes.

2.3.7 δ^{18}O-O_2 vs DO Analysis

2008 Seasonal Analysis

The seasonal δ^{18}O-O_2 – DO cross plot for 2008 at Belwood Lake (Figure 2.12A) showed differences between the reservoir and downstream; both days have the reservoir ~7‰ lower than downstream. At Conestogo Lake (Figure 2.12B) the differences are not as apparent, but there is still indication of the separation between upstream and the reservoir compared to downstream.

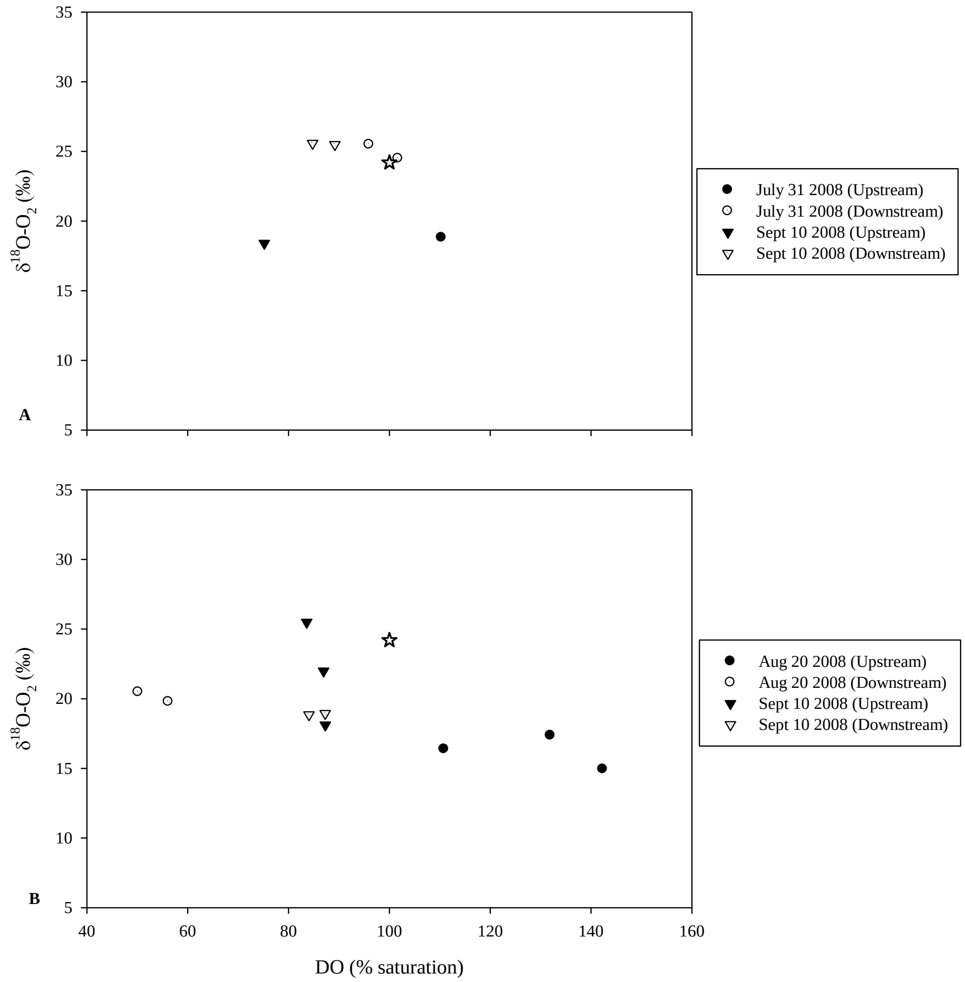

Figure 2.12: 2008 cross plots of δ^{18}O-O_2 vs DO (% saturation) for A) Belwood Lake and B) Conestogo Lake. Air-saturated water is indicated by the star (100% saturation, δ^{18}O-O_2 +24.2‰). Each sampling day is identified by shape; upstream (sites 1-6) are shaded, while downstream (sites 7-9) are outlined.

2009 Seasonal Analysis

The cross plot for $\delta^{18}O\text{-}O_2$ and DO for Belwood Lake (Figure 2.13A) in 2009 shows a pronounced grouping on June 23. The downstream sites tend to be positioned towards high $\delta^{18}O\text{-}O_2$ and low DO, while the upstream sites are typically (but not always, e.g. August 12) situated lower in $\delta^{18}O\text{-}O_2$ and higher in DO. These upstream downstream observations are also seen at Conestogo Lake (Figure 2.13B). The major difference is the further downstream (site 9) grouping more closely with the upstream sites for May 22 and August 25. Both upstream and downstream for November 04 form a tight cluster around air saturated water.

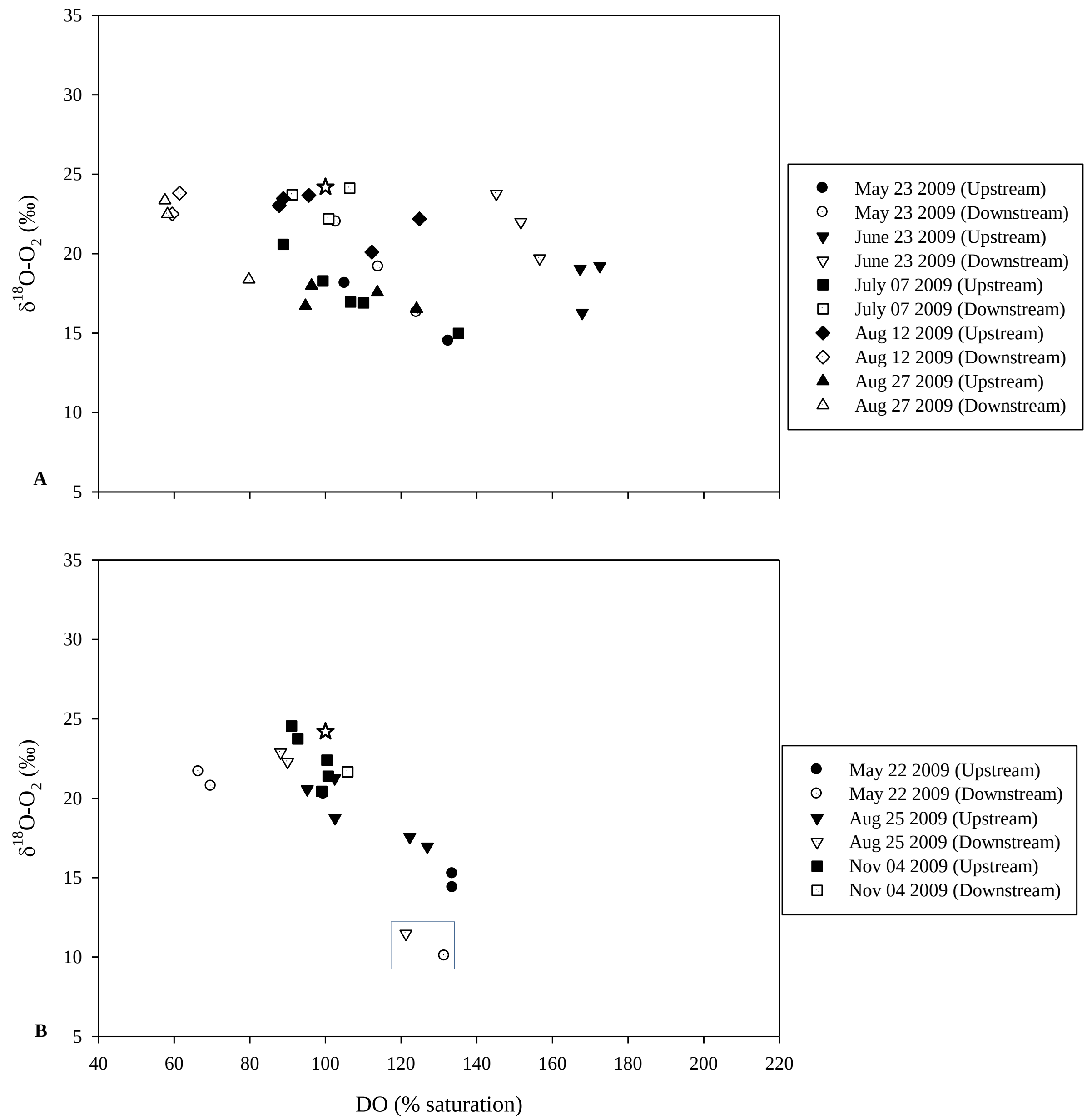

Figure 2.13: 2009 cross plots of δ^{18}O-O$_2$ vs DO (% saturation) for A) Belwood Lake and B) Conestogo Lake. Air saturated water is indicated by the star (100% saturation, δ^{18}O-O$_2$ +24.2‰). Belwood sampling times were 11:30 – 13:50 (May 23), 10:00 – 16:00 (June 23), 12:00 – 18:30 (July 7), 11:00 – 17:00 (August 12), 11:15 – 13:45 (August 27), and 9:45 – 12:30 (November 11). Conestogo sampling times were 11:40 – 13:45 (May 22), 11:00 – 18:00 (June 16), 11:00 – 15:00 (August 25) and 9:45 – 12:30 (November 4). Each sampling day is identified by shape; upstream (sites 1-6) are shaded, while downstream (sites 7-9) are outlined. The box on B) indicates site 9 at Conestogo.

Diel Analysis

As expected, the Belwood Lake diel (Figure 2.14A) in 2009 showed a distinct separation between the morning sampling and afternoon sampling at site 1. Site 7 was similar in general distribution, but not as widespread. For sites 8 and 9 the morning and afternoon separations are apparent, but their overall distribution was much smaller than the other sites. The reservoir grouping was low in $\delta^{18}O\text{-}O_2$ for the samplings, having only a small change at T3. The Conestogo Lake diel (Figure 2.14B) in 2009 shows a clear distinction between the afternoon samplings for sites 1, 2, and 9 compared with all others. Sites 7 and 8 had much smaller groupings, while the reservoir had the tightest grouping; it also remained closest to air-saturated water for the day.

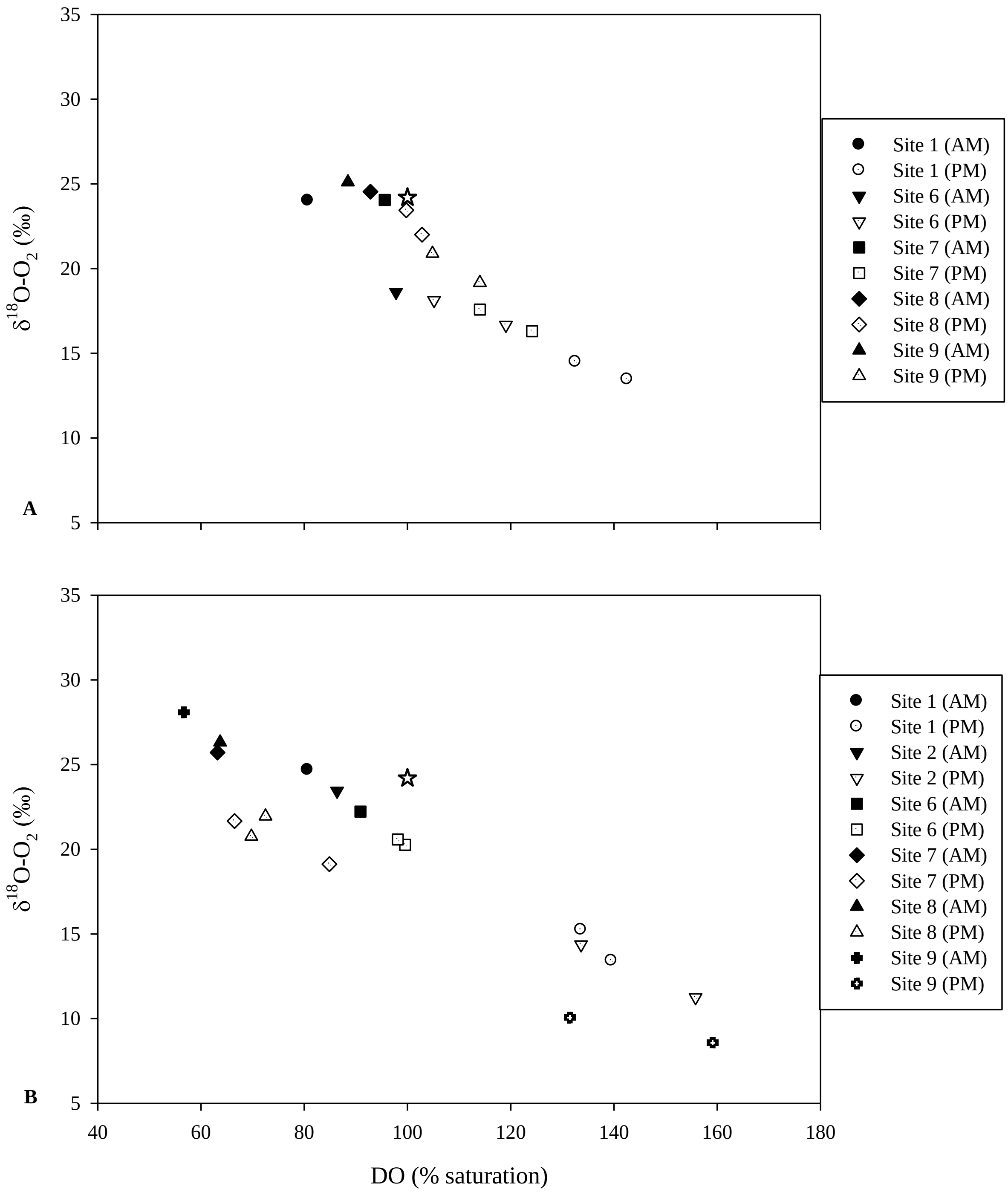

Figure 2.14: A) Belwood Lake (May 23, 2009) and B) Conestogo Lake (May 22, 2009) diel cross-plots of $\delta^{18}O-O_2$ vs DO (% saturation). Air saturated water is indicated by the star (100% saturation, $\delta^{18}O-O_2$ +24.2‰). Each site is identified by shape; mornings are shaded, while afternoons are outlined.

The most notable grouping for the 2010 diel at Belwood Lake is site 4 (Figure 2.15). This was caused by a bloom that was present (personal observation, supplementary data in Appendix D). Except for the afternoon samplings at sites 1 and 10 all others are in a tight cluster with DO less than 100% saturation and $\delta^{18}O\text{-}O_2$ above +20‰.

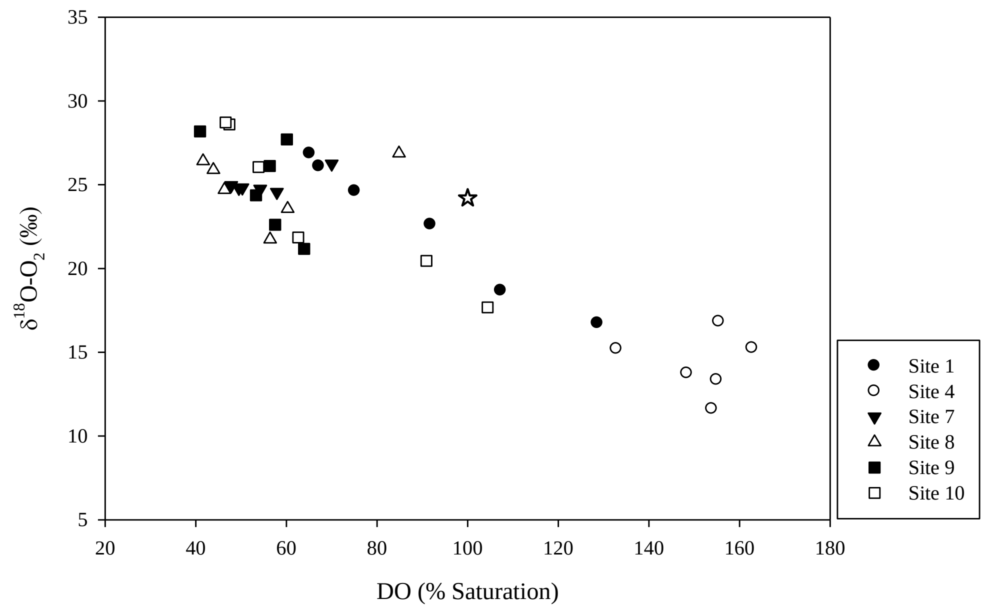

Figure 2.15: Belwood Lake (August 11, 2010) diel cross plot of $\delta^{18}O\text{-}O_2$ vs DO (% saturation). Air saturated water is indicated by the star (100% saturation, $\delta^{18}O\text{-}O_2$ +24.2‰).

2.3.8 P, R, and gas exchange

Dissolved oxygen analysis from field measurements using night time regression

Estimates of GPP and R were calculated for the furthest upstream site (site 1) and the furthest downstream site (site 10) for the August 10, 2010 diel (table 2.1). Site 4 was not calculated due to the presence of an algal bloom (personal observation, supporting data in Appendix D). Sites 7, 8 and 9 were also not calculated; they showed almost no change over the diel so a regression analysis was not possible.

Table 2.1: Estimates of GPP and R from night-time regressions of field measurements from the August 10, 2010 Belwood Lake diel.

Date	Site	$\frac{G}{z}$ (h^{-1})	GPP ($mgO_2/L/h$)	R ($mgO_2/L/h$)
August 10, 2010	1	0.14	1.38	0.72
August 10, 2010	10	0.29	1.86	1.44

Dissolved oxygen analysis from modelled gas exchange estimates

Using the gas exchange coefficients previously calculated through modelling (Venkiteswaran *et al*., 2015), P_{net} was calculated in combination with the data obtained from the May 2009 and August 2010 Belwood Lake diels (Table 2.2). The modelled sites from the DLG were either similar or the same as the sites chosen for this study. The modeled values were used for sites 1, 8, 9 and 10 only. For the upstream isotope values were used, whereas downstream concentrations were used. The physical differences (width, depth, morphometry, etc.) between the remaining sites and those modeled were too great to confidently use those values (Jha *et al*., 2004) and were thus excluded.

Table 2.2: Estimates of P_{net} at Belwood Lake, by modeled gas exchange coefficients (Venkiteswaran *et al*., 2015) and field oxygen measurements. Gas exchange coefficients were modeled using either $\delta^{18}O\text{-}O_2$ or dissolved oxygen concentrations.

Reservoir	Date	Site	$\frac{G}{z}$ (h^{-1})		P_{net} (mgO_2/L/h)	
			Isotope	Concentration	Isotope	Concentration
Belwood Lake	May 23, 2009	1	0.12		1.42	
		8		0.19		0.11
		9		0.19		0.49
Belwood Lake	August 10, 2010	1	0.12			0.7
		8		0.19		-0.34
		9		0.19		-0.91
		10		0.19		0.45

2.3.9 Dissolved oxygen changes at a hydraulic structure

The aeration efficiency was calculated for all three dates on which vertical profiles of DO in the reservoirs were done during the 2008 sampling season and in June 2009 at Belwood Lake. Though there was some aeration occurring at the dam, it was not enough to cause DO saturation in the plunge pool (Table 2.3). The 2009 profile had DO measurements above and below the dam well above saturation; thus, calculating whether the dam increased the DO to saturation as water travelled through was not applicable.

Table 2.3: Estimation of oxygen deficit ratios and re-aeration efficiencies from the 2008 profile samplings of Conestogo Lake (August 20, 2008 and September 10, 2008) and Belwood Lake (September 10, 2008). Samples were taken in-reservoir at depths indicated and below the dam.

Site	Date	Reservoir Sampling Depth (m)	DO at Sampling Depth (mg/L)	DO at Sampling Depth (% saturation)	Temperature at Sampling Depth (°C)	DO Below Dam (mg/L)	DO Below Dam (% saturation)	Below Dam Temperature (°C)	r	E
Conestogo Lake	Aug 20 2008	15	3.5	39.7	20.8	4.9	56.1	20.3	1.34	0.26
Conestogo Lake	Sep 10 2008	14	5.9	66.5	20.9	7.4	84.0	20.8	2.21	0.55
Belwood Lake	Sep 10 2008	12	5.7	63.8	19.9	8.0	89.2	19.6	3.74	0.73

2.4 Discussion

2.4.1 Alteration of DO and temperature in reservoir-impacted rivers

It is important to put the changes observed between upstream and downstream reaches into context with regards to the prevailing conditions at the time of sampling. These conditions varied greatly over the sampling events. The differences between sampling events were the result of seasonal variation in combination with management practices; in addition, differences between the lakes were demonstrated. Both Belwood Lake and Conestogo Lake altered the DO and temperature in the downstream environment, each occasionally acting as sources or sinks for heat and DO. Though some design mechanisms and practices are in place to mediate the effects of these reservoirs, impacts were observed.

2.4.2 Spring Filling and Fall Drawdown

Both Belwood Lake and Conestogo Lake follow yearly management cycles similar to most temperate reservoirs (e.g., Boyd *et al*., 2000; Shantz *et al*., 2004; Baldwin *et al*., 2008). They are filled in spring from rain and snowmelt and drained throughout the summer as needed for flow augmentation; the remaining volume is reduced in the fall in preparation for the following spring melt.

Measurements taken at Belwood Lake after spring filling showed that the downstream DO was already affected by the presence of the dam, as the DO in the reservoir was lower than that found upstream during the day. Temperature was also affected; the upstream sites were several degrees warmer than the downstream. Similarly, after filling Conestogo Lake, DO concentrations were higher above the reservoir than below. Conestogo also had higher temperatures above the reservoir than below.

The post-drawdown observations at both lakes were similar in that the DO seemed unaffected by the presence of the dams, as the upstream sites were comparable to the downstream for each reservoir. Both Belwood and Conestogo Lakes had temperatures that were cool upstream, but within the reservoir and downstream remained warm.

These observations are consistent with what was initially anticipated as well as what has been documented in literature. As explained by McCartney (2009), the practice of spring filling causes the water volume to increase, thus increasing the potential for heat retention. The act of changing the reservoir from lotic to lentic and back again through the year can stress the downstream environment, particularly with regards to temperature and the downstream biotic life (Baxter, 1977; Thornton *et al*., 1990).

2.4.3 Storm Events, Rain Events, and Flow

Both storm and rain events play significant roles in the function and operation of reservoirs (Yakobowski, 2008). The incoming water increases the volume held within the reservoir; this volume may not be aligned with seasonal targets, or worse, may exceed structural capabilities. Regardless, the outflow will be increased in compensation so that the required target volumes can be achieved. This may lead to higher than normal flow maintained for extended periods of time.

The 2009 sampling season at both Belwood and Conestogo Lakes was heavily influenced by higher than average rainfall. The effects on the reservoirs and downstream thus varied from initial expectations that were based on data gathered from the Day in the Life of the Grand (section 2.1.6). At Belwood, it was expected that there would have been a stronger deep-release related influence much earlier in the summer (July). It was not observed until late August, when the DO concentrations were much lower downstream. The temperature also remained much more stable; a larger difference was expected between the in-reservoir and downstream. It was anticipated that the water entering the reservoir would be much warmer than what was leaving the reservoir and downstream. For Conestogo, the large DO change that was observed in 2008 was not observed in 2009, and temperature was only impacted largely on one occasion (June 16).

Though changes in the Grand River flow occur, in Brazil, for instance, torrential rain storms have been known to occur unexpectedly requiring large dams to increase flow by over 4000 m^3/s (de Oliveira Naliato *et al*., 2009). The general idea of how the different zones in a reservoir function, as modeled by Thornton *et al*. (1990), has recently been questioned. As the importance of flow management increases, some now realize that the zonation likely changes, sometimes drastically, as the flow changes to accommodate differing conditions (Brooks *et al*., 2011).

2.4.4 Residence Time and Stratification

The preceding sections discussed both seasonal management practices and flow. Though discussed separately for ease, they both have effects on residence time and therefore stratification potential; specifically, the higher the flow the shorter the residence time, the lower the flow the longer the residence time. Stratification in reservoirs is often similar to that in lakes; occurring in summer and winter with mixing usually in the spring and fall (Thornton *et al*., 1990). However, the potential for a reservoir to stratify depends on both the flow and management practices (section 2.4.2).

For much of 2008 and 2009 little evidence of stratification was observed at Belwood Lake. This lack of stratification could have been caused by short residence times as the reservoirs were in a state of volumetric equilibrium due to the high rain volume (high outflows were maintained to compensate for the continually high inflows). In June, a temperature difference was apparent between the surface of the reservoir and downstream, but DO remained unaffected. In August of 2009, stratification appeared at Belwood Lake for both temperature and DO. Conestogo Lake had thermal stratification in June of 2009, but this was not observed later in August. There was more evidence for stratification in 2008 than 2009.

The downstream temperature and DO corresponded to the depth of water from which the withdrawal was made, similar to observations made by Casmitjana *et al*. (2003). For bottom-draw dams the release of cold water downstream is a direct consequence of stratification (McCartney, 2009).

It is possible to alter stratification patterns both naturally and artificially. Though not believed to have occurred during this study, differing water densities between the reservoir and the inflow may cause events such as interflow or underflow (Thornton *et al*., 1990). Changes in dam discharge patterns, for instance selective withdrawal from various depths, can also disrupt stratification (Casamitjana *et al*., 2003).

2.4.5 Aeration

For most jurisdictions, there are guidelines as to the minimum DO required downstream of a reservoir (Friedl and Wüest, 2002). For most, the water quality minimum is between 5 to 6 mg/L (Friedl and Wüest, 2002). As the nature of some dams is to release water with low DO, many dam structures are designed to aid in the aeration process and thus increase the DO in the downstream environment (Baylar *et al*., 2010).

In the Grand River, the water quality minimum is considered to be 4 mg/L for DO (Boyd, 2008). The Shand dam was occasionally providing some amount of reaeration to the outflowing water. The dam at Conestogo only indicated DO increases once, in June 2009. In 2010, the diel at the Shand dam validated the prediction that despite some re-aeration occurring over the surface of the dam to achieve the target DO concentrations at site 7, by sites 8 and 9, the DO had once again dropped to 3.7 mg/L and 3.65 mg/L. Below site 7, biologically driven DO cycling began to re-establish. This diel variation (from biological cycling) may account for the decrease in oxygen in the downstream sites despite the physical re-aeration of the outflowing water.

During the diel sampling of 2010 there was an increase in flow between T5 and T6. This caused an increase in DO at the immediate downstream sites. Site 10, though, appeared to have no changes from the increase in flow. It is possible that the water affected by the change in aeration had not yet reached that site by the time of sampling. An issue addressed by many (e.g., Bednarek, 2001; Friedl and Wüest, 2002; de Oliveira Naliato *et al*., 2009) is that, in combination with an increase in flow, the gas exchange that will occur over a structure could potentially lead to further downstream stress in the form of supersaturation. This is particularly of concern for fish populations; as it has been shown that during prolonged periods in environments with gases of 115% saturation or more can potentially lead to death from emboli accumulating in gill capillaries (Mesa *et al*., 2000). Though it is unlikely to occur in the Grand River, as flows do not normally reach extremely high rates, it is still a general concern for dam operators.

From engineering perspectives it is often important to determine the aeration efficiency of a hydraulic structure. There are several ways in which the re-aeration of water coming through or over a dam can be calculated; Gulliver and Rindels (1993) and Baylar *et al*. (2010) are examples. These measurements and calculations are sensitive to numerous parameters such as depth of release, tailwater depth, temperature and discharge rate (Gulliver and Rindels, 1993); though only temperature and discharge are thought to be major contributing factors here as the others do not change dramatically at this site. Both the deficit ratio (r) and transfer efficiency (E) were calculated at the Shand dam and Conestogo dam. The downstream water was never fully saturated with DO for the 2008 season, nor was there supersaturation, as no E value reached 1 or more. Some re-aeration was occurring, as none of the values were 0. Calculations were not performed for June 23, 2009, as the DO above the dam structure was already above saturation, as was the DO below the dam (Butts *et al*., 1999). This was likely due to high reservoir throughput and high discharges from the dam during May and June (2009) in response to the large amount of rainfall.

2.4.6 Calculating P, R, and gas exchange

An attempt was made to calculate the rates of photosynthesis, respiration and gas exchange downstream of Belwood Lake, using the nighttime regression method (Parker *et al.* (2005), Young *et al*. (2004), Grace and Imberger (2006)). This was done using only the August 2010 diel data, as it was the only sampling with nighttime data. Only sites 1 and 10 could be used, as they were the only two sites that had enough of a diel DO change to allow for calculation. The remaining sites could not be used as they did not have a large enough change in DO during the diel. Additionally, the change in flow in the early morning would have also affected any calculations made for sites 7, 8, and 9. Considering that the method used was intended for free-flowing waters it logical that it would work for the most river-like sites; the results also show the difficulty in using this method for the non-typical river reach (i.e. under an impoundment).

A secondary attempt was made using field measurements from Belwood Lake and modeled gas transfer velocities from Venkiteswaran *et al*. (2015). This was possible as the sites modeled were from the DLG and thus similar to the current study sites. Using modelled data from similar locations has been shown to be a useful approach when other measurements are unavailable (Jha *et al.*, 2004). The 2010 Belwood Lake data produced expected results, with the furthest upstream and downstream sites behaving in the most river-like manner. The 2009 sampling unfortunately does not have the furthest downstream site and thus the results are more difficult to interpret. The positive P_{net} calculated may be valid in this instance, as the prevailing river conditions for the 2009 diel were different than the 2010 (DO concentration, temperature, etc.).

Both attempts at determining the downstream metabolic functions proved difficult, as the methods are intended for an undisturbed river ecosystem. Though results were obtained for sites below the dam, they are questionable.

Few have studied the downstream changes to P and R caused by the presence of an impoundment. One study, done on the Clearwater River, Idaho (Munn and Brusven, 2004) used an in-stream chamber method to calculate P and R rates. They showed high rates of both production and respiration below the Dworshak Reservoir. The production was attributed to the aquatic moss *Fontinalis neo-mexicana* that was growing in the area. Further study on the Dworshak Dam has shown that it has a multiple level release system used to maintain downstream river temperatures between 3°C and 13°C for the nearby fish hatchery

(Munn and Brusven, 2004). In the present study, very little to no macrophyte growth was observed downstream of Belwood Lake. In addition, the temperatures downstream of the Shand dam are much more variable, and in particular, warmer in the summer season. The other factor differentiating the Dworshak Dam study from this is the design below the dam structure. The Dworshak Dam was designed with its spillway to one side and a hydroelectric station and hatchery to the other, leaving a large bay-like area with slow moving water (Department of the Army Corps of Engineers, 2011). This area would allow for biological cycling to begin before water is transported downstream. Another study of downstream calculations of P and R were made by Ideva *et al*. (2008). Their study was done over 6 km downstream of the dam, which may not have been truly representative of the downstream effects that a dam would have on P and R.

There are methods of calculating P and R in streams (e.g., Odum, 1956; Izagirre *et al*., 2007) and there are also many in-reservoir models that have proven successful (Gergel *et al*., 2005). These are all helpful in the greater story of river-reservoir-dam-river interaction, but clearly several areas need improvement. The development of a model has been considered, but was not the goal of this thesis.

2.4.7 Downstream and Recovery

The DO downstream of the reservoirs was seasonally variable. In several cases (Belwood: September and October of 2008 and May, June, July and November 2009; Conestogo: June, November 2009) site 7 had higher or similar DO as what was measured either upstream or in the reservoir. This happened despite thermal differences measured in the reservoir compared to downstream (the reservoir surface was much warmer than what was measured at site 7). In these circumstances thermal stratification did not have an effect on the downstream DO concentrations at the time of measurement, and thus no effect on DO recovery, as it was not necessary. For August 2008, May 2009 and August 2009 at Conestogo and July 2008, August 12, 2009 and August 27, 2009 at Belwood, the downstream DO (both concentration and saturation) was observed to be lower than that measured at the reservoir surface despite that the temperatures only differed by a couple of degrees. This indicates that at both reservoirs, thermal stratification may not necessarily have an effect on the downstream DO recovery. However, a lack of thermal stratification may not mean that other clines are not present. For example DO, as shown here, may not reach the concentration measured upstream for some distance downstream of the dam.

An attempt was made to determine where the downstream cycle returns to amplitudes similar to those above the reservoirs by means of diel sampling when the reservoirs are indeed having an impact; this distance has been referred to as the discontinuity distance (Ward and Stanford, 1983) or the recovery distance (Palmer and O'Keeffe, 1990). Overall, the DO cycles tended to be muted downstream of both the dam at Belwood Lake and the dam at Conestogo Lake for the diel sampling trips in both 2009 and 2010. It was not possible to determine the recovery distance downstream of Belwood Lake for 2009 or even with the addition of site 10 in 2010 at 4 km downstream. In theory, the downstream river should return to a similar cycle as seen above the reservoir, or reach a new equilibrium, unfortunately, below site 10 there is a retention pond for a weir-type dam. Strong respiration rates and other processes such as oxidation could also contribute to the delay in the full oxygen cycle return. Therefore, the Grand River did not fully recover to temperatures or DO concentrations and cycles by site 9 in 2009 or before reaching the next impoundment for the 2010 sampling event. For Conestogo Lake the DO cycle did return to the upstream cycle amplitude by site 9 at Glen Allan, 2.6 km downstream for the 2009 trip.

Though the downstream distance travelled by the Grand River was longer than that of the distance travelled by the Conestogo River, it was unable to recover. There are several possible explanations for this. One is the summer low-flow minimum that the Grand River Conservation Authority attempts to maintain. The flows below Belwood tend to be greater than that at Conestogo. This may give the Conestogo River more time to recover before reaching the next site. There were more submersed macrophytes in the Conestogo than in the Grand for the reaches below the dams. Downstream of Conestogo Lake is much deeper than the downstream of Belwood Lake. It also has a much softer bottom, whereas the Grand River is hardened.

A study done locally some time ago at Guelph Lake (on the Speed River, a major tributary to the Grand River) observed that the dam was releasing anoxic water and causing low concentrations below the dam (Mackie *et al*., 1983). Their findings showed that within 1 km downstream of the dam the DO concentrations had re-established to the upstream conditions. Though it is in the same area, this study may not be a good comparison as the size and morphology of Guelph Lake differs from Belwood and Conestogo Lakes.

Some have found that selective withdrawal mechanisms have allowed for faster recovery downstream of a dam. Cassidy and Dunn's (1985) study of Applegate Lake, Oregon, found downstream DO concentration above 10 mg/L at 1 km downstream, though this did not hold during periods of stratification when the downstream DO dropped below 4 mg/L violating their minimum targets.

2.4.8 $\delta^{18}O$-O_2 vs DO Analysis

Most of the information from the 2008 and 2009 $\delta^{18}O$-O_2 – DO cross-plots (Figures 2.12 and 2.13) is the general distinction between upstream and downstream sites, as well as what was the dominating force between photosynthesis and respiration. The diels at both lakes showed how much the upstream and furthest downstream sites change throughout the day with regards to photosynthesis and respiration. Both Belwood and Conestogo's reservoir and immediately downstream sites clustered together; though, sites 6 and 7 at Belwood did have a photosynthetically driven change in the afternoon. A notable difference between the reservoirs is that Conestogo tended to have its cluster located in a more respiration driven area where as Belwood was more photosynthetically driven. The 2010 diel (Figure 2.15) clearly showed the full diel cycle at sites 1, and 10. Sites 7, 8, and 9, again, tended to cluster but with respiration as the dominating factor (compared to 2009 with photosynthesis). The algae bloom at site 4 was very productive for the entire diel duration.

2.4.9 Model Comparison

In section 2.1.7 a conceptual model was presented to summarize how a river-reservoir-river ecosystem is expected to behave. The data from the May 2009 Belwood Lake and Conestogo Lake, and the August 2010 Belwood Lake diels, have been superimposed onto the conceptual model in order to see how well the expected DO cycles and observed DO cycles compare (Figure 2.16, 2.17, 2.18). The upstream data were either within or close to the expected ranges. Site 2 at Conestogo Lake (Figure 2.17) was well above the expected range for daily variation; a possible explanation could be that this is an area with abundant agriculture and an increase in nutrients could cause high productivity. In the transition zone, the data at Belwood in 2010 (Figure 2.18) were not what was expected, though this may have been due to the presence of an algae bloom.

The samples taken below the dams were variable at Belwood, and differed between the reservoirs. In 2009, Belwood Lake showed indication of reaeration in the plunge pool and the expected muted DO cycle starting further downstream at site 8 and recovery starting at site 9. At Conestogo Lake, site 7 was closer to what was expected in the conceptual model; though, similar to Belwood, it was site 8 that has the muted cycle. Site 9 for Conestogo exceeded both the upper and lower estimations for DO.

For the 2010 Belwood diel (Figure 2.18), site 7 fell almost entirely within the expected range. Further downstream of the dam is where there was the greatest difference from what was predicted; the site 8 and site 9 DO cycle was overall much lower and muted, as previously mentioned this could be from several processes, including high respiration rates or oxidation. The DO cycle did not begin to increase until much further downstream than expected. As previously discussed, the anomaly at T6 was likely due to an increase in flow which then increased downstream DO concentrations.

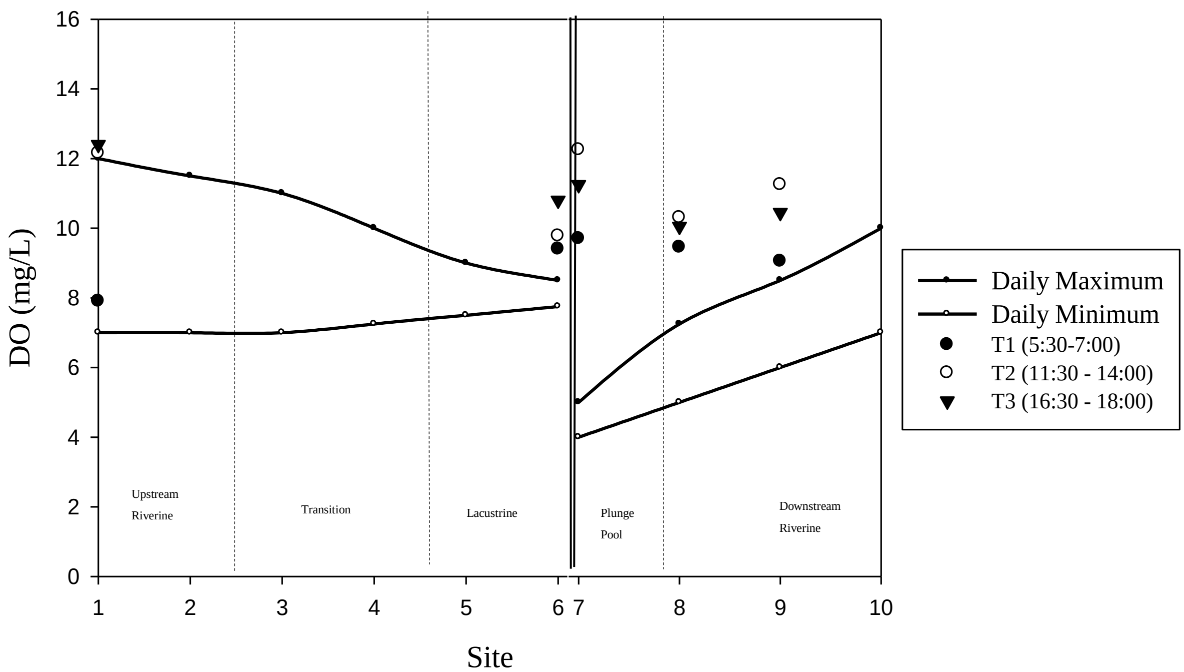

Figure 2.16: Belwood Lake (May 23, 2009) diel DO (mg/L). The difference between the theoretical daily maximum and minimum represents the amplitude of the DO cycle. The graph is further divided into the traditional reservoir zones as described by Thornton et al. (1990). The lacustrine represents the surface waters, the dam is between sites 6 and 7.

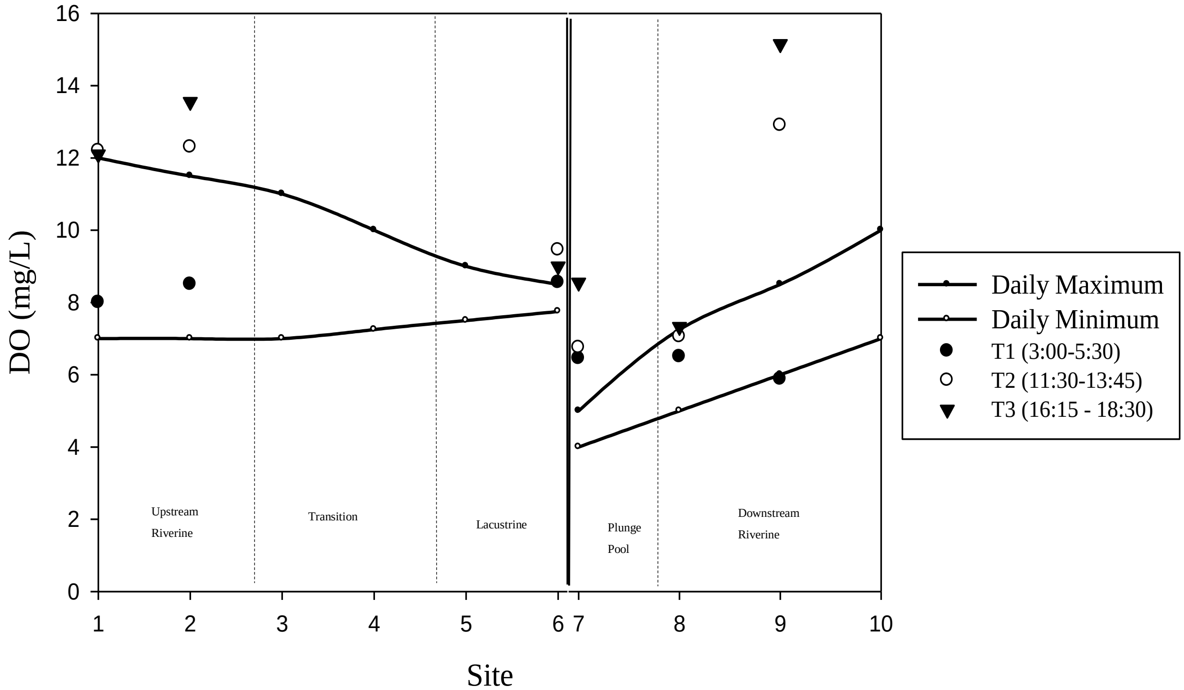

Figure 2.17: Conestogo Lake (May 22, 2009) diel DO (mg/L). The difference between the theoretical daily maximum and minimum represents the amplitude of the DO cycle. The graph is further divided into the traditional reservoir zones as described by Thornton et al. (1990), with the lacustrine representing the surface waters. The dam is between sites 6 and 7.

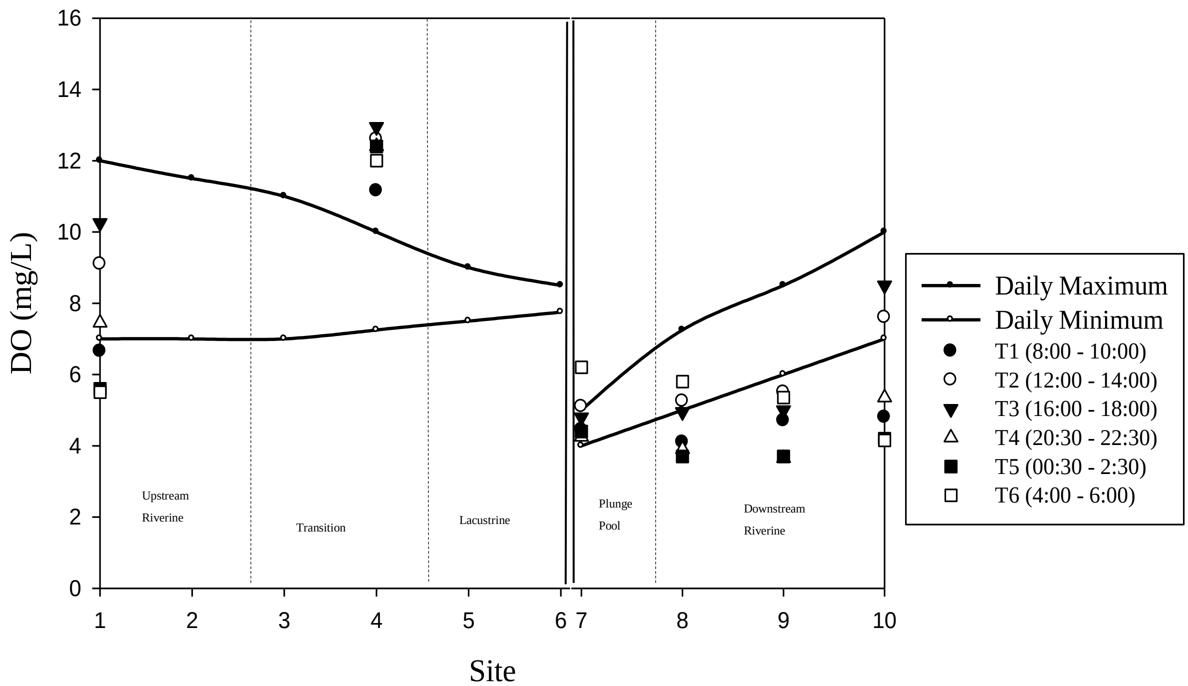

Figure 2.18: Belwood Lake August 2010 diel DO (mg/L). The difference between the theoretical daily maximum and minimum represents the amplitude of the DO cycle. The graph is further divided into the traditional reservoir zones as described by Thornton et al. (1990), with the lacustrine representing the surface waters. The dam is between sites 6 and 7.

2.4.10 The importance of reservoir management for downstream DO and temperature, a comparison of the DLG and this study

The seasonal influences on both of the reservoirs studied are apparent when compared to the initial DLG findings. As previously discussed, weather influences management and thus has a major effect on how the Shand Dam is operated. The first DLG took place in 2007 (September), an extremely dry year with only 672 mm annual precipitation (UW Weather Station Report, 2007). During a dry year the best practice for dam management is to retain as much water as possible in spring, increasing the residence time (which at summer peak volume and low flow targets could be 165 days at Belwood and 202 days at Conestogo), stratification potential and the time for biological processes to occur. In a dry season, in-reservoir process would have a great impact on the downstream environment.

For 2008, the recorded annual precipitation was 1160 mm (UW Weather Station Report, 2008). In 2009, when the majority of sampling for this study was conducted, the annual precipitation was 899 mm, with much of it coming in the spring between the May and June samplings (UW Weather Station Report, 2009). During a wet year, the best practice for management is to keep the reservoir draining at a reasonable rate so that there will always be enough room to absorb a flood without reaching the structural limits of the dam. This type of management would lead to a much shorter residence time (considering volume at 90% capacity and a small increase in flow could allow, for example, 87 days at Belwood, and 100 days at Conestogo). The shorter residence time would reduce the amount of time for biological processes to occur. These differences between dry and wet years (and seasons) generate variability in the impact of reservoirs on the downstream environment. Another factor to consider is that even during wet years there is the possibility of dry periods, which in turn would lead to altering the management of the dam for that particular period.

These differences are enough to alter the river-reservoir-river ecosystem from year to year. In 2007, the low flow of the incoming water would have minimized the release of water from the dam, increasing the residence time and the potential for stratification to develop. In 2009, a high precipitation year, higher flow was maintained decreasing the residence time and stratification potential. As such, the impact that the reservoir had on the downstream environment with regards to temperature was more pronounced in the 2007 DLG1, though the influence on temperature was still seen at Belwood Lake in June and July of 2009. The largest changes in DO were at Belwood in August of both 2009 and 2010; indicating that despite being wetter years overall, periods of low precipitation could lead to less water released from the reservoir, as was the case in August of 2009 (GRCA, 2009), and August of 2010 (GRCA, 2010).

2.5 Summary and Conclusions

The observations made during this study have given insight into the variable nature of the river-reservoir-river ecosystem, as well as the seasonal operation of two hydroelectric dams;

all within the context of DO concentration and DO cycling. The DO was variable across all sites, seasons and years; most notably in the regions downstream of the dams. The $\delta^{18}O\text{-}O_2$ had little pattern, except that it was higher below the dam than at the surface of the reservoir for all but one sampling. The management practices for reservoir operation change depending on the prevailing weather conditions. An increase in precipitation allows for a higher throughput of water and decreases the residence time. This decreases the likelihood for stratification to develop, which lessens the impact on the downstream environment as the DO and temperature of the water leaving the reservoir is similar to what had entered the reservoir. There was difficulty in determining the P, R and G downstream of the reservoirs given the current information. The night-time regression was successful in the most 'river-like' sites. Further attempts were made by using previously modelled values, these gave results. More study is required to determine better and more accurate values for P, R and G. The ability of the dams to provide reaeration was also calculated. The dams were providing some, but not always at a high efficiency. On some occasions the DO was already at or above 100% saturation. The Shand Dam at Belwood Lake was more efficient than the Conestogo Dam for adding DO through the dam structures. Though 100% saturation can be achieved below the dam, it is not always so further downstream. The original conceptual model proposed did not compare well with the measured data. Though the conceptual model may have been valuable for describing overall processes, it did not always account for time specific events, such as algae blooms, increases in reservoir discharge, and other potential occurrences not quantified in this study.

<u>3.0 Effects of impoundments on nitrogen species in the Grand and Conestogo Rivers</u>

3.1 Introduction

3.1.1 Introduction

The large watershed of a reservoir in comparison to a natural lake translates to a strong terrestrial influence on reservoirs by means of both nutrients and sediments (Kimmel *et al*., 1990; Wetzel, 2001). This basin influence can result in reservoirs with distinct gradients along the longitudinal aspect, which in turn has the potential to alter the distribution of ecological processes such as primary production (Kimmel *et al*., 1990). Annual variation of land-water interactions also contributes to altered nutrient inputs and thus gradients. The presence of a dam and associated reservoir allow time for processing that can modify nutrient species, as well as change the timing, method and concentration of nutrient delivery downstream (Baldwin *et al*., 2010).

3.1.2 Nitrogen cycling in and downstream of reservoirs

Human activities such as intensive agriculture with increased fertilizer usage, sewage input and atmospheric deposition have greatly increased the rate of nitrogen input to the terrestrial and aquatic environment causing such ill effects as global nitrate contamination (Vitousek *et al*., 1997; Xue *et al*., 2009). The World Health Organization has suggested a global drinking water limit for nitrate at 10 mg N/L (Xue *et al*., 2009). Despite the global increases, it has been noted that a considerable amount of nitrogen can be removed in reservoirs (Kunz *et al*., 2001).

Nitrification (see 1.1.6) is the biological conversion of ammonia or ammonium to a more oxidized state, and ultimately to nitrate (Wetzel, 2001). During aerobic conditions in a reservoir, nitrification may alter the concentration of different nitrogen species and thus what is flowing downstream.

Denitrification (see 1.1.6) is the microbially facilitated reduction of nitrate or nitrite, usually progressing to N_2 via N_2O (Wetzel, 2001). Aquatic ecosystems have hypoxic habitats in which denitrification can occur, e.g., in sediments or in the water column during thermal stratification when the hypolimnion becomes anoxic (Harrison *et al*., 2009). This process has the ability to considerably alter the availability of reactive nitrogen in aquatic environments (Abe *et al*., 2003).

Reservoirs are heterogeneous and dynamic systems; attributes and processes vary with space and time. In particular, the transition zone can be a location of significant cyanobacterial N-fixation (see 1.1.6) as well as a very active zone for other processes (Scott *et al*., 2009).

Ammonium usually enters the nitrogen cycle by the excretion of animals or by decomposition of organic matter. It is usually found in low concentrations, as it is the preferred form for uptake by bacteria and algae as well as a substrate for nitrification (Kalff, 2002). A high NH_4^+ concentration in flowing water is usually from allochthonous sources (Kalff, 2002).

3.1.3 Global Examples

Dams vary in size, geomorphology, temperature, precipitation inputs etc. As impounding has become more prevalent, research has been conducted and information is now becoming increasingly available on their impacts.

In the United States, much work has been done on the Mokelumne River in California. Here, Henson *et al*. (2007) determined that Lake Lodi, a reservoir on the Mokelumne River, acted as a source for dissolved nutrients but a sink for particulate matter. Studies were done comparing a free-flowing river in a similar watershed to the impounded Mokelumne River; they found that the free flowing river had higher NO_3^- concentrations during the wet months, while the reservoirs were generally retaining NO_3^- during the wet months and releasing it over time (Ahearn *et al*., 2005). In some cases there was no indication of high nitrogen inputs, and the high nitrate measured was caused by mineralization and nitrification within the reservoir (Ahearn *et al*., 2005). Missouri has also been an area of reservoir research. Both Perkins *et al*. (1998) and Jones and Knowlton (2005) determined the importance of the watershed cover (agricultural versus forest cover) as influences on the inflow of nutrients and suspended solids into the reservoirs located downstream. In Texas, three different reservoirs were found to have varying N-fixation rates; both the upstream riverine and the lacustrine zones had low rates while the transition zones of all three had high rates (Scott *et al*., 2009). In the Gunnison River, Colorado, Stanford and Ward (1983) found increased nitrate concentrations downstream; they attributed this to fixation and nitrification.

In Tanzania, the Kihansi Reservoir is one of the few cases for which there are both pre- and post-construction data. With this information, Ideva *et al*. (2008) were able to determine that upon construction, the Kihansi Reservoir became a nutrient retention site as high upstream and low downstream concentrations were observed. Nitrogen concentrations in the reservoir were directly related to the surface runoff of the area and during the dry seasons the nutrients were being stored within the reservoir (Ideva *et al*., 2008).

Brazil has become a major contributor to reservoir information and research. In the Paranapanema River, de Oliveira Naliato *et al*. (2009) measured the operational procedures on an hourly basis to determine the effects on the downstream ecosystem. They found a direct influence from the reservoir water column on the downstream river.

The Sau Reservoir, located in the Catalonia region of Spain, has extreme nutrient gradients from the upstream river to the dam; it is suspected that the cause of the great longitudinal variation is its geomorphology, as it is in a long, deep and narrow river valley (Comerma *et al*., 2003). A group of reservoirs on small mountain streams has also been studied in the central part of Spain. For all four of these reservoirs, there was an increase downstream in NO_3^- and NH_4^+ (Camargo *et al*., 2005).

The Hume Dam on the Murray River, Australia, directly influences the downstream water chemistry on a seasonal basis (Baldwin *et al*., 2010). Lake Hume acted as a source for carbon, phosphorus and iron; conversely, it was a sink for manganese. Baldwin *et al*. (2010) calculated that 9 tonnes of nitrogen was exported from the dam, whereas over 50 tonnes of nitrogen had entered the reservoir. It was clear that in this particular circumstance, the reservoir had a significant role in altering the nitrogen load in the river (Baldwin *et al*., 2010). The N transformations were thought to include the transformation of nitrate to ammonia by dissimilatory nitrate reduction and denitrification (Baldwin and Williams, 2007; Baldwin *et al*., 2010).

3.1.4 Current Reservoir Nutrient Models

Much of the current information on reservoirs comes from modeled systems. An example would be the Nitrogen Retention in Reservoirs and Lakes (NiRReLa) model, designed for the estimation of global nitrogen removal in reservoirs (Harrison *et al*., 2009). The model estimates that reservoirs have the potential to remove 33% of the total nitrogen in lentic systems at roughly 6.6 Tg N/year, globally.

Another model was developed by Tomaszek and Koszelnik (2003) specifically for the Solina Reservoir and Rzeszów Reservoir in Poland. It was designed to predict the retention of nitrogen within the reservoirs. Though they had some success with the capabilities and accuracy of the model, the authors suggested that the model may be unsuitable for other reservoirs and that differences in parameters such as flow, residence time, P, R, etc., would likely alter the effectiveness of the model, as they would each alter the nitrogen retention potential (Tomaszek and Koszelnik, 2003).

A model introduced by Gergel *et al*. (2005) was used to simulate flood plain responses in order to determine natural annual nitrogen cycling versus an area affected by impoundment. They explained the possibility that the presence of a dam may reduce the efficiency of biological processes which aid in the water quality maintenance by returning nitrogen to the atmosphere. In particular they were concerned with the loss of the floodplain as it is an important site for both N-retention and denitrification. In both tropical and temperate systems nitrogen seems to be able to stay in a floodplain, but dam presence decreases flood peaks and thus both the volume and frequency of overbank flow decreases (Gergel *et al*., 2005).

3.1.5 Management Practices

The many examples given above illustrate not only reservoir-to-reservoir differences but seasonal differences as well. Interacting with these differences would be the management of the dam structure and reservoir water retention (Watts, 2000).

It has been suggested that increasing retention time, particularly during dry seasons, results in higher concentrations of some water quality parameters, particularly N and P (Ideva *et al*., 2008). This variation in retention time has often made reservoirs act as either sources (dry years) and sinks (wet years) for nutrients (Ahearn *et al*., 2005). During extreme drawdown events, reservoir water quality may be poor, with the potential for bloom formation in the reservoir (Naselli-Flores, 2003; Baldwin *et al*., 2010).

Dam operation also appears to be an important factor downstream; changes in discharge that occur suddenly (pulse flow or daily variation) will have downstream effects (de Oliveira Naliato *et al*., 2009). In the opposite case, too much regulation of flow may also have negative repercussions (Ideva *et al*., 2008).

3.1.6 A Day in the Life of the Grand

A description of this study can be found in section 2.1.6. In addition to the previously described methods and results, several other parameters were investigated and are reported here including NO_3^-, NH_4^+, and N_2O (Figure 3.1).

Once of the most striking differences is between the concentrations of NO_3^- for DLG 1 and DLG 2. The DLG 1 occurred on June 14, 2007. There was an increase of about 1 mg/L from what was entering the reservoir to what was measured downstream. In comparison, for the DLG 2 (September 7, 2007), the concentration of NO_3^- decreased slightly downstream of the reservoir relative to what was entering. NH_4^+ was higher downstream of the dam for both DLG1 and DLG2. For both the DLG1 and DLG2 no large differences were seen for N_2O in comparing upstream and downstream of the dam.

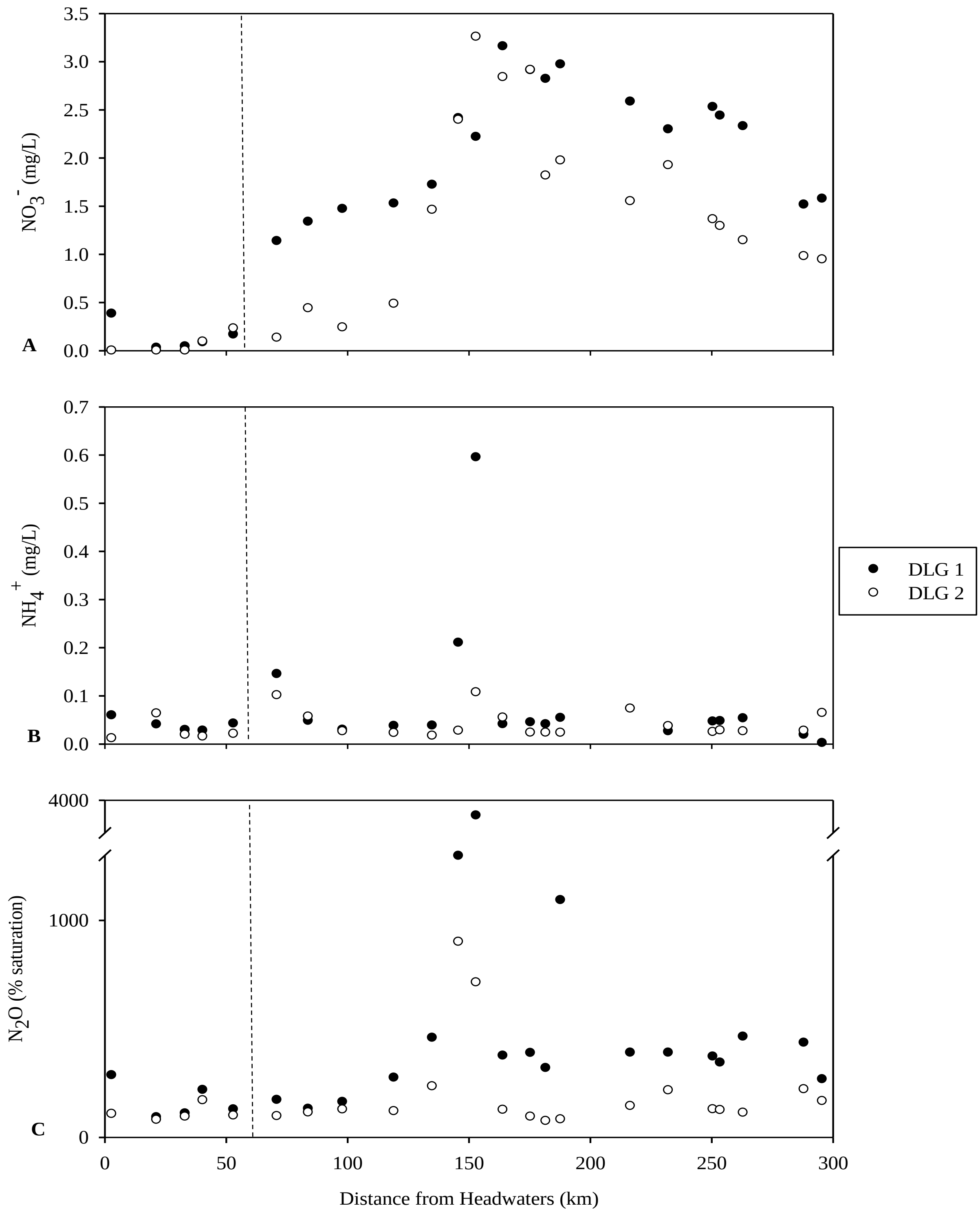

Figure 3.1: A) NO_3^-, B) NH_4^+ (mg/L), and C) N_2O (% saturation) from A Day in the Life of the Grand; June 14, 2007 (DLG1) and September 5, 2007 (DLG2) early afternoon samplings (Rosamond, 2013). The Shand Dam at Belwood Lake is indicated by the dashed line. The other large change observed (~150 km) is downstream of urban centres and their wastewater treatment plant outflows.

3.1.7 Conceptual Diagram

Reservoirs can influence the river downstream in many ways. However, it should be noted that these effects are not necessarily consistent in time. Stratification and actions such as draw down can alter downstream concentrations.

The following (Figure 3.2) is a conceptual diagram for what can potentially occur in the river-reservoir-river ecosystem. The upstream riverine zone is the initial source of the chemical species in question. Upon entering the transition zone, that chemical may be used, transformed, or flow through unchanged. Whatever the end result of the travel through the transition zone, the chemical then enters the lacustrine zone, where it will again either be used, transformed, or flow through unchanged; this zone may have a longer residence time and sedimentation of particulates may occur. The concentration of the chemical species in the plunge pool and downstream riverine zones is the result of the upstream inputs and reservoir processing (or lack thereof).

During spring runoff it is expected that NO_3^- will be high in the upstream riverine, the transition and at the reservoir surface. Downstream of the dam is expected to be low in NO_3^- as it is the hypolimnetic water from winter being released. In summer, the upstream riverine, transition and reservoir surface are expected to be low in NO_3^-. Below the dam would be receiving the stored water from the spring and would be high in NO_3^- for much of the summer and decreasing towards the season's end (Quirós, 2003). The fall is expected to have low concentrations of NO_3^- throughout the river-reservoir-river system, as there should be little inputs and low storage with a high flow-through rate.

Overall, there should be very little input of NH_4^+ from the upstream sources regardless of season. There should also not be large quantities at the reservoir surface. Late summer accumulation may occur at depth due to low oxygen conditions, the reduction of nitrate, and potentially compounds released from sediments (Jung, 2010). Downstream NH_4^+ would only be seen in late summer as the reservoir's hypolimnetic water is released.

Direction of Flow

NO_3^-	Upstream Riverine	Transition	Lacustrine (surface)	Plunge Pool	Downstream Riverine
Spring	[High]	[High]	[High]	[Lower]	[Low]
Summer	[Low]	[Low]	[Low]	[Higher]	[High]
Fall	[Low]	[Low]	[Low]	[Low]	[Low]

A

NH_4^+	Upstream Riverine	Transition	Lacustrine (surface)	Plunge Pool	Downstream Riverine
Spring	[Low]	[Low]	[Low]	[Low]	[Low]
Summer	[Low]	[Low]	[Low]	[High]	[Low]
Fall	[Low]	[Low]	[Low]	[Low]	[Low]

B

Figure 3.2: Conceptual model of changes in water quality parameters through a reservoir by season; A) NO_3^- and B) NH_4^+. The arrows indicate direction of flow. Spring is pre-stratification, summer is stratified, and fall is after stratification.

3.1.8 Chapter Goals

The primary goal of this chapter is to gain an understanding of the processes that affect dissolved nitrogen species both in and downstream of a man-made reservoir. It will focus on the changes seen daily, seasonally and annually, and determine whether the reservoirs are storing, transforming, utilizing, producing, or simply transporting downstream. The differences between two reservoirs will also be observed to determine whether they are acting similarly and synchronously.

Specific objectives that will be examined include: 1) whether the reservoirs act as NO_3^- sinks in spring as they are being filled for summer flow augmentation, then sources as the stored NO_3^- is released downstream; 2) if the hypolimnetic draw structures of the dams promote the export of NH_4^+ to the downstream during summer; 3) whether, in fall, the reservoirs have little impact on N-species due to the decrease in volume from draw down.

3.2 Methods

3.2.1 Sampling Procedures

Refer to section **2.2.1**

3.2.2 Field and Laboratory Analysis

Samples collected for NO_3^- were filtered through a 0.45-μm filter. The sample was then analysed using an auto sampler with a Dionex ICS-90 ion chromatograph. Precision was estimated at ±0.05 mg-N/L.

NH_4^+ samples were filtered through a 0.45-μm filter. The samples were then acidified with 10% H_2SO_4 to a pH of 5. The method used was based on the procedure by Solorzano (1969) with analysis completed on a Beckman DU 530 Spectrophotometer. Precision was estimated at ±0.005 mg-N/L.

Total dissolved nitrogen (TDN) was filtered through a 0.45 μm filter. Following filtration, samples were analysed on a Tekmar-Dohrmann Apollo 9000 equipped with an auto sampler. Precision was estimated at ±0.1 mg-N/L

Samples for N_2O were collected in 60 mL serum bottles with no headspace, then preserved with 0.15 mL of saturated $HgCl_2$ solution. Helium was added (5 mL) then the sample was equilibrated. Manual injections from the headspace were done on a Varian CP-3800 gas chromatograph. The precision for this procedure was approximately ± 0.4 μg N/L.

3.3 Results

3.3.1 2008 Seasonal Analysis

Reservoir Conditions and Precipitation

The 2008 sampling season had a high amount of precipitation, particularly from July to early September (GRCA, 2008). The reservoirs remained at their higher buffer volume or slightly above between June and October; this was similar for both Belwood and Conestogo (GRCA, 2008).

NO_3^-

The NO_3^- measured in the 2008 preliminary sampling season showed Belwood Lake with consistently lower concentrations than Conestogo Lake (Table 3.1). At Belwood, the downstream NO_3^- concentration was higher than at the reservoir surface. A seasonal difference was also observed at Conestogo Lake; in the summer the reservoir surface had higher concentrations than downstream, while the fall showed no difference. The Mallet River regularly had higher concentrations than the Conestogo River.

NH_4^+

In the summer, Conestogo Lake had higher NH_4^+ concentrations than upstream, but slightly lower than downstream (Table 3.1). Fall measurements at both lakes show low concentrations at all sites.

TDN

The TDN at Belwood was consistent at the reservoir surface and downstream, only a small change (0.3 mg/L) was observed in July (Table 3.1). In Conestogo Lake and downstream was similar to Belwood Lake, as it was consistent with a small change seen in August. Similar to the NO_3^-, the Mallet River had higher TDN concentrations than the Conestogo River.

N_2O

The N_2O measured during the preliminary sampling varied considerably, not only between sites but also between sampling days (Table 3.1). In September, Belwood Lake had a slightly higher N_2O than downstream. The October sampling found the highest N_2O below the dam; this was not seen further downstream (site 8).

For sites 3 and 4 at Conestogo, the N_2O was relatively low on both sampling occasions. The reservoir was much higher than upstream on both occasions, at 222% saturation in August and 449% saturation in September. The largest difference was downstream of the dam in August, where saturations of 1117% and 1159% were measured. In September, N_2O downstream was only slightly higher than in the reservoir.

Table 3.1: Summary of 2008 N Data for samples taken above, in and below Belwood and Conestogo Lakes. The dams are located between sites 6 and 7. Site 6 represents the surface of the reservoir; site 7 is below the dam and would be representative of the outflow.

Parameter	Reservoir	Date (2008)	Site 3 (Mallett River)	Site 4	Site 6	Site 7	Site 8
NO_3^- (mg N/L)	Belwood	Jul. 31	-	-	0.14	0.57	0.46
		Sept. 10	-	-	0.03	0.05	0.06
	Conestogo	Aug. 20	1.4	1.1	2.6	1.8	2.0
		Sept. 10	2.7	1.6	1.7	1.7	2.0
NH_4^+ (mg N/L)	Belwood	Sept. 10	-	-	0.01	0.04	0.03
	Conestogo	Aug. 20	0.02	0.03	0.11	0.13	0.19
		Sept. 10	0.05	0.05	0.01	0.03	0.03
TDN (mg N/L)	Belwood	Jul. 31	-	-	1.1	1.4	1.3
		Sept. 10	-	-	0.7	0.7	0.7
	Conestogo	Aug. 20	2.3	1.7	3.1	2.7	2.7
		Sept. 10	3.8	2.9	2.4	2.4	2.5
N_2O (% saturation)	Belwood	Sept. 10	-	-	138	111	105
		Oct. 17	-	-	85	370	159
	Conestogo	Aug. 20	128	154	222	1117	1159
		Sept. 10	194	135	449	515	501
N_2O (µg N/L)	Belwood	Sept. 10	-	-	0.34	0.28	0.26
		Oct. 17	-	-	0.26	1.16	0.5
	Conestogo	Aug. 20	0.33	0.38	0.5	2.78	2.89
		Sept. 10	0.58	0.36	1.09	1.27	1.23

3.3.2 2008 Vertical Profiles from Conestogo Lake and Belwood Lake

NO_3^-

A range in NO_3^- concentrations were observed in the reservoirs (Table 3.2). The lowest concentrations were measured at Belwood Lake in September. Here the concentration was only 0.3 mg/L at 12 m. For both August and September profiles at Conestogo Lake, the NO_3^- was higher than at Belwood Lake. In September, both reservoirs had higher concentrations in the hypolimnion than at the surface; conversely, in August, Conestogo Lake had a lower hypolimnion NO_3^-.

NH_4^+

The NH_4^+ concentration was higher at the surface in comparison to the 15 m depth for the August Conestogo Lake profile (Table 3.2); the September sampling showed the opposite, as NH_4^+ increased with depth. Belwood Lake had no change in concentration with depth as well as low concentrations (0.01 mg/L), downstream only indicated a slight increase.

TDN

Both of the profiles completed in September had no TDN changes from the surface to depth (Table 3.2). Belwood had the lowest overall concentrations reaching a maximum at 0.71 mg/L at the surface. Both Conestogo profiles had higher concentrations than Belwood; and only the August profile had a difference in concentration by depth, as it decreased slightly.

N_2O

The N_2O profiles for the reservoirs varied (Table 3.2). The August profile at Conestogo had the highest N_2O overall at depth and below the dam. The September profile at Conestogo Lake had higher N_2O than Belwood Lake, though neither had the drastic differences seen in August at Conestogo.

Table 3.2: Summary of N profiles from Belwood Lake (September 10, 2008) and Conestogo Lake (August 20, 2008 and September 10, 2008). Profiles were done in the deeper areas of the lacustrine zone as indicated by the depth. The downstream samples were taken below the dams.

Reservoir	Location	DO (mg/L)	Temperature (°C)	NO_3^- (mg N/L)	NH_4^+ (mg N/l)	TDN (mg N/L)	N_2O (% sat)	N_2O (µg N/L)
Conestogo	Surface	9.24	23.51	2.62	0.11	3.09	222	0.5
(Aug 20, 2008)	15 m	3.49	20.8	2.13	0.02	2.81	818	2.01
	Downstream	4.98	20.33	1.83	0.13	2.66	1117	2.78
Conestogo	Surface	7.65	21.03	1.71	0.01	2.42	449	1.09
(Sept 10, 2008)	14 m	5.85	20.87	1.75	0.06	2.45	516	1.26
	Downstream	7.4	20.8	1.66	0.03	2.44	515	1.27
Belwood	Surface	6.69	20.27	0.02	0.01	0.71	138	0.34
(Sept 10, 2008)	12 m	5.72	19.92	0.30	0.01	0.71	141	0.36
	Downstream	8.04	19.62	0.05	0.04	0.73	111	0.28

3.3.3 2009 Seasonal Analysis

Reservoir Conditions and Precipitation

Average rainfall was recorded for the 2009 sampling season, with above average in April, May and August (GRCA, 2009). Both Belwood and Conestogo reservoir volumes remained at or above their upper buffer between May and July, and above from September to November (GRCA, 2009).

NO_3^-

For both reservoirs each sampling event differed with regards to NO_3^-. In the spring and fall at Conestogo Lake (Figure 3.3A), the upstream and transition zone concentrations were high (over 3 mg/L); the summer concentrations were lower. The reservoir and downstream followed the upstream trend in some cases (May), and differed in others, as the NO_3^- both increased (June, August) and decreased (November). The plunge pool in all cases had a lower concentration than what was measured at the surface of Conestogo Lake.

Concentrations of NO_3^- at Belwood Lake (Figure 3.4A) were much lower than at Conestogo Lake. All upstream and transition zone measurements were under 1 mg/L. The reservoir and downstream in May, June and July was higher than 1 mg/L observed upstream. The downstream concentrations at Belwood Lake do not show the same decrease in concentration below the dam as seen at Conestogo Lake. An increase in NO_3^- was observed on several occasions (May, June, and November). In all cases, despite an increase or decrease, site 8 was never the same as site 7 despite only 250 m separating the two.

NH_4^+

NH_4^+ at sites 1 through 6 at Conestogo Lake were low, never exceeding 0.1 mg/L except for site 6 in June which still only reached 0.14 mg/L (Figure 3.3B). No noticeable differences occurred between the Conestogo River and the Mallet River. The most apparent change occurs within the plunge pool. On the three dates sampled, NH_4^+ was much higher just below the dam and at site 8. For all sampling days, the NH_4^+ at site 9 was much closer in concentrations to those found upstream and within the reservoir. The largest change was from 0.43 mg/L at site 8 to 0.04 mg/L at site 9, which is almost a 90% reduction in concentration.

Sites 1 through 6 at Belwood Lake were similar to Conestogo Lake as the NH_4^+ concentrations all remained under 0.1 mg/L regardless of month (Figure 3.4B). The changes are seen below the dam at site 7. Here May, November and July are similar to what was measured in the reservoir, as well as at site 8 and 9, all below 0.1 mg/L. The other three sampling trips show much higher NH_4^+ at site 7, with decreasing concentrations through sites 8 and 9.

TDN

The TDN at Conestogo Lake (Figure 3.3C) follows a similar pattern as was observed for NO_3^-; with spring and fall having higher upstream concentrations, with lower seen in the summer. Then, again, both an increase (June, August) and decrease (November) observed in the reservoir and downstream, while the other sites remained similar throughout (May).

The overall range of TDN at Belwood Lake in 2009 was small, ranging from 0.74 mg/L to 1.92 mg/L (Figure 3.4C). In May, June, and July there were increases in concentrations below the reservoir. On August 12 and November 11 the reservoir appears to be retaining the TDN. The August 27 sampling showed little variation between upstream, in reservoir or downstream.

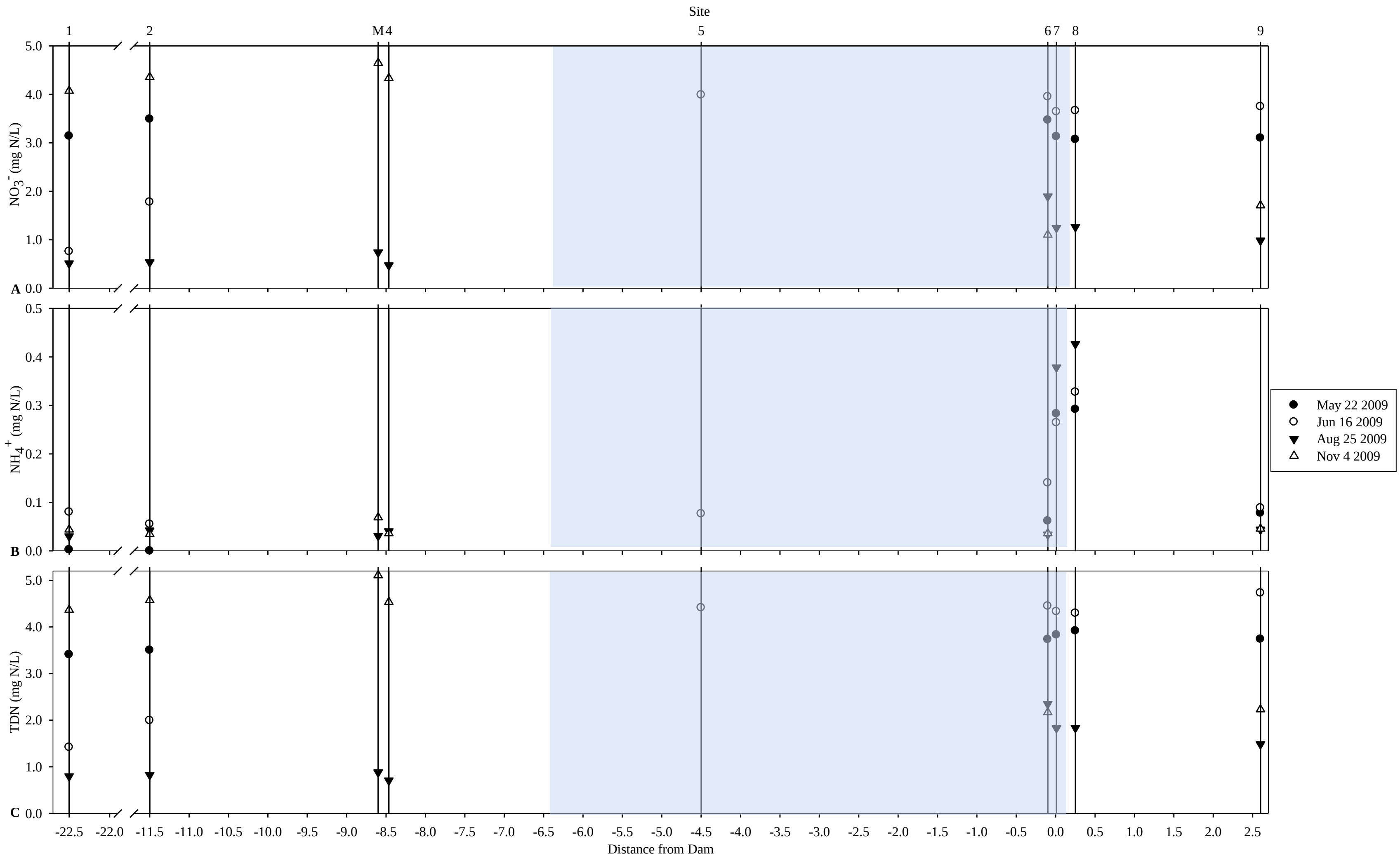

Figure 3.3: A) NO_3^- (mg N/L), B) NH_4^+ (mg N/L), and C) TDN (mg N/L) for the 2009 sampling season at Conestogo Lake. Sampling times were 11:40 – 13:45 (May 22), 11:00 – 18:00 (June 16), 11:00 – 15:00 (August 25), and 9:45 – 12:30 (November 4). The Conestogo dam is located between sites 6 and 7 or at 0 km, the lacustrine zone is shaded.

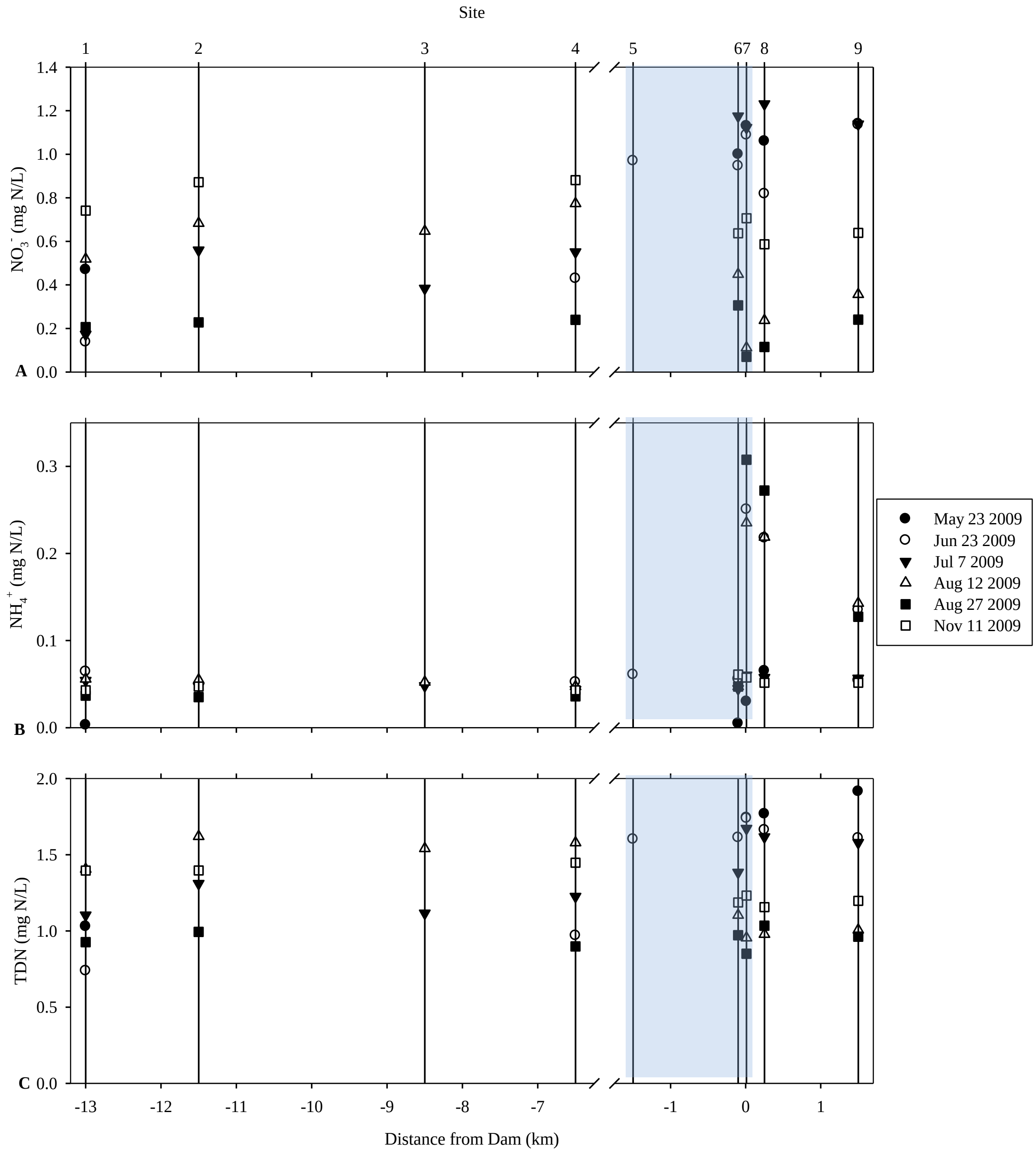

Figure 3.4: A) NO_3^- (mg N/L), B) NH_4^+ (mg N/L), and C) TDN (mg N/L) for the 2009 sampling season from Belwood Lake, 2009. Sampling times were 11:30 – 13:50 (May 23), 10:00 – 16:00 (June 23), 12:00 – 18:30 (July 7), 11:00 – 17:00 (August 12), 11:15 – 13:45 (August 27), and 9:45 – 12:30 (November 11). The Shand dam is located between sites 6 and 7 or at 0 km. The lacustrine zone of the reservoir is shaded; sites 3 and 4 represent the transition zone.

3.3.4 2009 Diel Analysis: Conestogo Lake, May 22

NO_3^-

The concentrations of NO_3^- remained high throughout the diel sampling (Figure 3.5A). Only sites 1 and 2 showed some decrease in concentration over the diel. Site 6 was stable and did not deviate from 3.4 mg/L. Of the downstream sites, site 7 had the highest concentration (3.86 mg/L for T3) as well as the highest concentrations overall. Site 8 and 9 both followed a similar path of increase as site 7.

NH_4^+

The concentration of NH_4^+ for sites 1, 2 and 6 remained stable throughout the day, though higher at site 6 (Figure 3.5B). Both sites 7 and 8 had much higher concentrations for all three measurements taken, with a peak occurring at T2. Site 9 did not follow the same trend as the other downstream sites, it decreased throughout the day ending at 0.05 mg/L.

TDN

For TDN the three downstream sites followed the same pattern of lower concentrations at T1 and steadily rising through T2 and T3 (Figure 3.5C). Sites 2 and 6 remained the same for the three sampling times. Site 1 decreased throughout the day from 3.57 mg/L for T1 to 3.02 mg/L for T3.

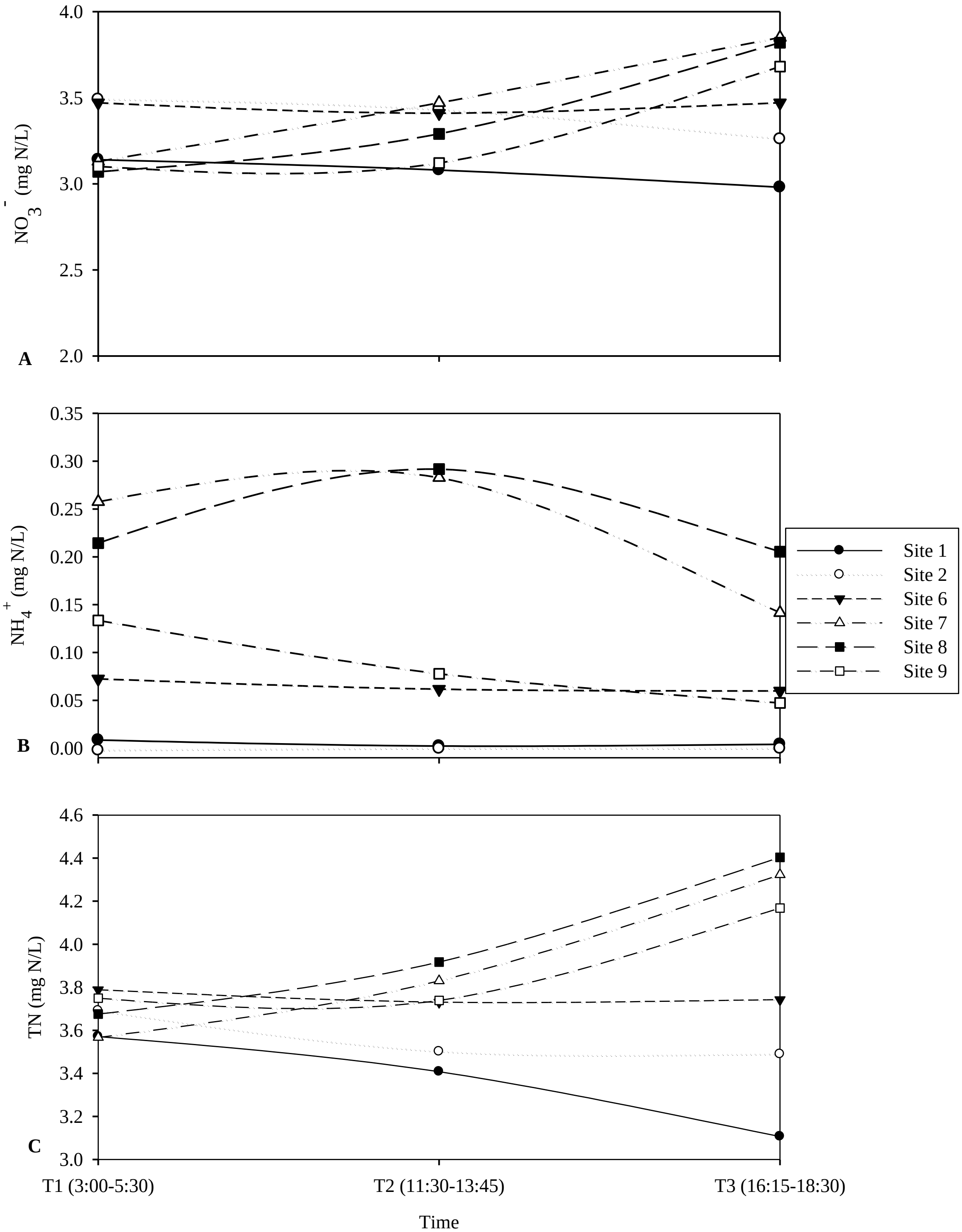

Figure 3.5: A) NO_3^- (mg N/L), B) NH_4^+ (mg N/L), and C) TDN (mg N/L) from the Conestogo Lake diel on May 22, 2009. The time is indicated on the x-axis as early morning (T1 3:00-5:30), noon (T2 11:30-13:45), and late afternoon (T3 16:15-18:30).

3.3.5 2009 Diel Analysis: Belwood Lake, May 23

NO_3^-

The NO_3^- for this diel sampling clearly show the difference between the upstream riverine and the reservoir and downstream riverine (Figure 3.6A). The low concentrations entering from the upstream as measured at site 1, ranged from 0.41 mg/L to 0.53 mg/L. The higher concentrations in the reservoir and downstream (sites 6-9) ranged between 1.0 mg/L and 1.9 mg/L. The distinction between these groups indicates that the downstream environment is influenced by processes occurring within the reservoir.

NH_4^+

Similar to the NO_3^-, the concentration measured at site 1 is much lower than what was measured downstream (Figure 3.6B). One difference though, is that the reservoir was much closer in concentrations to the upstream. Downstream from the dam the concentrations were higher and they followed a similar pattern to site 6. All but site 1 started with a higher concentration of NH_4^+ and declined during the course of the diel.

TDN

For all sites measured during the Belwood 2009 diel the TDN remained stable (Figure 3.6C). Similar to both NO_3^- and NH_4^+, there is a separation of data where the reservoir and downstream all have higher concentrations than that flowing into the reservoir from site 1.

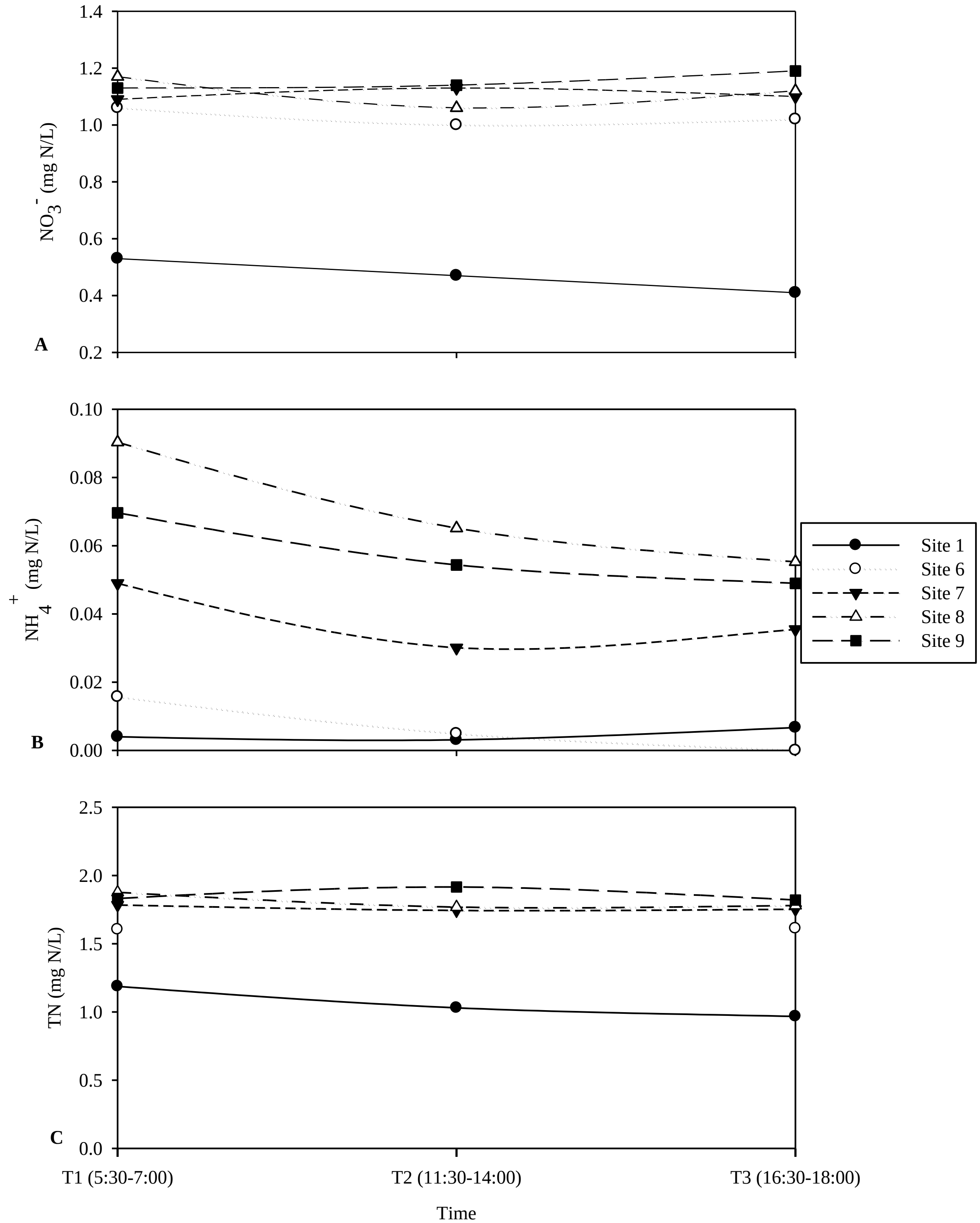

Figure 3.6: A) NO_3^- (mg N/L), B) NH_4^+ (mg N/L), and C) TDN (mg N/L) from the Belwood Lake diel on May 23, 2009. The time is indicated on the x-axis as early morning (T1 5:30-7:00), noon (T2 11:30-14:00), and late afternoon (T3 16:30-18:00).

3.3.6 2010 Diel Analysis: Belwood Lake, August 11

NO_3^-

The total range in NO_3^- for the 2010 diel was 0.16 mg/L to 0.43 mg/L (Figure 3.7A, 3.8A). Site 1 peaked at 0.43 mg/L at T3; it showed some movement but overall only displayed a 0.11 mg/L change over the day. Site 4 displayed almost a mirror image to that of site 1; beginning at a lower concentration, a slight increase for T2, followed by a drop at T3 then continually increased for T4-T6.

Overall lowest concentrations were found at site 7, though an increase at T6 likely indicated a response to a change in flow from the dam. Sites 8, 9 and 10 all peaked early (T1 or T2). Site 10 remained in steady decline for the duration of the diel. Sites 8 and 9 differed from each other in that site 8 began at a lower concentration before hitting a peak at T2; an increase in concentration was not seen until T6. Site 9 had a second peak at T4, followed by a small drop, then resuming a similar pattern to sites 7 and 8 for T6.

NH_4^+

As part of the 2010 August diel, NH_4^+ was measured (Figure 3.7B, 3.8B**)**. There was little variation over the diel period for sites 1 and 4, both falling in the range of 0.02 mg/L to 0.03 mg/L. The remaining sites had higher overall concentrations in comparison to the upstream as well as larger diel cycles. Site 7 had the highest concentrations throughout the entire sampling duration, reaching a maximum of 0.35 mg/L at the night sampling time point; it also had the largest range in concentration, changing by 0.14 mg/L over the day. Sites 8, 9, and 10 appear to follow a delayed pattern from the previous upstream site; as the maximums and minimums are offset by time. There is the general trend of a decrease in NH_4^+ concentration travelling downstream from sites 7 to 10. The large change at site 7 between T5 and T6 may have been caused by an increase in water released from the dam.

N_2O

The N_2O over the course of the diel changed considerably for some sites, and barely at all for others (Figure 3.7C, 3.8C). Most notably, at T3, site 7 had an increase to 4629% (from 558% at T2). Though not as high, T4 and T5 at site 7 were also high at 1210% and 1494%. Downstream, site 8 increased for the same three sampling intervals as site 7, though not to the same degree. Sites 1, 4, 9, and 10 all maintained similar N_2O saturations for the duration of the diel.

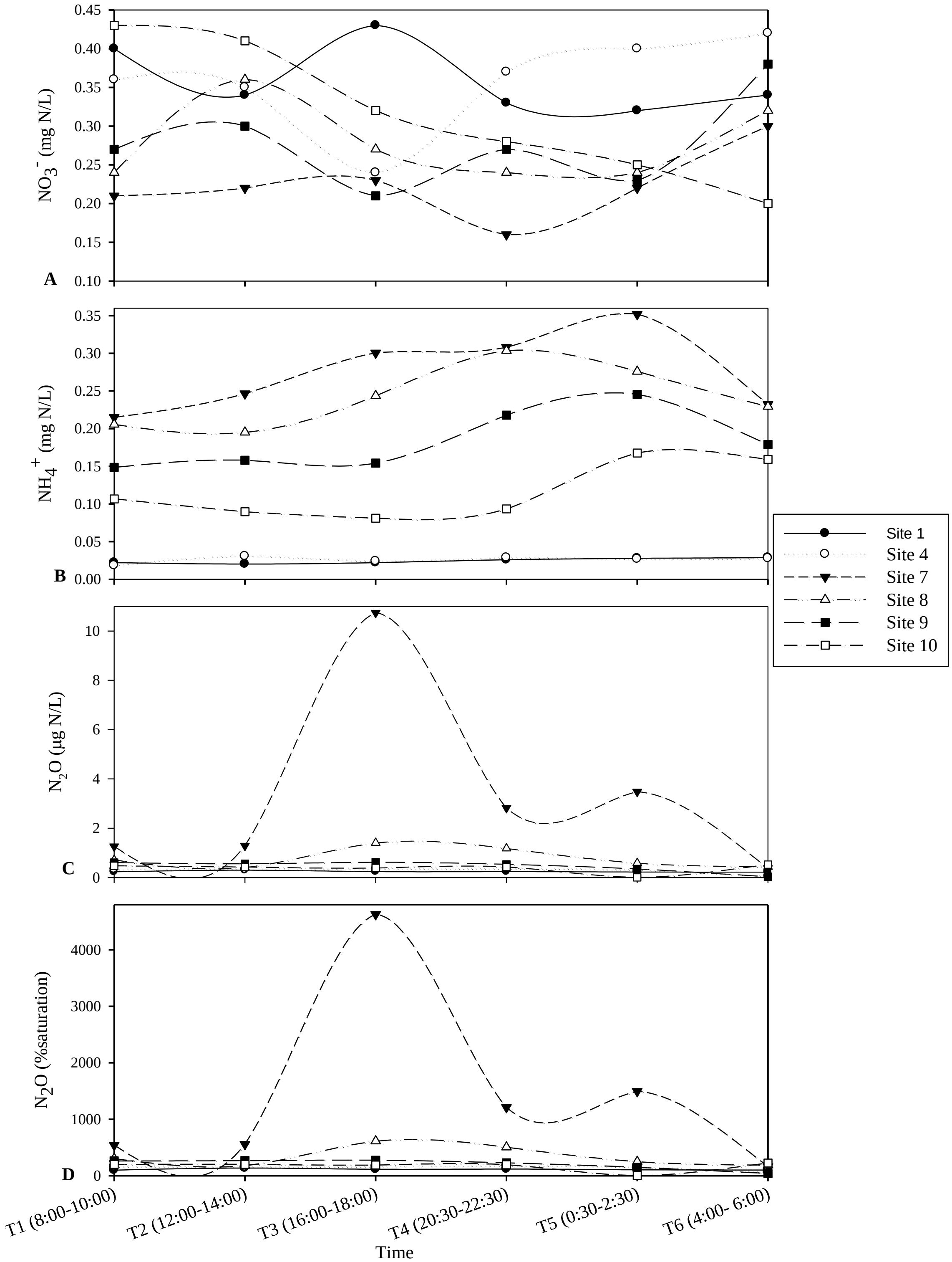

Figure 3.7: A) NO_3^- (mg N/L), B) NH_4^+ (mg N/L), C) N_2O (µg N/L); and D) N_2O (% saturation) from the Belwood Lake Diel on August 11, 2010. The time is indicated on the x-axis as morning (T1 8:00-10:00), noon (T2 12:00-14:00), late afternoon (T3 16:00-18:00), evening (T4 20:30-22:30), night (T5 0:30-2:30), and early morning (T6 4:00-6:00).

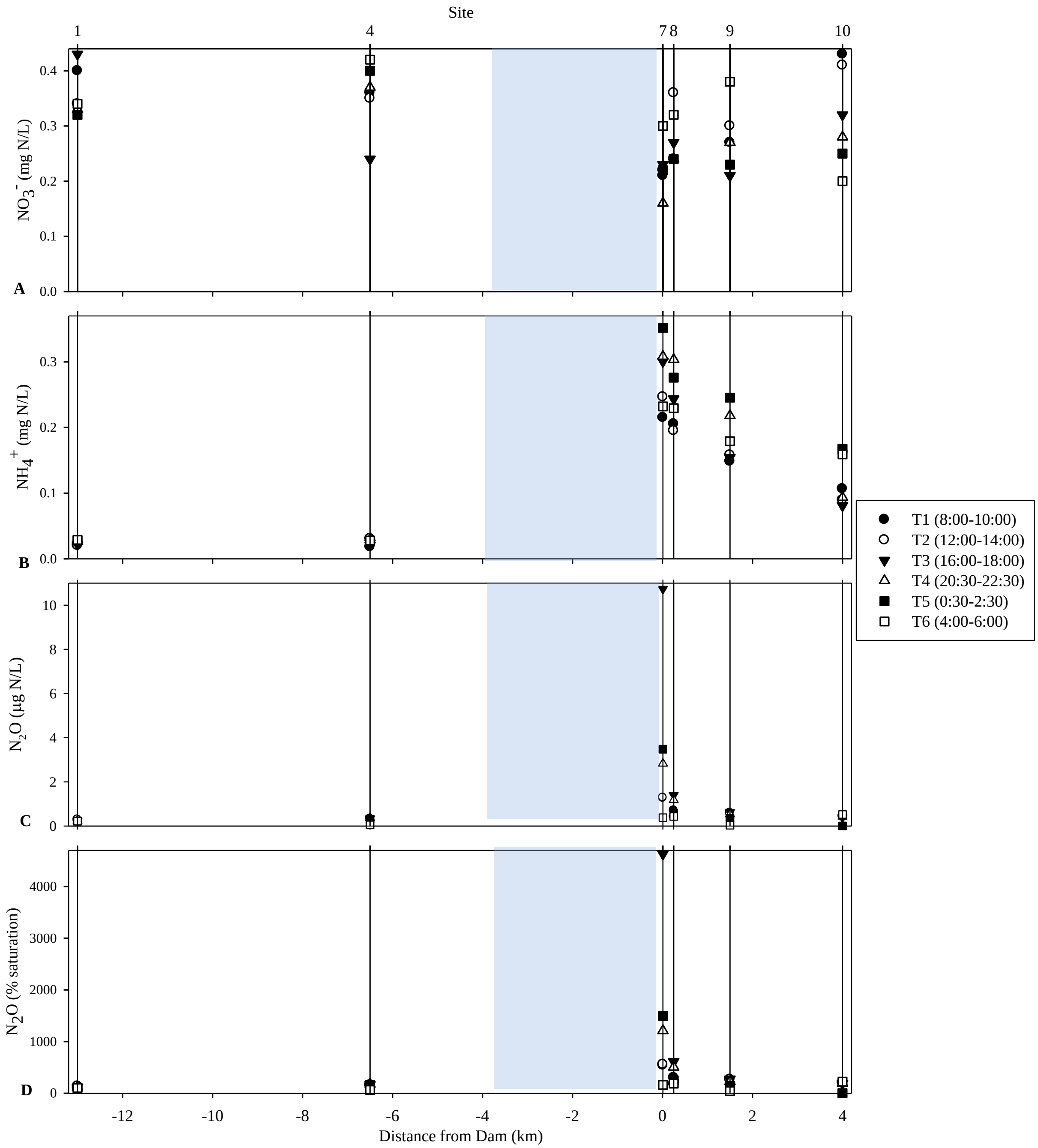

Figure 3.8: A) NO_3^- (mg N/L), B) NH_4^+ (mg N/L), C) N_2O (µg N/L); and D) N_2O (% saturation) from the Belwood Lake Diel on August 11, 2010 for each site all sampling times are included. Site 1 is upstream riverine, site 4 is the transition zone and the reservoir is shaded. Sites 7 – 10 are all below the dam.

3.4 Discussion

3.4.1 Alteration of Nitrogen Cycling Through Reservoirs

Belwood Lake and Conestogo Lake were observed over several years in an attempt to understand the nitrogen cycling in these reservoirs. Similar to other studies (Ahearn *et al*., 2005; Stanford and Ward, 1983), the reservoirs behaved as both a source and a sink for NO_3^-, depending upon the prevailing conditions. Both reservoirs were also shown to release NH_4^+ from the reservoir to the downstream, presumably from the reservoir. N_2O, a powerful greenhouse gas, was also measured downstream of the reservoirs on several occasions.

3.4.2 N-species Behaviors in Reservoirs

Weather conditions varied (sometimes dramatically) among years. The DLGs took place in 2007, one of the driest years on record (GRCA, 2010); 2008 was one of the wettest (GRCA, 2010). Both 2009 and 2010 were closer to the average amount of annual rainfall, but above average for the summer months (GRCA, 2010). These differences in precipitation required different reservoir management strategies in order to meet the operational standards. In 2007, the water in the reservoirs would have been released at a very low rate, leading to potentially longer residence times (e.g. 135-138 days in June, based on daily flow or monthly average). Conversely, in 2008, the water would have been allowed to flow through the reservoir at a much higher rate (e.g. 75 – 62 days retention time in July, based on daily flow or monthly average). The results from 2009 and 2010 may be more typical (e.g. 140 – 90 days in June 2009, based on daily flow or monthly average).

In 2009 at Belwood Lake, May, June and July all had low upstream NO_3^- concentrations, high reservoir concentrations and high downstream concentrations, indicating the reservoir was a source for NO_3^-; it is likely that the anticipated high spring concentrations upstream of the dam had already passed by the May sampling. Neither of the August samplings followed the pattern of the previous three excursions. The most likely explanation for this would be the rain event that occurred on August 09, 2009, where 51.9 mm of rain fell (Environment Canada historical data). Following this, there was an increase in upstream NO_3^- concentrations and a decrease in downstream concentrations – similar to what is expected in the spring. What may have aided in this result is that there was not an immediate increase in released flow, while the reservoir level increased to the upper bounds (Figure 3.9). Conestogo Lake, also in 2009, did not display the expected NO_3^- concentrations; with higher upstream and lower downstream. By June, the expected NO_3^- concentrations were observed, with lower upstream and higher downstream, indicating that the reservoir was a source at that time.

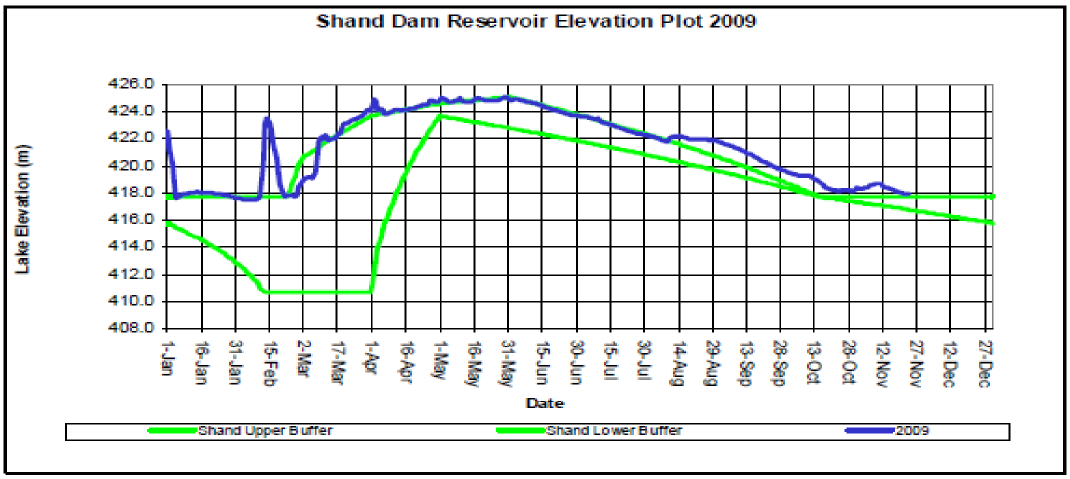

Figure 3.9: Volume of water contained within Belwood Lake compared with the upper and lower boundaries for management, indicating the rain event on August 09 (GRCA, 2009).

These results compare well with what others have found; the Mokelumne River for example showed similar patterns (Ahearn *et al*., 2005). Though, while they indicate some of the high NO_3^- measured downstream was from what was stored in the reservoir from the rainy season, they believe that a greater proportion of it is likely coming from mineralization and nitrification processes; this was also the conclusion of a previous study in Colorado (Stanford and Ward, 1983). The study on the Mokelumne River showed that its reservoirs were sources or sinks in different years (Ahearn *et al*., 2005). The authors establish that the downstream NO_3^- is a product of both weather conditions and operational procedures.

Higher concentrations of NH_4^+ were consistently measured downstream of both dams during the summer months. With low concentrations upstream, as well as low concentrations at the reservoirs surface, it is likely that from June to September the NH_4^+ measured downstream was coming from the hypolimnion of the reservoirs. As there was no indication of large concentrations of NH_4^+ entering the reservoirs, upstream fertilization inputs were unlikely. This is similar to observations made at the sediment–water interface in several lakes in China (Gao *et al*., 2005). Though this study did not measure NH_4^+ downstream, their in-reservoir measurements showed production of NH_4^+ (Gao *et al*., 2005). During a study in California measurements for NH_4^+ both in and downstream of reservoirs were taken; all of the concentrations were very low and showed little variation (Henson *et al*., 2007). As indicated by the results from both Belwood and Conestogo Lakes, it is possible that there was no release of NH_4^+; but, the measurements from California may not have been close enough to the dam structure, as demonstrated by the occasional decrease in NH_4^+ between the dam and the downstream sites.

There were several occasions on which concentrations changed in the short distance between the dam and the downstream sites. This includes several instances of high plunge pool NH_4^+ concentrations that had decreased by the final site; but also, low plunge pool NH_4^+ concentrations that increased by the final site. This was true for both reservoirs, though not necessarily at the same time. It is possible that for the high to low circumstances that uptake is occurring or possibly discharge from the dam was variable.

Some of the observations made in November of 2009 at both reservoirs were unexpected. Originally, after the fall drawdown, it was expected that the reservoirs would behave much more like a river. This was true for NH_4^+ for both reservoirs. Conversely, NO_3^- was observed at very high concentrations upstream of Conestogo Lake; but then lower concentrations were measured below the dam, showing retention occurring in the reservoir. At Belwood Lake, the NO_3^- was high, but was consistent from upstream to downstream as expected.

The transition zone in reservoirs is often considered to be a 'hot-spot' for biogeochemical processes compared to other zones (Scott *et al*., 2009). This was true in the current study as well. The seasonal observations in both reservoirs varied with both high and low NO_3^- concentrations as expected. On August 10, 2010 a large algae bloom was present in the transition zone.

3.4.3 Reservoir Comparisons

This study has shown that despite many similarities (catchment, climate, size, etc.) between Belwood Lake and Conestogo Lake, their differences are great. Occasionally both reservoirs would follow a similar longitudinal pattern (e.g. June 2009 NO_3^-), but with Conestogo having much higher concentrations. Conversely, there were several instances in which there were no similarities between patterns (e.g. November 2009 NO_3^-). There were also notable seasonal differences (e.g. TDN 2009), where Belwood had more similar concentrations throughout, whereas Conestogo showed distinct differences between the spring and fall measurements and the summer. These observations add to the growing amount of evidence showing that policy development should not be based on conclusions from individual examples (Moss, 2008).

3.4.4 DLG vs. Current Study

The current study has shown some of the changes that can occur as a result of impoundment. Some of these were also evident in the "Day in the Life of the Grand" studies, though differences did occur.

One similarity between the two studies is the behavior of NO_3^-. In early summer the current study showed Belwood Lake retaining water from spring melt and later acting as a source of NO_3^- to the downstream. This was supported by the findings of the DLG1 (June 14). Also similar were the late summer/fall observations, as both studies show the reservoir to be a sink for NO_3^-.

Both the DLG1 and DLG2 showed increased NH_4^+ downstream of the reservoirs. This was also evident in the current study. The DLG site 6 is equivalent to my site 9. At this site NH_4^+ was within a small range (0.10 mg/L to 0.15 mg/L). As this study was more intense in the reservoir area, other changes were observed that were not visible during the DLGs. The sites closer to the dam showed even higher NH_4^+; thus there was some biological processing of NH_4^+ in the distance from the dam to the downstream site that was not evident in the DLG data.

N_2O measured during the DLGs showed very small changes downstream of the reservoir. The largest, during DLG1, was from 128% saturation above the reservoir to 172% saturation below. The N_2O recorded for this study was only slightly higher at the same site. However, a much great concentration was measured immediately below the dam, 4629% saturation. This is higher than what was measured downstream of the cities of Kitchener and Waterloo and their WWTPs in the DLG study. As was for NH_4^+, the DLG provided a wealth of information, but the location of the site downstream of the Shand Dam was not able to fully convey the impacts on the river.

3.4.5 Conceptual Model Comparison

The conceptual model that was presented in **3.1.7** was developed to aide in visualizing the predictions of how reservoirs alter nitrogen species based on season and zone. This predictive model was correct for both the upstream riverine and reservoir sites. An alteration that should be made is the addition of a column for further downstream sites; as biological processes between the plunge pools and downstream may be different than that between two downstream sites.

3.4 6 Production of Greenhouse Gases in Reservoirs

The production and subsequent release of greenhouse gases in reservoirs has been heavily studied, though most have only considered CO_2 and CH_4 (e.g., Duchemin *et al*., 1995). Most studies have only considered the short term effects of reservoir formation, accounting for the decomposition of the submerged vegetation, and have found a decrease in release over time (e.g. Galy-Lacaux *et al*., 1999).

The August 2010 Belwood Lake sampling found a large release of N_2O from the dam. This event has implications for the total effect of hydroelectric dams and impoundments in general on greenhouse gas emissions. This study has shown that N_2O from reservoirs could be a concern and perhaps a contributor to global climate change. In addition, it shows that although CO_2 and CH_4 may be the initial concerns, reservoirs may still be contributing to greenhouse gas production well after construction. These findings also suggest that the previous studies done on reservoirs, dams, and other impoundments may not have fully observed the affect they were actually having. Many studies position their downstream sites a significant distance from the dam (e.g. Baldwin *et al.*, 2010) and so potentially causing their study to miss some events or simply miss degassing in general over that distance. This may further lead to questioning the usage of hydroelectric dams as an alternative energy source.

3.5 Conclusions

This study has given insight into how different N-species behave in the river-reservoir-river ecosystem. The hypothesis that the reservoirs are a sink for NO_3^- in the spring may still be viable despite that it was not observed here. The influx of the NO_3^- from spring runoff may have occurred earlier than the first sampling excursions. The reservoirs behaved as a source of NO_3^- during the summer, though large rain events (August 2009) may have disrupted this behavior. Unexpectedly, Conestogo Lake was a sink for NO_3^- in the fall, having high concentration entering from upstream; Belwood Lake had little change. Both reservoirs were a source of NH_4^+ on several occasions, despite no inputs from upstream, indicating that there may have been NH_4^+ released from the reservoir sediments. The NH_4^+ downstream of the dam at both reservoirs tended to be utilized by the furthest sampling station. A large amount of N_2O was observed below the reservoirs; by the final site, it was no longer detected. This indicates that it may have been lost to the atmosphere.

The observations made during the DLG and the 2008 sampling may have represented the extremes between a wet and dry year. The results from 2009 and 2010 may be more typical. Though close to the current average, that may change as global climate models are predicting more extreme weather events which may lead to greater frequency of both drought and flood. Understanding how a reservoir functions under these circumstances will help prepare for the weather to come.

4.0 Summary and Future Directions

4.1 Summary

Creation of impoundments is a common means of flood control, flow management, and hydroelectricity production that alters the natural hydrologic cycle of a watershed. For this study, two reservoirs in the Grand River watershed were studied: Belwood Lake on the Grand River and Conestogo Lake on the Conestogo River. The focus of this study was to determine what effects that these reservoirs, both of which release water from depth, have on the downstream dissolved oxygen and nitrogen dynamics.

The river-reservoir-river ecosystem is complex, with many processes and effects being interconnected. The major aspects considered in this study were: filling and drawdown, precipitation volume, flow, residence time, aeration, and management practices. Each of these will alter the seasonal, chemical and physical factors on which normal river health relies (Henson *et al*., 2007).

The concentration and daily cycle of DO in both the Grand and Conestogo Rivers was altered by the presence of impoundments. Operational procedures for the dams include filling in the spring to reduce the peak flow and provide storage for flow augmentation through the summer. This results in a long water residence time, which can allow for stratification to develop. The hypolimnetic water released from the dam was often (but not always) cooler than what was found upstream or at the reservoir surface. Downstream DO was variable during the summer season for all three sampling years. In some instances DO downstream was much lower than what was measured upstream or in the reservoir; on other occasions the dams were providing reaeration to the outflowing water increasing the DO to levels similar or higher than measured in the reservoir or upstream. After drawing down the reservoirs in fall, there was little effect on the DO downstream but the reservoirs retained enough heat that the reservoirs and downstream were warmer than the inflowing water.

Weather varied from year to year during this study. Dry years tended to show stronger stratification and thus the impoundments had a larger impact on the downstream river by releasing hypolimnetic water very low in DO; whereas in wetter years the water released was much higher in DO. Though this was generally true, it should also be noted that the presence or absence of thermal stratification was not always an indicator of DO concentrations within or downstream of the reservoir.

The impoundments at Belwood Lake and Conestogo Lake altered the rivers' nitrogen cycling; this was done both physically and biologically. These modifications varied seasonally and with the corresponding management protocols. The changing conditions caused both lakes to behave as either a source or a sink for NO_3^-, NH_4^+, TDN, and N_2O, depending on precipitation.

In spring, the reservoirs were filled with water high in NO_3^-. This water was released downstream through the summer, causing high NO_3^- downstream. During the late summer months the reservoirs were no longer behaving as a source for NO_3^-, instead they were sites of transformation and utilization. For NH_4^+, there was evidence that the reservoirs were behaving as a source during spring and summer. The N_2O measured below the reservoirs was high on occasion, but dissipated quickly downstream.

For ease of explanation and understanding, the observations for oxygen and nitrogen were separated in this thesis. Considering them together as part of a much larger ecosystem is also valuable. With lower DO often there was elevated NH_4^+, this was most apparent in August of 2009 at both lakes (Figures 4.1 and 4.2). In considering the previous discussion regarding the reaeration effect that a dam can have, it may be that linking DO with NH_4^+ may not be entirely possible, as is shown in Figures 4.1 and 4.2 during the month of June, as both DO and NH_4^+ are elevated.

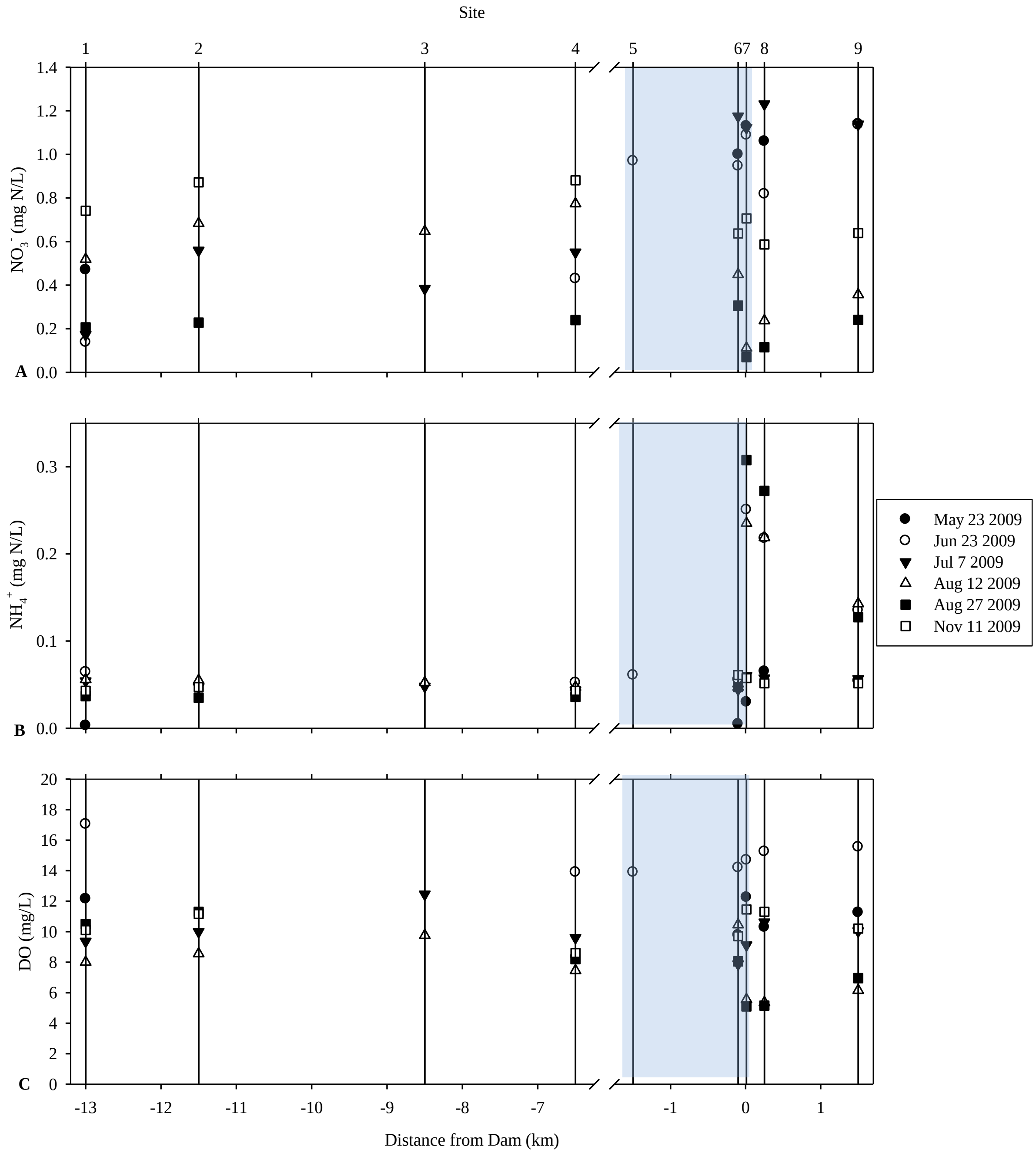

Figure 4.1: A) NO_3^- (mg N/L), B) NH_4^+ (mg N/L), and C) DO (mg/L) for the 2009 sampling season from Belwood Lake, 2009. Sampling times were 11:30 – 13:50 (May 23), 10:00 – 16:00 (June 23), 12:00 – 18:30 (July 7), 11:00 – 17:00 (August 12), 11:15 – 13:45 (August 27), and 9:45 – 12:30 (November 11). The Shand dam is located between sites 6 and 7 or at 0 km. The lacustrine zone of the reservoir is shaded; sites 3 and 4 represent the transition zone.

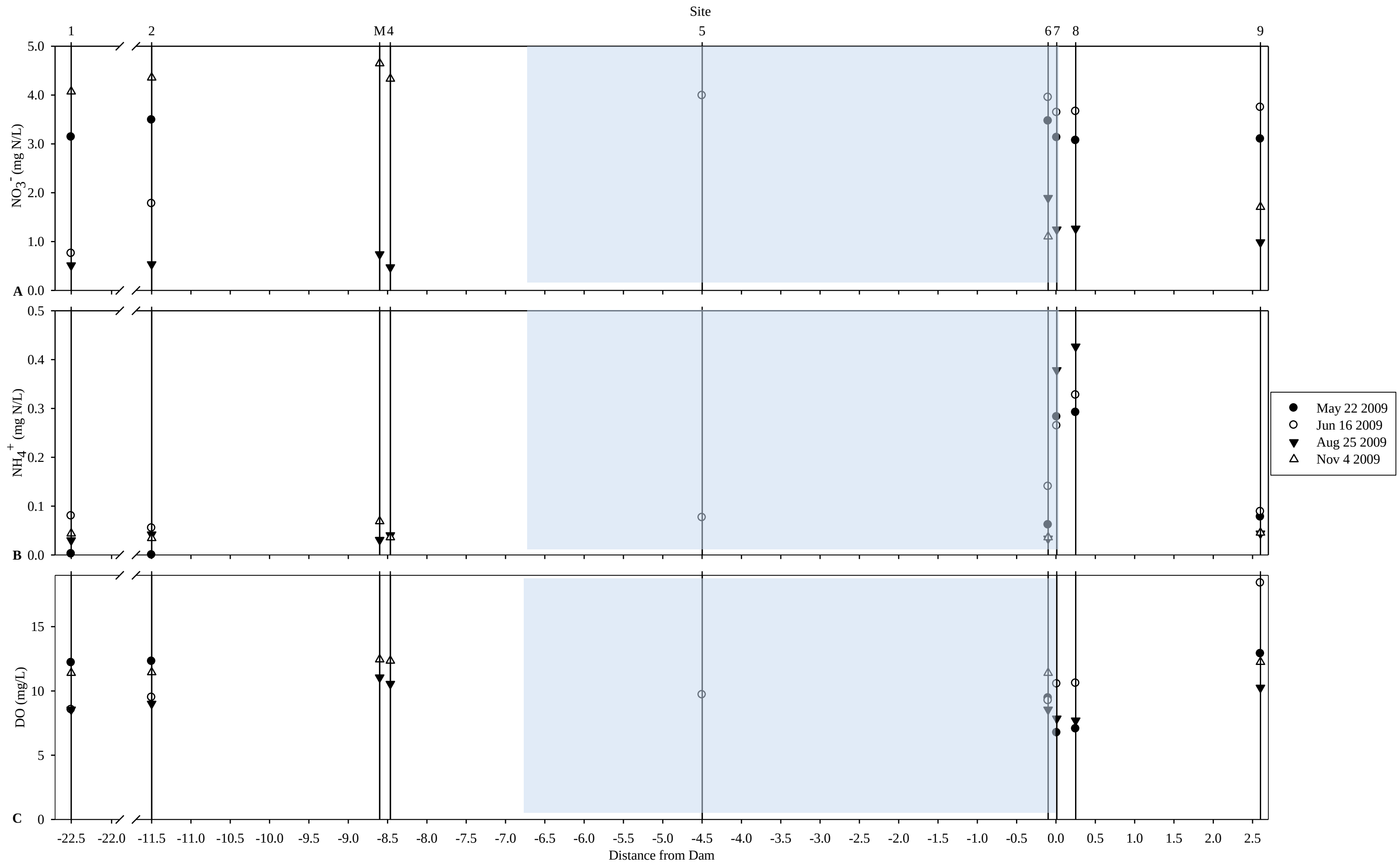

Figure 4.2: A) NO_3^- (mg N/L), B) NH_4^+ (mg N/L), and C) DO (mg/L) for the 2009 sampling season at Conestogo Lake. Sampling times were 11:40 – 13:45 (May 22), 11:00 – 18:00 (June 16), 11:00 – 15:00 (August 25), and 9:45 – 12:30 (November 4). The Conestogo dam is located between sites 6 and 7 or at 0 km, the lacustrine zone is shaded .

Greater concentrations of NO_3^- were seen at Conestogo Lake than at Belwood Lake. Though similar reservoirs are often compared and despite the common assumption that the changes caused by their inundation would be similar. I have found, in agreement with others (eg. Kalff, 2002; Ahearn *et al*., 2005; and McCartney, 2009) that the effects of regulation are river and reservoir specific. As such, any downstream disturbances and their magnitude can be unique. The various ecological effects will differ even between dams of similar size, often due to their residence time, which is influenced by season and operational management (Poff and Hart, 2002).

4.2 Concept Discussion

The River Continuum Concept or RCC (Vannote *et al*., 1980) emphasizes linkages between upstream and downstream; portraying a natural and gradual continuum in flow, water quality and species composition. As with this study, there are few rivers that remain without impoundments, which alter parameters and interrupt the continuum (McCartney, 2009).

The Serial Discontinuity Concept (SDC) was developed (Ward and Stanford, 1983) to accommodate for anthropogenic disturbances to rivers. The SDC assumes that the RCC is valid, but attempts to account for the presence of impoundments. One assumption that the SDC makes is that, other than the impoundment itself, there are no other disturbances; this is not the case for the rivers I studied; for example, there is a large agricultural influence particularly above Conestogo Lake.

The affected area downstream of a dam is termed the discontinuity distance. The river is assumed to recover eventually. The changes that occur and the physical distance vary based upon the parameter measured and stream order. One prediction that the SDC makes is that, in mid-order streams, reservoirs would be sinks for most nutrients. As seen in this work, that may not always be true. The tendency of reservoirs to act as sinks or sources may be related to season, weather and management, and should be considered at different time scales (season, year, 5 years, etc.).

The downstream recovery of a river affected by impoundment may not be as the SDC describes. In this study, the full recovery of the Grand River downstream of Belwood Lake was not observed due to the presence of another impoundment. To describe for the situation of more than one impoundment on a river, Barbosa *et al*. (1999) introduced the Cascading Reservoir Continuum Concept (CRCC). Even though in this study there were no measurements taken downstream of the second dam, it is possible that the combined effects they have is different than each on its own (Bergkamp *et al*., 2000).

4.3 Future Directions

Universality

As reservoir science continues, a major consideration will need to be the linking of both limnology, hydrology and engineering (Kennedy *et al*., 2003). For example, plunge pool and stilling basin design should not only to meet engineering criteria, but biological needs as well. Further research is needed to determine the best physical solution for re-aeration in the downstream environment. Governing bodies, scientists, and engineers need to develop and adapt a universal language regarding dams so that there is consistency in expectations of operation (Poff and Hart, 2002) and within scientific literature.

Individuality

Often impoundments are considered together for various reasons, particularly by governing bodies (Moss, 2008). This is a problem as no two reservoirs are the same. Each will have its own local geology, size, retention time, dam construction, etc. Based on physical characteristics alone, each reservoir should be considered individually (Whalen *et al*., 1982). Additionally, generalizations about downstream impacts should also be avoided (Ahearn *et al*., 2005) as the effects will differ in degree and kind (Kalff, 2002).

Seasonal Observations

Reservoirs cannot be studied from a single sampling; the seasonal variation alone is too large. Management practices add a layer of complexity as they change with season. Several studies of the same reservoir should be conducted, observing a yearly reservoir cycle; post-drawdown, post-filling, during stratification, etc.

Ecosystem Studies

Though I attempted to show the connectedness of these reservoir ecosystems, this study only considered a few aspects of these complex systems. This is true of many studies; e.g., the focus of hydroelectric dam impact studies has been on fish populations (Geen, 1974). Rather than focusing on a specific component an attempt should be made to study the entire ecosystem to fully understand the impacts that dams and reservoirs have (Bergkamp *et al*., 2000).

Increased Longitudinal Study

Much of the literature I reviewed for this study focused on small portions of the river-reservoir ecosystem, with very few addressing the upstream-reservoir-downstream connectivity. Though many recognized it – few showed it. Some even admit that downstream impacts are often assumed based on in-reservoir observations (de Oliveira Naliato *et al*., 2009).

More studies should be conducted in the plunge pool and under 5 km downstream. This study has shown that many processes occur in this reach, though most studies would fail to observe them.

In addition to a general increase in longitudinal sampling, increasing the downstream distance further may aid in assessing the total impact of an impoundment. As experienced in this study, many rivers have several impoundments. Bergkamp *et al*., (2000) has suggested that the cumulative effects of many dams may indeed differ from the summed impacts of individual impoundments. As full recovery would not occur between dams, the negative effects from the upstream impoundment would flow in to the downstream impoundment, compounding the impact. This potential complication should be quantified.

Management Practices

To minimize effects of a reservoir on the downstream environment, operational procedures must be considered, whether considering dams that currently exist or a new construction. New dams should be designed with the understanding that there is a direct relationship between limnology, reservoir and dam design, and management (Kennedy *et al*., 2003). Better communication between limnologists, engineers, and management agencies needs to develop so that as many factors as possible can be considered and a product with the least impact can be developed (Kennedy *et al*., 2003). As management influences both design and hydrological regime (Brierley and Harper, 1999), managers should consider alternative water level and flow management to minimize impact (Moss *et al*., 2009).

It is difficult to change the design of a dam that has already been built, but management practices can be altered. With better understanding of a reservoir's residence time, adjustments in flow and water levels could aid in reducing impact on the downstream environment (Moss *et al*., 2009). Also, accounting for seasonal biological changes may aid in developing new strategies for the holding and discharge of water. A strategy currently used that could be developed further is selective withdrawal to manage both downstream and reservoir water quality (Casamitjana *et al*., 2003). Maintenance of higher water levels throughout the year could be used to prevent sediments from being exposed and drying (Watts, 2000). An alternative management strategy could be to consider the flood pulse concept (intentional, controlled flooding) which places importance on lateral connectivity between rivers and their floodplains (Bergcamp *et al*., 2000); where the river provides nutrients and sediments to the floodplain and through this settlement there is the potential to improve water quality (Bergcamp *et al*., 2000).

Despite best practices for reservoir management, it must be conceded that ecological impacts will not be fully eliminated (Bednarek, 2001).

Future of Impoundments

While construction of dams continues, there is also a movement for the removal of dams. The opposition movement stems from several areas, the first being cost analysis. Many dams built before 1980 have become structurally compromised and the cost of maintenance and repair are great (Poff and Hart, 2002). These structures can be unstable and a danger to downstream locations. The other main driver for dam removal movements is for stream and river restoration (Bushaw-Newton *et al*., 2002).

Just as uncertainty exists in the construction of a dam, as much or more exists with removal (Poff and Hart, 2002). There have been few studies on environmental responses to dam removal (Hart *et al*., 2002). A factor contributing to the lack of information is privately owned dams. A survey found that only 54% of hydroelectric dams in the United States have monitoring programs, and of those most only included presence or absence of fish species (Bednarek, 2001).

Regardless of the long-term benefits, which may take 10+ years to realize (Bushaw-Newton *et al*., 2002), dam removal is a disruptive process (Stanley and Doyle, 2003). Removal disturbs the ecosystem that has developed and begins the development of a new one with its own chemical, biological and physical processes (Stanley and Doyle, 2003).

Each new removal should provide information regarding the practice (Babbitt, 2002). As this thesis and other studies have demonstrated, the great variability in reservoir ecosystems dictates that each removal should be considered individually (Whalen *et al*., 1982) as one success or failure may not necessarily be applied to another circumstance (Babbitt, 2002).

Though the construction of dams in the western world has slowed in recent years; the same cannot be said for developing countries (Kennedy *et al*., 2003). These areas tend to not be well studied (Kennedy *et al*., 2003). New impoundments will tend to be larger and more remote as most suitable locations have already been used (Malmqvist and Rundle, 2002). Some have suggested an ethical code of conduct to ensure that environmental concerns are adequately addressed and human rights respected (McCartney, 2009).

As the world moves forward with the alteration of the hydrological cycle, it must be realized that these anthropogenic changes are not fully understood but are having effects at regional, national, and global scales (Rosenberg *et al*., 2000).

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Appendix A: Chapter 1 Supplementary Information

Table A1: Geographical coordinates for sites at Conestogo Lake

Conestogo Lake	Latitude	Longitude
Site 1	43° 49′ 6″ N	80° 37′ 13″ W
Site 2	43° 45′ 22″ N	80° 40′ 20″ W
Site 3	43° 44′ 18″ N	80° 41′ 47″ W
Site 4	43° 44′ 17″ N	80° 41′ 43″ W
Site 5	43° 42′ 29″ N	80° 42′ 45″ W
Site 6	43° 40′ 34″ N	80° 43′ 4″ W
Site 7	43° 40′ 30″ N	80° 42′ 54″ W
Site 8	43° 40′ 23″ N	80° 42′ 45″ W
Site 9	43° 39′ 17″ N	80° 42′ 6″ W

Table A2: Geographical coordinates for sites at Belwood Lake

Belwood Lake	Latitude	Longitude
Site 1	43° 50′ 41″ N	80° 17′ 49″ W
Site 2	43° 49′ 43″ N	80° 17′ 56″ W
Site 3	43° 48′ 22″ N	80° 18′ 31″ W
Site 4	43° 47′ 28″ N	80° 19′ 1″ W
Site 5	43° 45′ 1″ N	80° 20′ 42″ W
Site 6	43° 44′ 6″ N	80° 20′ 12″ W
Site 7	43° 44′ 2″ N	80° 20′ 13″ W
Site 8	43° 43′ 58″ N	80° 20′ 24″ W
Site 9	43° 43′ 29″ N	80° 20′ 38″ W
Site 10	43° 42′ 41″ N	80° 22′ 8″ W

Appendix B: Chapter 2 Data

Table B1: Conestogo Lake 2008 Data

Date	Site	Time	Temp (°C)	DO (mg/L) (YSI)	$\delta^{18}O\text{-}O_2$ (‰)
Aug. 20, 2008	3	11:45	19.33	11.93	17.36
	4	11:45	20.48	12.58	14.94
	6-0m	13:30	23.51	9.24	16.38
	6-1m		23.92	8.77	15.86
	6-2m		23.47	7.7	18.13
	6-3m		23.6	7.1	19.13
	6-4m		23.41	6.38	20.74
	6-5m		23.34	6.13	20.97
	6-6m		23.31	6.29	21.2
	6-7m		23.23	6.17	20.61
	6-8m		23.11	5.71	21.57
	6-9m		23.1	5.23	22.37
	6-10m		22.07	4.02	25.43
	6-15m		20.8	3.49	27.05
	7	17:00	20.33	4.98	19.78
	8	12:00	20.3	4.45	20.47
Sept. 10, 2008	3	9:48	14.1	8.8	21.98
	4	10:10	14.3	8.42	25.49
	6-0m		21.03	7.65	18.11
	6-0.5m		21.43	7.33	17.73
	6-1m		20.74	6.99	19.36
	6-1.5m		20.72	6.54	18.88
	6-3m		20.9	6.29	20.32
	6-5m		20.95	6.2	20.8
	6-8m		21	6.14	20.99
	6-11.5m		21	6	21.3
	6-13m		20.99	5.9	21.48

Table B1 Continued: Conestogo Lake 2008 Data

Date	Site	Time	Temp (°C)	DO (mg/L) (YSI)	$\delta^{18}O-O_2$ (‰)
Sept. 10, 2008	6-14m		20.87	5.85	20.99
	7	11:00	20.8	7.4	18.84
	8	11:30	20.69	7.7	18.94
Oct. 30, 2008	3	12:00	4	24.07	
	4	12:10	3.75	22.49	
	6	11:07	8.2	17.6	
	7	9:52	8.04	16.76	
	8	10:15	8.04	16.78	

Table B2: Belwood 2008 Data

Date	Site	Time	Temp (°C)	DO (mg/L) (YSI)	$\delta^{18}O-O_2$ (‰)
Jul. 31, 2008	6	14:30	23.60	9.20	18.82
	7	13:50	20.80	8.45	25.49
	8	14:10	21.00	8.92	24.49
Sept. 10, 2008	6-0m	17:30	20.27	6.69	18.40
	6-3m	19:00	19.95	5.90	
	6-8m	18:39	20.20	5.41	
	6-11m	18:24	20.03	5.90	
	6-12m	18:00	19.92	5.72	19.91
	7	19:40	19.62	8.04	25.49
	8	19:55	19.43	7.67	25.57
Oct. 17, 2008	6	7:50	13.36	15.83	
	7	8:10	13.03	15.38	
	8	8:30	12.98	15.12	

Table B3: Conestogo Lake 2009. Note the YSI was malfunctioning in early summer (shaded).

Date	Site	Time	Temp (°C)	DO (mg/L) (YSI)	DO (mg/L) (Winkler)	$\delta^{18}O\text{-}O_2$ (‰)
May. 22, 2009	1	5:30	14.83	8.01	8.00	24.68
	2	5:10	15.19	8.43	8.50	23.44
	6	4:20	17.33	8.61	8.55	22.22
	7	3:58	13.56	6.99	6.45	25.71
	8	3:30	13.56	6.84	6.50	26.32
	9	3:05	12.85	5.69	5.88	28.09
	1	13:45	18.76	9.60	12.20	15.24
	2	13:20	18.37	9.85	12.30	14.37
	6	12:45	16.90	8.39	9.45	20.26
	7	12:20	13.79	6.41	6.75	21.66
	8	12:00	13.98	7.22	7.05	20.76
	9	11:40	15.34	11.74	12.90	10.07
	1	18:25	21.34	6.19	12.10	13.42
	2	18:00	21.20	6.90	13.55	11.25
	6	17:25	18.56	6.71	9.00	20.58
	7	17:00	14.14	6.74	8.55	19.12
	8	16:45	14.01	6.52	7.33	21.95
	9	16:15	16.76	10.57	15.15	8.59
Jun. 16, 2009	1	10:45	19.95	1.06	8.55	
	2	11:15	21.01	1.34	9.50	
	5	12:45	20.50	5.37	9.70	
	6	14:00	19.61	6.11	9.25	
	7	19:00	14.47	7.38	10.55	
	8	18:30	14.50	7.44	10.60	
	9	18:30	18.04	10.30	18.40	

Table B3 Continued: Conestogo Lake 2009

Date	Site	Time	Temp (°C)	DO (mg/L) (YSI)	DO (mg/L) (Winkler)	$\delta^{18}O$-O_2 (‰)
Aug. 25, 2009	1	11:00	19.41	9.19	8.55	20.57
	2	11:45	20.56	9.78	9.00	18.75
	3	12:40	20.99	12.34	11.05	16.95
	4	12:20	21.45	11.6	10.55	17.56
	6	13:10	23.18	8.80	8.55	21.26
	7	13:50	20.90	8.64	7.85	22.29
	8	14:28	20.83	8.01	7.70	22.89
	9	14:50	22.52	10.81	10.25	11.47
Nov. 04, 2009	1	9:45	5.04	14.92	11.40	24.55
	2	10:05	5.56	15.11	11.45	23.73
	3	10:30	5.54	16.24	12.45	21.38
	4	11:00	5.73	16.34	12.35	22.40
	6	11:30	8.42	14.48	11.40	20.44
	9	12:30	8.23	15.47	12.25	21.66

Table B4: Belwood Lake 2009. Note the YSI was malfunctioning in early summer (shaded).

Date	Site	Time	Temp (°C)	DO (mg/L) (YSI)	DO (mg/L) (Winkler)	$\delta^{18}O$-O_2 (‰)
May 23, 2009	1	7:00	15.33	3.22	7.90	24.01
	6	6:50	16.18	2.63	9.40	18.61
	7	6:15	13.68	2.20	9.70	24.05
	8	6:00	13.54	1.65	9.45	24.54
	9	5:40	13.31	2.59	9.05	25.10
	1	12:50	18.45	3.99	12.10	14.49
	6	12:25	17.77	3.61	9.78	18.13
	7	12:00	14.95	3.76	12.25	16.30
	8	11:30	14.27	3.35	10.30	22.00
	9	13:50	14.96	3.42	11.25	19.17
	1	16:40	21.07	5.59	12.40	13.45
	6	17:00	18.96	5.40	10.80	16.66

Table B4 Continued: Belwood Lake 2009. Note the YSI was malfunctioning in early summer (shaded).

Date	Site	Time	Temp (°C)	YSI (mg/L)	Winkler (mg/L)	$\delta^{18}O-O_2$ (‰)
May 23, 2009	7	17:20	14.96	5.70	11.25	17.58
	8	17:45	14.02	5.45	10.05	23.44
	9	18:00	14.50	5.76	10.45	20.88
Jun. 23, 2009	1	16:00	28.21	4.25	17.05	14.25
	4	16:30	24.37	2.68	13.90	16.28
	5	14:00	24.19	3.12	13.90	19.05
	5-4m	14:30	19.95		13.25	20.93
	6-0m	15:30	24.69	3.94	14.20	19.22
	6-7m	15:00	15.83	1.61	12.05	23.90
	6-14m	15:15	17.85	2.28	10.25	26.43
	7	11:00	14.44	1.12	14.70	23.76
	8	10:45	14.76	1.18	15.25	21.99
	9	10:00	15.34	1.53	15.55	19.71
Jul. 07, 2009	1	12:25	17.6		9.36	18.28
	2	11:45	17.82		10.00	16.96
	3		18.71		12.45	14.98
	4	15:15	21.43		9.60	16.90
	4	15:30	19.54		4.45	
	6	16:54	20.64		7.87	20.59
	7	17:24	14.83		9.12	23.71
	8	18:00	14.9		10.62	24.13
	9	18:30	15.02		10.04	22.19
Aug. 12, 2009	1	10:45	20.08	8.93	8.00	23.48
	2	11:05	20.41	10.28	8.55	23.67
	3	13:00	21.91	12.37	9.75	20.09
	4	14:00	23.07	11.12	7.45	23.03
	6	15:00	23.82		10.45	22.19
	6-14m	15:15	21.38			22.21
	7	16:30	19.9		5.55	23.81
	8	16:45	20.1		5.35	22.50
	9	17:10	20.4		6.15	

Table B4 Continued: Belwood Lake 2009

Date	Site	Time	Temp (°C)	YSI (mg/L)	Winkler (mg/L)	$\delta^{18}O\text{-}O_2$ (‰)
Aug. 27, 2009	1	11:30	18.2	11.75	10.50	17.56
	2	11:45	18.87	12.76	11.30	16.53
	4	12:15	22.27	9.20	8.20	17.99
	6	12:40	22.36	8.68	8.05	16.71
	7	13:00	20.23	5.84	5.10	23.36
	8	13:15	20.35	5.66	5.15	22.49
	9	13:40	21.09	7.77	6.95	18.37
Nov. 11, 2009	1	9:45	3.88	16.32	10.10	
	2	10:15	4.34	17.54	11.15	
	4	10:45	3.55	16.46	8.60	
	6	11:10	7.04	15.70	9.70	
	7	11:30	7.04	15.44	11.45	
	8	11:45	7.09	15.88	11.30	
	9	12:15	7.35	16.20	10.20	

Table B5: Belwood Lake 2010 Diel

Date	Site	Time	Temp (°C)	YSI (mg/L)	Winkler (mg/L)	$\delta^{18}O\text{-}O_2$ (‰)
Aug.11, 2010	1	8:00	20.99	8.55	6.65	24.62
	4	8:20	23.77	12.17	11.15	15.19
	7	9:00	21.04	6.21	4.45	24.82
	8	9:20	21.04	5.73	4.10	24.69
	9	9:30	21.22	6.14	4.70	24.37
	10	9:45	20.65	6.44	4.80	26.05
	1	12:10	23.24	12.09	9.10	18.68
	4	12:40	25.64	13.14	12.60	16.82
	7	13:00	21.28	6.47	5.10	24.56
	8	13:15	21.83	6.43	5.25	23.56
	9	13:30	22.46	7.21	5.50	21.17
	10	13:40	23.97	9.32	7.60	20.45

Table B5 Continued: Belwood Lake 2010 Diel

Date	Site	Time	Temp (°C)	YSI (mg/L)	Winkler (mg/L)	$\delta^{18}O\text{-}O_2$ (‰)
Aug.11, 2010	1	15:45	26.58	13.07	10.25	16.73
	4	16:15	26.67	15.12	12.95	15.25
	7	16:30	21.01	6.19	4.80	24.74
	8	16:40	21.46	6.08	4.95	21.73
	9	17:00	21.94	6.45	5.00	22.61
	10	17:15	25.39	10.11	8.50	17.67
	1	20:50	25.56	9.28	7.45	22.62
	4	21:10	26.34	15.77	12.40	13.35
	7	21:45	20.82	5.94	4.25	24.96
	8	21:59	20.81	5.31	3.90	25.88
	9	22:24	20.67	5.00	3.65	28.18
	10	22:40	22.82	7.34	5.35	21.84
	1	0:26	24.14	7.24	5.60	26.10
	4	0:50	25.96	15.9	12.40	11.61
	7	1:21	20.84	5.82	4.40	24.80
	8	1:34	20.79	5.10	3.70	26.41
	9	1:59	20.78	5.01		26.12
	10	2:23	20.98	4.87	4.20	28.59
	1	3:59	23.45	6.80	5.50	26.86
	4	4:27	25.77	14.2	12.00	13.74
	7	5:04	20.98	9.30	6.20	26.24
	8	5:20	20.89	7.53		26.87
	9	5:45	20.76	7.15	5.35	27.70
	10	6:06	20.68	4.53	4.15	28.71

Appendix C: Chapter 3 Data

Table C1: Conestogo Lake 2008 Data

Date	Site	NO_3^- (mg N/L)	NH_4^+ (mg N/L)	TN (mg N/L)	N_2O (%)	Temp (°C)
Aug. 20, 2008	3	1.42	0.02	2.27	128	19.33
	4	1.08	0.03	1.73	154	20.48
	6-0m	2.62	0.11	3.09	818	23.51
	6-15m	2.13	0.02	2.81	222	20.80
	7	1.84	0.13	2.66	1117	20.33
	8	1.98	0.19	2.68	1159	20.30
Sept. 10, 2008	3	2.67	0.05	3.77	194	14.1
	4	1.62	0.05	2.87	135	14.3
	6-0m	1.71	0.01	2.42	449	21.03
	6-14m	1.75	0.06	2.45	516	20.87
	7	1.66	0.03	2.44	515	20.8
	8	1.95	0.03	2.49	501	20.69
Oct. 30, 2008	3				149	4.00
	4				113	3.75
	6				166	8.20
	7				168	8.04
	8				160	8.04

Table C2: Belwood 2008 Data

Date	Site	NO_3^- (mg N/L)	NH_4^+ (mg N/L)	TN (mg N/L)	N_2O (%)	Temp (°C)
Jul. 31, 2008	6	0.14		1.07		
	7	0.57		1.35		
	8	0.46		1.35		
Sept. 10, 2008	6-0m	0.03	0.01	0.71	138	
	6-12m	0.30	0.01	0.71	141	
	7	0.05	0.04	0.73	111	
	8	0.06	0.03	0.73	105	
Oct. 17, 2008	6				85	
	7				370	
	8				159	

Table C3: Conestogo Lake 2009

Date	Site	NO_3^- (mg N/L)	NH_4^+ (mg N/L)	TN (mg N/L)	N_2O (%)	Temp (°C)
May. 22, 2009	1	3.14	0.01	3.57		14.83
	2	3.49	0	3.69		15.19
	6	3.47	0.07	3.79		17.33
	7	3.13	0.26	3.57		13.56
	8	3.07	0.21	3.68		13.56
	9	3.10	0.13	3.75		12.85
	1	3.08	0	3.41		18.76
	2	3.43	0	3.50		18.37
	6	3.41	0.06	3.73		16.90
	7	3.47	0.28	3.83		13.79
	8	3.29	0.29	3.92		13.98
	9	3.12	0.08	3.74		15.34
	1	2.98	0	3.11		21.34
	2	3.26	0	3.49		21.20
	6	3.47	0.06	3.74		18.56
	7	3.85	0.14	4.32		14.14

Table C3 Continued: Conestogo Lake 2009

Date	Site	NO_3^- (mg N/L)	NH_4^+ (mg N/L)	TN (mg N/L)	N_2O (%)	Temp (°C)
May 22, 2009	8	3.82	0.21	4.40		14.01
	9	3.68	0.05	4.17		16.76
Jun. 16, 2009	1	0.76	0.08	1.42		19.95
	2	1.78	0.06	1.99		21.01
	5	3.99	0.08	4.41		20.50
	6	3.95	0.14	4.45		19.61
	7	3.64	0.26	4.33		14.47
	8	3.66	0.33	4.29		14.50
	9	3.75	0.09	4.73		18.04
Aug. 25, 2009	1	0.51	0.03	0.79		19.41
	2	0.53	0.04	0.82		20.56
	3	0.74	0.03	0.88		20.99
	4	0.47	0.04	0.71		21.45
	6	1.89	0.03	2.35		23.18
	7	1.25	0.38	1.83		20.90
	8	1.26	0.43	1.83		20.83
	9	0.98	0.04	1.48		22.52
Nov. 04, 2009	1	4.07	0.04	4.37		5.04
	2	4.36	0.04	4.57		5.56
	3	4.65	0.07	5.11		5.54
	4	4.33	0.04	4.53		5.73
	6	1.10	0.04	2.17		8.42
	9	1.71	0.05	2.23		8.23

Table C4: Belwood Lake 2009 Data

Date	Site	NO_3^- (mg N/L)	NH_4^+ (mg N/L)	TN (mg N/L)	N_2O (%)	Temp (°C)
May 23, 2009	1	0.53	0	1.19		15.33
	6	1.06	0.02	1.60		16.18
	7	1.09	0.05	1.78		13.68
	8	1.17	0.09	1.88		13.54
	9	1.13	0.07	1.83		13.31
	1	0.47	0	1.03		18.45
	6	1.00	0.01			17.77
	7	1.13	0.03	1.74		14.95
	8	1.06	0.07	1.77		14.27
	9	1.14	0.04	1.92		14.96
	1	0.41	0.01	0.97		21.07
	6	1.02	0.000	1.61		18.96
	7	1.10	0.04	1.75		14.96
	8	1.12	0.06	1.78		14.02
	9	1.19	0.05	1.82		14.5
Jun. 23, 2009	1	0.14	0.06	0.74		28.21
	4	0.43	0.05	0.97		24.37
	5	0.97	0.06	1.60		24.19
	5-4m	1.07	0.07	1.65		19.95
	6-0m	1.04	0.06	1.61		24.69
	6-7m	0.95	0.16	1.72		15.83
	6-14m	0.93	0.27	1.68		17.85
	7	1.09	0.25	1.74		14.44
	8	0.82	0.22	1.66		14.76
	9	1.14	0.14	1.61		15.34
Jul. 07, 2009	1	0.17	0.05	1.10		17.6
	2	0.56	0.05	1.31		17.82
	3	0.38	0.05	1.12		18.71
	4	0.55	0.04	1.23		21.43

Table C4 Continued: Belwood Lake 2009 Data

Date	Site	NO_3^- (mg N/L)	NH_4^+ (mg N/L)	TN (mg N/L)	N_2O (%)	Temp (°C)
Jul 07, 2009	6	1.17	0.05	1.38		20.64
	7	1.12	0.06	1.67		14.83
	8	1.23	0.06	1.62		14.9
	9	1.14	0.06	1.58		15.02
Aug. 12, 2009	1	0.52	0.06	1.40		20.08
	2	0.68	0.06	1.62		20.41
	3	0.65	0.05	1.54		21.91
	4	0.77	0.05	1.58		23.07
	6	0.45	0.05	1.10		23.82
	7	0.11	0.24	0.95		19.9
	8	0.24	0.22	0.98		20.1
	9	0.36	0.14	1.01		20.4
Aug. 27, 2009	1	0.21	0.04	0.93		18.2
	2	0.23	0.04	0.99		18.87
	4	0.24	0.04	0.90		22.27
	6	0.31	0.05	0.97		23.36
	7	0.07	0.31	0.85		20.23
	8	0.16	0.27	1.03		20.35
	9	0.24	0.13	0.92		21.09
Nov. 11, 2009	1	0.74	0.04	1.40	110	3.88
	2	0.87	0.05	1.40	117	4.34
	4	0.88	0.04	1.45	120	3.55
	6	0.64	0.06	1.19	140	7.04
	7	0.71	0.06	1.23	104	7.04
	8	0.59	0.05	1.16	146	7.09
	9	0.64	0.05	1.20	121	7.35

Table C5: Belwood Lake 2010 Diel

Date	Time	Site	NO_3^- (mg N/L)	NH_4^+ (mg N/L)	N_2O (%)	Temp (°C)
Aug.11, 2010	8:00	1	0.40	0.02	99	20.99
	8:20	4	0.36	0.02	158	23.77
	9:00	7	0.21	0.22	542	21.04
	9:20	8	0.24	0.21	303	21.04
	9:30	9	0.27	0.15	261	21.22
	9:45	10	0.43	0.11	200	20.65
	12:10	1	0.34	0.02	136	23.24
	12:40	4	0.35	0.03	156	25.64
	13:00	7	0.22	0.25	558	21.28
	13:15	8	0.36	0.2	179	21.83
	13:30	9	0.3	0.16	268	22.46
	13:40	10	0.41	0.09	201	23.97
	15:45	1	0.43	0.02	114	16.73
	16:15	4	0.24	0.02	164	15.25
	16:30	7	0.23	0.30	4629	24.74
	16:40	8	0.27	0.24	612	21.73
	17:00	9	0.21	0.15	274	22.61
	17:15	10	0.32	0.08	188	17.67
	20:50	1	0.33	0.03	118	22.62
	21:10	4	0.37	0.03	173	13.35
	21:45	7	0.16	0.31	1210	24.96
	21:59	8	0.24	0.30	504	25.88
	22:24	9	0.27	0.22	227	28.18
	22:40	10	0.28	0.09	193	21.84
	0:26	1	0.32	0.03	106	26.10
	0:50	4	0.4	0.03	154	11.61
	1:21	7	0.22	0.35	1494	24.80
	1:34	8	0.24	0.28	249	26.41
	1:59	9	0.23	0.25	148	26.12
	2:23	10	0.25	0.17		28.59

Table C5 Continued: Belwood Lake 2010 Diel

Date	Time	Site	NO_3^- (mg N/L)	NH_4^+ (mg N/L)	N_2O (%)	Temp (°C)
Aug 11, 2010	3:59	1	0.34	0.03	101	26.86
	4:27	4	0.42	0.03	63	13.74
	5:04	7	0.3	0.23	161	26.24
	5:20	8	0.32	0.23	185	26.87
	5:45	9	0.38	0.18	40	27.70
	6:06	10	0.2	0.16	223	28.71

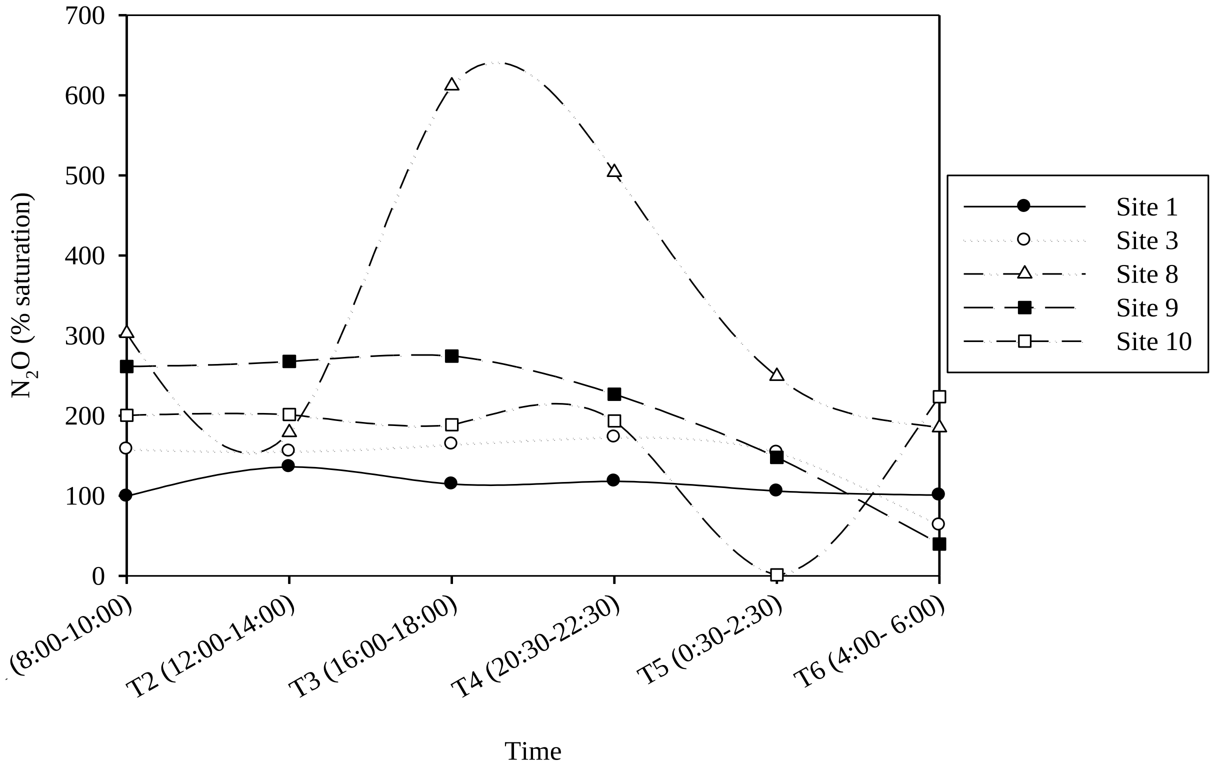

Figure C1: Belwood Lake N_2O from August 11, 2010 without site 7.

Appendix D: Unused Data

Table D1: Conestogo Lake 2008 Unused Data

Date	Site	pH	TSS (mg/L)	DOC (mg/L)	DON (mg/L)	CO_2 (%)	CH_4 (%)	Temp (°C)
Aug. 20, 2008	3	8.46	13.5	7.7	0.8			19.33
	4	8.44	10.5	6.1	0.6			20.48
	6-0m	8.67	4.3	5.2	0.4			23.51
	6-1m	8.75						23.92
	6-2m	8.6						23.47
	6-3m	8.52						23.6
	6-4m	8.45						23.41
	6-5m	8.42						23.34
	6-6m	8.41						23.31
	6-7m	8.4						23.23
	6-8m	8.33						23.11
	6-9m	8.28						23.1
	6-10m							22.07
	6-15m	7.8	16.0	5.9	0.7			20.8
	7	8.04	20.7	5.9	0.7			20.33
	8	8	23.9	5.7	0.5			20.3

Table D1 Continued: Conestogo Lake 2008 Unused Data

Date	Site	pH	TSS (mg/L)	DOC (mg/L)	DON (mg/L)	CO_2 (%)	CH_4 (%)	Temp (°C)
Sept. 10, 2008	3	7.56	4.9	6.2	1.0			14.1
	4	7.48	23.2	8.1	1.2			14.3
	6-0m	8.36	9.7	5.1	0.7			21.03
	6-0.5m	8.34						21.43
	6-1m	8.29						20.74
	6-1.5m	8.25						20.72
	6-3m	8.22						20.9
	6-5m	8.2						20.95
	6-8m	8.19						21
	11.5m	8.19						21
	13m	8.18						20.99
	6-14m	8.18	19.4	6.6	0.6			20.87
	7	8.35	11.3	5.5	0.7			20.8
	8	8.27	12.2	5.5	0.5			20.69
Oct. 30, 2008	3	8.21	3.4			2057	3930	4
	4	8.04	2.1			2199	2892	3.75
	6	8.43	13.7			2363	5177	8.2
	7	8.59	14.0			2279	4184	8.04
	8	8.35	17.3			2235	4559	8.04

Table D2: Belwood 2008 Unused Data

Date	Site	pH	TSS (mg/L)	DOC (mg/L)	DON (mg/L)	CO_2 (%)	CH_4 (%)	Temp (°C)
Jul. 31, 2008	6	10.4	6.0	8.2		2060	11638	23.60
	7	8.05	7.7	9.1		3196	32287	20.80
	8	8.24	7.1	8.3		3040	10513	21.00
Sept. 10, 2008	6-0m	8.32	6.9	8.4	0.7			20.27
	6-3m	8.23						19.95
	6-8m	8.26						20.20
	6-11m	8.34						20.03
	6-12m	8.33	20.9	8.5	0.4			19.92
	7	8.19	25.9	8.5	0.7			19.62
	8	8.25	14.1	8.3	0.6			19.43
Oct. 17, 2008	6							13.36
	7							13.03
	8							12.98

Table D3: Conestogo Lake 2009 Unused Diel Data

Date	Time	Site	pH	TSS (mg/L)	Chlorophyll a (ug/L)	Phaeopigments (ug/L)	DOC (mg/L)	Cl (mg/L)	SO_4 (mg/L)	TP (ug/L)
May. 22, 2009	5:30	1	8.68	10.5	1.2	1.6	5.7	18.5	4.0	39.4
	5:10	2	8.69	6.4	0.8	1.2	5.8	19.8	4.1	26.1
	4:20	6	8.82	3.0	0.2	0.6	5.1	18.4	3.4	37.8
	3:58	7	8.53	11.2	0.5	1.3	5.2	18.5	3.4	110.5
	3:30	8	8.49	10.5	0.7	1.3	5.1	18.4	3.3	87.5
	3:05	9	8.33	8.6	1.9	1.6	4.4	18.5	3.5	57.2
	13:45	1	8.74	6.6	0.8	1.2	6.0	22.1	4.1	40.4
	13:20	2	8.79	5.0	0.9	1.0	5.6	20.2	4.0	30.5
	12:45	6	8.74	2.4	0.8	0.8	4.8	18.5	3.2	27.0
	12:20	7	8.38	12.2	0.7	0.8	5.9	18.1	3.4	95.7
	12:00	8	8.31	10.8	1.5	3.1	5.2	20.0	3.3	106.7
	11:40	9	8.76	5.2	1.0	1.1	5.6	18.9	3.4	59.6
	18:25	1	8.81	6.8	1.0	1.0	6.3	19.0	4.0	32.3
	18:00	2	8.88	6.4	0.5	0.8	5.5	20.1	4.1	27.6
	17:25	6	8.78	4.8	1.1	1.1	5.4	18.6	3.3	53.8
	17:00	7	8.57	35.8	10.0	4.4	6.0	20.1	3.5	106.2
	16:45	8	8.45	9.7	1.0	1.3	5.5	19.9	3.4	110.3
	16:15	9	9.01	19.0	6.2	3.1	5.6	18.3	3.4	87.8

Table D4: Conestogo Lake 2009 June, August and November Unused Data

Date	Site	pH	TSS (mg/L)	Chlorophyll a (ug/L)	Phaeopigments (ug/L)	DOC (mg/L)	DON (mg/L)	Cl (mg/L)	SO_4 (mg/L)	SRP (ug/L)	TP (ug/L)
Jun. 16, 2009	1	8.67	15.6	2.1	1.9	7.5		25.0	3.9	57.7	
	2	8.62	5.9	1.1	1.5	7.1		27.4	3.8	38.4	
	5	8.69	2.4	2.0	0.2	5.9		18.6	3.2	30.5	
	6	8.61	1.3	1.7	0.4	6.1		18.7	3.2	26.8	
	7	8.47	10.5	4.4	1.7	5.9		18.5	3.2	102.2	
	8	8.53	3.8	1.0	0.9	5.6		18.3	3.3	74.5	
	9	8.98	17.7	4.9	2.9	6.2		19.6	3.5	106.6	
Aug. 25, 2009	1	8.26	17.4	2.8	3.7	8.4	0.5				34.5
	2	8.37	9.4			8.5	0.4				41.6
	3	8.62	2.9	1.1	2.9	6.3	0.4				21.7
	4	8.58	4.24	2.1	2.3	7.9	0.5				20.2
	6	8.61				5.5	0.6				26.1
	7	8.07	7.7	2.7	6.7	5.4	0.6				50.3
	8	7.96	6.0	1.4	3.1	5.5	0.6				67.1
	9	8.44	17.4	4.1	21.8	5.7	0.6				32.2
Nov. 04, 2009	1	8.21	9.0	0.8	1.5	6.6				21.4	
	2	8.3	7.1	0.7	1.4	6.1				20.9	
	3	8.3	6.5	3.1	2.2	5.0				19.6	
	4	8.32	6.0	1.1	1.5	6.3				20.8	
	6	8.5	3.0	25.3	9.7	6.3				18.3	
	9	8.5	10.378	21.3	5.7	6.2				20.0	

Table D5: Belwood Lake 2009 Unused Diel Data

Date	Time	Site	pH	TSS (mg/L)	Chlorophyll a (ug/L)	Phaeopigments (ug/L)	DOC (mg/L)	Cl (mg/L)	SO_4 (mg/L)	TP (ug/L)
May 23, 2009	7:00	1	8.64	3.3	1.4	2.2	9.5	10.4	2.5	34.4
	6:50	6	8.84	4.9	6.2	0.7	10.3	14.0	2.7	26.5
	6:15	7	8.62	8.2	0.7	1.1	10.0	14.5	2.8	58.3
	6:00	8	8.59	7.9	0.8	1.3	9.9	12.7	2.8	62.5
	5:40	9	8.44	9.1	0.9	1.7	10.2	14.6	2.9	57.7
	12:50	1	9	3.0	1.2	1.4	9.9	12.7	2.5	35.4
	12:25	6	8.8	5.2	8.4	0.3		14.3	2.7	37.9
	12:00	7	8.81	10.6	5.0	3.2	9.6	13.7	2.8	73.2
	11:30	8	8.64	6.2	1.0	1.2	9.6	15.9	2.8	47.4
	13:50	9	8.75	4.6	0.9	1.2	9.9	13.1	2.9	64.5
	16:40	1	9.01	4.0	1.1	1.2	10.0	9.3	2.3	35.9
	17:00	6	8.63	6.8	13.0	0.8	9.4	14.2	2.7	49.6
	17:20	7	8.65	14.0	4.2	2.6	9.9	14.0	2.7	76.6
	17:45	8	8.41	7.0	1.0	1.5	9.4	13.2	2.9	55.0
	18:00	9	8.49	6.5	1.0	1.2	9.8	13.5	2.9	57.7

Table D6: Belwood Lake 2009 June and July Unused Data

Date	Site	pH	TSS (mg/L)	Chlorophyll a (ug/L)	Phaeopigments (ug/L)	DOC (mg/L)	DON (mg/L)	SRP (ug/L)
Jun. 23, 2009	1	9.01	3.6	3.5	3.0	10.4		7.5
	4	8.61	4.4	15.1	1.2	9.8		12.2
	5	8.62	1.8	4.4	1.7	10.2		5.1
	5-4m	8.57	2.1	5.8	0	9.8		
	6-0m	8.78	2.6	5.3	1.1	10.0		5.0
	6-7m	8.15	3.6	0.2	0.7	9.5		
	6-14m	8.28	2.5	0.9	0.5	8.9		
	7	8.34	2.2	0.3	0.5	10.3		19.2
	8	8.35	4.2	0.6	0.8	9.6		15.1
	9	8.38	3.6	0.8	1.2	9.8		12.8
Jul. 07, 2009	1	8.45	4.0	4.5	7.7	11.6	0.9	
	2	8.57	2.2	5.0	7.4	11.3	0.7	
	3	8.51	3.4	3.8	5.5	11.6	0.7	
	4	8.73	9.4	20.9	6.1	10.6	0.6	
	6	8.62	8.8	11.6	2.8	10.0	0.2	
	7	7.77	6.1	0.7	1.5	9.9	0.5	
	8	8.12	6.0	1.5	2.6	9.8	0.3	
	9	8.23	5.2	1.3	2.0	9.7	0.4	

Table D7: Belwood Lake 2009 August Unused Data

Date	Site	pH	TSS (mg/L)	Chlorophyll a (ug/L)	Phaeopigments (ug/L)	DOC (mg/L)	DON (mg/L)	SRP (ug/L)
Aug. 12, 2009	1	8.29	11.4	2.5	2.7	12.4	0.83	16.64
	2	8.37	9.3	3.2	3.7	13.2	0.88	27.42
	3	8.41	7.2	2.7	2.5	12.8	0.84	12.32
	4	8.35	5.1	4.8	2.8	11.8	0.76	15.52
	6	8.73	6.7	12.8	1.5	9.9	0.61	10.62
	7	8.04	3.7	0.8	1.4	9.7	0.61	19.19
	8	8.22	4.0	1.1	1.3	9.2	0.52	19.97
	9	8.27	5.4	2.0	2.7	9.3	0.51	14.90
Aug. 27, 2009	1	8.36				11.7	0.68	
	2	8.45	1.9			11.5	0.73	
	4	8.54				10.5	0.62	
	6	8.58	8.6			9.4	0.62	
	7	7.97	5.1			10.1	0.47	
	8	7.91	6.1			10.1	0.65	
	9	8.07				9.5	0.59	

Table D8: Belwood Lake 2009 November Unused Data

Date	Site	pH	TSS (mg/L)	Chlorophyll a (ug/L)	Phaeopigments (ug/L)	DOC (mg/L)	SRP (ug/L)	CO_2 (%)	CH_4 (%)	Temp (°C)
Nov. 11, 2009	1	8.26	3.1	2.5	1.6	11.1	5.8	2621	1610	3.88
	2	8.35	4.0	8.7	5.0	10.8	9.4	2433	1475	4.34
	4	8.28	3.7	6.0	2.5	10.4	20.0	2063	7477	3.55
	6	8.46	10.0	21.3	7.0	10.1	16.0	3172	5700	7.04
	7	8.44	7.7	17.9	3.9	10.3	10.1	3172	2128	7.04
	8	8.46	7.1	23.0	6.1	10.1	8.6	3100	2329	7.09
	9	8.49	7.7	22.8	5.8	9.8		3327	1924	7.35

Table D9: Belwood Lake 2010 Diel Unused Data

Date	Time	Site	pH	TSS (mg/L)	Chlorophyll a (ug/L)	Phaeopigments (ug/L)	Cl (mg/L)	SO_4 (mg/L)	SRP (ug/L)	CO_2 (%)	CH_4 (%)	Temp (°C)
Aug.11, 2010	8:00	1	8.27	7.4	7.7	9.3	16.2	2.8		3766	22906	20.99
	8:20	4	8.94	12.9	31.7	7.0	12.4	2.4		4428	118845	23.77
	9:00	7	8.19	6.2	2.5	1.9	9.3	1.9		6857	983849	21.04
	9:20	8	8.17	4.9	2.5	1.8	8.9	1.9		6765	329542	21.04
	9:30	9	8.15	4.6	2.2	1.8	8.7	1.9		6870	131383	21.22
	9:45	10	8.11	4.7	1.0	1.8	12.8	2.7		6826	42156	20.65
	12:10	1	8.52	7.4	6.2	3.8	9.4	1.9		5164	18891	23.24
	12:40	4	8.99	22.7	81.1	15.3	12.6	2.5		4332	130499	25.64
	13:00	7	8.23	6.5	3.4	2.0	10.0	2.0		7199	1028037	21.28

Table D9 Continued: Belwood Lake 2010 Diel Unused Data

Date	Time	Site	pH	TSS (mg/L)	Chlorophyll a (ug/L)	Phaeopigments (ug/L)	Cl (mg/L)	SO_4 (mg/L)	SRP (ug/L)	CO_2 (%)	CH_4 (%)	Temp (°C)
Aug. 11, 2010	13:15	8	8.21	4.9	3.41	1.33	12.5	2.6		7438	519699	21.83
	13:30	9	8.27	4.0	2.79	1.82	10.5	2.1		7686	148672	22.46
	13:40	10	8.44	3.9	0.72	1.18	12.9	2.7		6149	332	23.97
	15:45	1	8.73	6.5	4.64	3.15	12.5	2.7	2.0	662	17109	26.58
	16:15	4	9.08	16.5	31.33	10.49	7.9	1.6	27.0	266	178652	26.67
	16:30	7	8.13	7.7	2.39	2.49	9.8	2	32.2	2954	1428072	21.01
	16:40	8	8.17	5.2	3.72	2.65	11.7	2.3	22.5	2597	623612	21.46
	17:00	9	8.18	2.8	2.34	2.43	6.5	1.4	28.5	2387	164344	21.94
	17:15	10	8.60	4.0	1.95	1.51	9.4	1.9	14.4	1809	37001	25.39
	20:50	1	8.77	7.4	5.21	4.54	10.8	2.3	11.1	6758	18801	25.56
	21:10	4	9.24	22.3	52.20	10.18	12.0	2.6	2.0	5380	77193	26.34
	21:45	7	8.13	4.8	3.00	2.40	10.2	2.2	23.2	7982	1416246	20.82
	21:59	8	8.11	5.5	3.70	2.26	12.6	2.4	11.8	8414	765368	20.81
	22:24	9	8.15	5.2	4.12	3.51	12	2.5	2.2	8794	265591	20.67
	22:40	10	8.34	4.9	1.83	3.19	8.5	1.9	27.1	7982	72270	22.82
	0:26	1	8.56	8.3	5.44	4.06	9.7	1.9	8.3	6664	25194	24.14
	0:50	4	9.30	20.0	54.63	13.42	13.7	3.1	10.5	4935	66862	25.96
	1:21	7	8.21	7.1	3.18	2.28	12.6	2.6	18.8	8907	1363450	20.84
	1:34	8	8.11	6.6	1.92	2.30	11.9	2.5	15.2	8020	673148	20.79

Table D9 Continued: Belwood Lake 2010 Diel Unused Data

Date	Time	Site	pH	TSS (mg/L)	Chlorophyll a (ug/L)	Phaeopigments (ug/L)	Cl (mg/L)	SO_4 (mg/L)	SRP (ug/L)	CO_2 (%)	CH_4 (%)	Temp (°C)
Aug 11, 2010	1:59	9	8.16	8.0	1.1	1.7	9.3	1.9	31.1	8197	243275	20.78
	2:23	10	8.24	4.9	2.4	3.2	7.4	1.7	23.0	5349	112285	20.98
	3:59	1	8.57	7.2	4.7	4.6	11.3	1.9	23.5	4622	99130	23.45
	4:27	4	9.32	9.1	22.8	4.9	13.7	3.1	1.7	1640	86910	25.77
	5:04	7	8.5	9.5	3.9	4.5	12.0	2.4	18.8	4783	74053	20.98
	5:20	8	8.29	6.0	1.6	2.1	12.1	2.4	21.4	6601	32785	20.89
	5:45	9	8.26	4.3	1.8	2.7	12.4	2.8		5096	9604	20.76
	6:06	10	8.29	4.3	2.5	2.7	5.9	1.4		5667	63099	20.68

Appendix E: List of Equations

Chapter 1

1.1 $6CO_2 + 6H_2O \rightarrow C_6H_{12}O_6 + 6O_2$

1.2 $NH_4^+ + 1\ ½\ O_2 \rightarrow 2H^+ + NO_2^- + H_2O$

1.3 $NO_2^- + ½\ O_2 \rightarrow NO_3^-$

1.4 $NH_4^+ + 2O_2 \rightarrow NO_3^- + H_2O + 2H^+$

1.5 $NO_3^- + 2H^+ + 2e^- \rightarrow NO_2^- + H_2O$

1.6 $NO_2^- + 2H^+ + e^- \rightarrow NO + H_2O$

1.7 $2NO + 2H^+ + 2e^- \rightarrow N_2O + H_2O$

1.8 $N_2O + 2H+ 2e^- \rightarrow N_2 + H_2O$

Chapter 2

2.1 $\frac{d[\mathrm{O_2}]}{dt} = \left(\frac{G}{Z}\right)([\mathrm{O_2}]_\mathrm{s} - [\mathrm{O_2}]_\mathrm{w}) - R + P + A$

2.2 $R = \left(\frac{G}{Z}\right)([\mathrm{O_2}]_\mathrm{s} - [\mathrm{O_2}]_\mathrm{w}) - \frac{d[\mathrm{O_2}]}{dt}$

2.3 $P - R = P_{net} = \frac{d[\mathrm{O_2}]}{dt} - \left(\frac{G}{Z}\right)([\mathrm{O_2}]_\mathrm{s} - [\mathrm{O_2}]_\mathrm{w})$

2.4 $r = \frac{(C_s - C_a)}{(C_s - C_b)}$

2.5 $E = 1 - \frac{1}{r}$