Assessing the Effects of Deep Release and Surface Release Reservoirs on Downstream Benthic Macroinvertebrate Communities in the Grand River Watershed: Implications for Planning and Management

by

Elise Elsha denHeyer

A thesis
presented to the University of Waterloo
in fulfillment of the
thesis requirement for the degree of
Master of Environmental Studies
in
Planning

Waterloo, Ontario, Canada, 2007

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.

I understand that my thesis may be made electronically available to the public.

Abstract

River regulation and reservoirs can provide a variety of services including flood protection, flow management and flow augmentation, however, there is increasing concern regarding these effects on downstream lotic environments and aquatic ecosystems. While a growing body of knowledge regarding the ecological effects of regulation exists, little is still known about the effects of reservoirs and their management strategies on benthic macroinvertebrates in the Grand River watershed and further research is needed for sufficient watershed planning and reservoirs management practices. In this study, the downstream effects of river regulation and reservoir on aquatic ecosystems were evaluated using benthic macroinvertebrate biomonitoring techniques.

Field research was conducted on five reservoirs (three deep release and two surface release) located within the Grand River watershed during three sampling periods in May-June, August and November, 2006. Benthic macroinvertebrates were collected using a T-sampler in reaches upstream and downstream of each reservoir across stream riffles perpendicular to stream flow direction. Changes in benthic macroinvertebrate community structure were quantified using nine summary indices. Downstream of reservoirs, invertebrate abundance, Hilsenhoff's Biotic Index (HBI) values and Isopoda and Chironomidae abundance increased, while taxa richness, Ephemeroptera, Plecoptera and Trichoptera (EPT) taxa and Ephemeroptera abundance decreased. Although comprehensive chemical testing was not conducted in the present study, changes in benthic macroinvertebrate abundance and diversity and a review of literature suggests that downstream ecosystems may have been impacted by changes in water quality, thermal alterations and modifications to habitat diversity induced by impoundments and most noticeably deep release reservoir designs.

Benthic macroinvertebrates are useful biological indicators and monitoring tools to assess the effects of reservoirs and their management strategies on downstream ecosystems. Information gained from this study may assist policymakers and planners in monitoring, developing and implementing improved watershed planning and reservoir management decision making.

Acknowledgements

I wish to extend my appreciation to my supervisor Dr. Mike Stone for all the advice, assistance and encouragement he so generously provided throughout the development of this thesis. Additional thanks to my committee member Dr. David Barton for his valuable input regarding sampling methods and interpretation of results. My appreciation is also extended out to committee member Dr. Stephen Murphy for his evaluation of this work.

I wish to extend my gratitude to my boyfriend George Sigeti for assisting me with field work over the sampling duration. Gratitude is also expressed toward Ecology Lab Manager Larry Lamb for graciously providing a work space to carry out benthic sorting and identification in addition to providing access to various resources. I am also grateful to Don Nguyen and Amanda Newell for assisting me with benthic analysis. Erin Harvey of the University of Waterloo Statistical Consulting Service provided valuable advice regarding data analysis. Thank you to the staff at the University of Waterloo Map and Design Library for their assistance in accessing geospatial data.

Finally, I wish to extend a special thanks to all my friends and family for their love and support from afar during this period of study.

Table of Contents

List of Figures

List of Tables

Chapter 1: Introduction

1.1 Statement of the Problem

Regulation is a common means of flood protection and flow management that alters physical, chemical and biological processes in rivers (Petts, 1984; Gore and Petts, 1989; Shantz *et al.*, 2004). Reservoirs capture water during high flows so that it can be released during periods when natural flows are inadequate to meet human water requirements (McCartney *et al.*, 2000). Dams and river regulation have become an integral part of our twentieth-century landscape and, during the past 70 years, nearly all of the major rivers of the world have been impounded to a certain degree (Petts, 1984; Collier *et al.*, 1996). River regulation can provide a variety of services, including drinking water, power generation, flood control, navigation, irrigation and recreational opportunities (Bednarek, 2001). However, the ecological effects of river regulation must also be considered.

There is a growing body of knowledge regarding the ecological effects of river regulation on downstream lotic environments (Stanford and Ward, 1979; Petts, 1984). Reservoirs create downstream alterations to the abiotic and biotic environment through changes in flow, water quality, thermal alterations, and substrate and vegetation modification (Petts, 1984). These changes can have significant ecological consequences and numerous studies have documented these effects on aquatic organisms including benthic macroinvertebrates.

Historically, reservoir design, which mainly specifies management and daily operation protocols, has emphasized maximizing the economic use of water. Less consideration has been directed to the long-term ecological consequences of physical alteration to flow volumes, flow patterns and water quality (Petts, 1984). Attention has been directed more recently to the management of regulated rivers to maintain ecological integrity. This term is defined as the ability of a stream to support a community of organisms having species composition, diversity and functional organization (Leopold, 1968; Gore and Petts, 1989).

One method of measuring ecological integrity is to develop environmental monitoring programs, which use a range of environmental indicators that evaluate species abundance and diversity (Fisher, 1998). While previous monitoring assessments have largely been focused on using physical and chemical indicators, the potential of biological indicators in biomonitoring has been recognized (Rosenberg and Resh, 1993). Benthic macroinvertebrates amongst biological taxonomic groups are a preferred means in biomonitoring studies (Hellawell, 1986). Many studies have used the benthic macroinvertebrate community to examine downstream impairment from regulated rivers (e.g. Petts, 1984; Gore and Petts, 1989; Hellawell, 1986). These studies show that benthic macroinvertebrates respond through changes in abundance and diversity and are therefore relevant indicators of environmental change in rivers.

Previous research has shown that reservoirs significantly alter downstream lotic ecosystems as a result of flow management (Petts, 1984). However, little is still known about the downstream effects of reservoirs and their management strategies on stream ecology and benthic macroinvertebrates in the Grand River watershed. This thesis examines the effects of deep release and surface release reservoirs on stream ecosystems using benthic macroinvertebrate biomonitoring techniques in the Grand River watershed. Such knowledge is required for the planning and management of natural resources and environmental health of watersheds, including the Grand River.

1.2 Objectives

The goal of the present study is to examine the downstream effects of reservoirs on stream ecosystems comparing reservoir management strategies of deep release and surface release reservoirs. Specific objectives are to:

1. Review literature pertaining to the environmental impacts of reservoirs and river regulation.

2. Evaluate the abundance and diversity of benthic macroinvertebrate communities upstream and downstream of five reservoirs in the Grand River watershed.

3. Discuss the management and monitoring implications of deep release and surface release reservoirs within the context of watershed health.

1.3 Thesis Organization

Five chapters are presented in this thesis. Chapter 1 summarizes literature pertaining to the effects of reservoirs on stream ecosystems in order to provide a context for the thesis. Chapter 2 describes the experimental design, study area characteristics and methods. The results and trends in benthic macroinvertebrate data are presented in Chapter 3. In Chapter 4, trends in benthic macroinvertebrates data are discussed in the context of the literature and implications of the study for watershed management and planning are presented. Finally, conclusions and recommendations for future research are presented in Chapter 5.

1.4 Literature Review

1.4.1 Introduction

The present review of literature focuses on the evaluation, management and monitoring of river regulation on stream ecology and benthic macroinvertebrates. Theoretical frameworks describing the ecology of natural and impacted streams are reviewed to gain insight into the effects of river regulation on biota downstream from reservoirs. The environmental impacts of river regulation are discussed and literature regarding the effects of reservoirs on stream ecosystems is examined. Finally, various monitoring approaches are discussed and literature regarding monitoring using benthic macroinvertebrates is reviewed. This review of literature provides a context in which to interpret the results of the current study.

1.4.2 Regulation of Rivers

River regulation is a common means of flood protection and flow management that alters the hydrologic cycle and related eco-hydrological processes (Petts, 1984; Gore and Petts, 1989; Shantz *et al*., 2004). The ecological effects of river regulation have become a major focus of environmental research and this is reflected by the triennial International Symposia on Regulated Streams and the foundation of the journal of *Regulated Rivers*. However, only recently has attention been directed to the management of regulated rivers to maintain ecological integrity (Gore and Petts, 1989).

Reservoirs regulate rivers by impounding water which is stored during spring melt and storm events so that it can be released during the times that natural flows are inadequate to meet human water requirements (McCartney *et al*., 2000). Reservoirs were first constructed for the purpose of river regulation over 5000 years ago in Egypt (Collier *et al*., 1996) although the era of major dam building activity did not begin until the early 1900's. Between 1945 and 1971, there was a period of increasing river regulation when a total of 8140 large dams were built world-wide (Petts, 1984). Dams and river regulation have become an integral part of our twentieth-century landscape and during the past 70 years, most of the major rivers of the world have been impounded to some degree (Collier *et al*., 1996; Petts, 1984).

Flow regulation provides benefits to society (McCartney *et al*., 2000) including services such as drinking water, power generation, flood control, navigation, irrigation and recreational opportunities (Bednarek, 2001). However, these structures cause a range of eco-hydrological and geomorphological impacts in river systems. Dams alter the natural cycle of flow which transforms the biological and physical characteristics of river channels and floodplains and alters the continuity of rivers (Petts, 1984; Bednarek, 2001). Although reservoirs have contributed immeasurably to the well-being of humans, they can also damage the environment by altering chemical, physical and biological processes that influence the health of stream ecosystems (Petts, 1984; Gore and Petts, 1989; Poff and Hart, 2002).

1.4.3 Theoretical Perspectives on Stream Ecology

In order to gain insight into the effects of river regulation on natural watercourses, theoretical perspectives on the ecology of natural and modified streams are discussed. Many theoretical frameworks have been developed by researchers to describe factors which influence biota, especially benthic macroinvertebrates, in freshwater streams. Benthic macroinvertebrates are operationally defined as organisms without backbones, which are retained by mesh sizes larger than 200 to 500 µm, and live on or in the bed of a stream (Rosenberg and Resh, 1993). These organisms include worms, leeches, clams, snails, crayfish and insects (Rosenberg and Resh, 1993; Barton, 1996). Theoretical perspectives describing the ecology of benthic macroinvertebrates in freshwater streams are reviewed in the following sections.

1.4.3.1 Stream Ecology

The ecology of streams is controlled by a complex set of physical, chemical and biological processes that are linked together in dynamic equilibrium (Leopold *et al.*, 1964). A number of theoretical frameworks have been developed to describe ecological processes that occur in natural streams (Table 1.1). These theoretical perspectives provide an understanding of the effects of physical, chemical and biological processes on benthic macroinvertebrates and provide a context for the present study.

Table 1.1 A comparison of conceptual frameworks on stream ecosystems (Lorenz *et al*., 1997).

Concept	Key Points	Reference
Zonation	▪ Rivers are divided into zones based on physical conditions such as flow velocity and temperature. ▪ Physical conditions determine the ecosystems structural zones of fish and benthic fauna.	Illies and Botosaneanu, 1963
River Continuum (RCC)	▪ Longitudinal dimension. ▪ Rivers are a continuous gradient of physical conditions. ▪ Biological communities approach equilibrium with the dynamic physical conditions of the channel so that nutrient processing strategies minimize energy loss.	Vannote *et al*., 1982
Stream Hydraulics	▪ Longitudinal dimension. ▪ Combination of Zonation and RCC. ▪ Stream hydraulics are determined by geomorphic and hydrologic characteristics. ▪ Hydraulics affect biological zonation.	Statzner and Higler, 1986
Nutrient Spiralling	▪ Longitudinal dimension. ▪ Riverine ecosystems are characterized by downstream transfer and storage of nutrients. ▪ Spiralling length is the longitudinal distance for one complete nutrient cycle to occur.	Newbold *et al*., 1981
Serial Discontinuity	▪ Longitudinal dimension. ▪ Dams are viewed as discontinuities within the river continuum. ▪ Discontinuity resets the river continuum. ▪ Biological communities can be predicted by their distance from the discontinuity and the extent to which the physical conditions of the stream depart from reference sites.	Ward and Stanford, 1983
Flood Pulse	▪ Lateral dimension. ▪ Rivers and floodplains are components of a single dynamic system. ▪ A flood pulse determines the connectivity and exchange between the river and floodplain ecosystems.	Junk *et al*., 1989
Riverine Productivity Model	▪ Lateral dimension. ▪ RCC overemphasizes diminishing riparian vegetation with increasing stream order and nutrient transport from lower order streams. ▪ Productivity in higher order streams is a combination of transport and input from riparian vegetation.	Thorp and Delong, 1994
Hyporheic Corridor	▪ Vertical dimension. ▪ The hyporheic zone is the ecotone between the surface stream and deep groundwater. ▪ Ecological processes are influenced by water movement and physical and chemical features of the two zones.	Stanford and Ward, 1993
Catchment	▪ Longitudinal, vertical, lateral and temporal dimensions. ▪ Emphasizes the relationship of the stream to the watershed. ▪ Integration of various stream concepts.	Petts, 1994

The Zonation Concept is one of the earliest conceptual frameworks developed to describe natural riverine ecosystems. This model divides the river into zones characterized by physical conditions, such as water temperature and flow velocity (Lorenz *et al*., 1997). The concept proposes that streams can be classified into three groups based on stream size: headwater (first to third order), medium sized streams (third to sixth order) and large rivers (larger than 6^{th} order) (Lorenz *et al*., 1997). The physical characteristics of each zone, such as flow velocity and temperature, affect the distribution of the stream biota and benthic macroinvertebrate community (Lorenz *et al*., 1997).

Vannote *et al*., (1980) developed the river continuum concept (RRC), which suggests that biotic stream communities adapt their structural and functional characteristics to the abiotic environment in a continuous longitudinal gradient from the headwater to the river mouth (Lorenz *et al.,* 1997). The physical gradient affects the composition of stream biota, which elicits a continuum of benthic macroinvertebrate functional feeding groups along the length of the river (Vannote *et al.,* 1980). The headwater area of the longitudinal continuum is influenced by shading from riparian vegetation which lowers the ratio of gross primary productivity (P) to community respiration (R) so that P/R<1. Terrestrial vegetation contributes a large amount of coarse or allochthonous particulate organic matter (CPOM, >1mm) to headwater streams in forested areas. The headwater fauna are dominated by shredders which break CPOM down into fine particulate organic matter (FPOM, 0.5 µm-1mm) that is used by collector species (Figure 1.1).

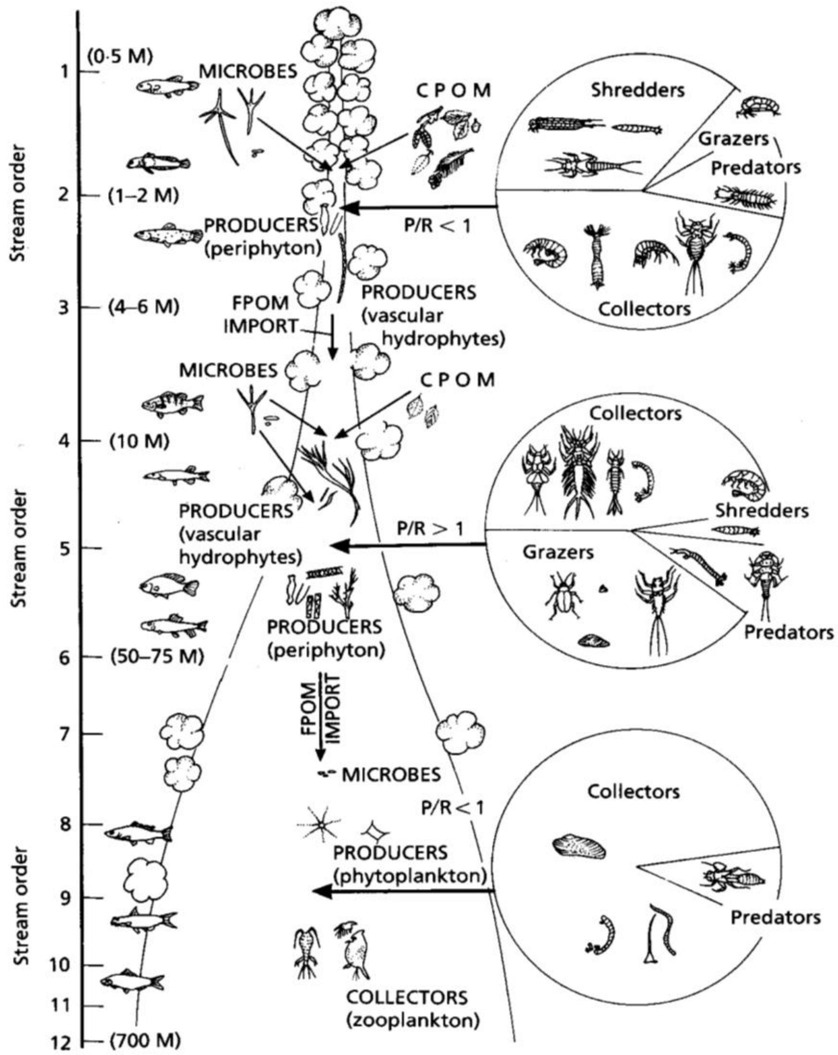

Figure 1.1. **The relationship between stream size and progressive shift in structural and functional attributes of aquatic communities (Vannote *et al*., 1980).**

Riparian vegetation and shading typically decrease as stream order increases, resulting in increased primary production in medium sized streams (P/R>1). These areas are dominated by grazers that feed on the primary trophic level and collectors that utilize FPOM transported from upstream. In large rivers, primary productivity is reduced by depth and turbidity (P/R <1). Collectors, which utilize FPOM transported from headwater and mid-sized streams are dominant in large rivers.

The RCC (Vannote *et al*., 1980) focuses on the longitudinal stream dimension and its effect on biological communities. In contrast, the Hyporheic Corridor Concept (Stanford & Ward, 1993) places emphasis on the vertical dimension of stream systems in defining aquatic ecology. The hyporheic zone is the interface between surface water and groundwater. Ecological processes in the hyporheic ecotone are influenced by lateral and horizontal water movement, permeability, substrate size and physiochemical features of the overlying stream and adjacent aquifers (Boulton *et al*., 1998). Substrate size plays a particularly important role in determining the abundance and diversity of benthic macroinvertebrates (Minshall, 1984) and is an important factor to consider when designing a benthic biomonitoring study.

1.4.3.2 Anthropogenic Disturbance and Aquatic Ecology

Disturbances in riverine ecosystems can be both natural and anthropogenic (Lorenz *et al*., 1997). Natural impacts result in events such as floods and droughts, while anthropogenic disturbances include channel and hydrological modification, land use change and transfer of pollutants to stream ecosystems (Milner, 1994). Most aquatic habitats and stream ecosystems are capable of adapting to natural impacts, whereas anthropogenic impacts may cause more significant damage (Vannote *et al.*, 1980).

Anthropogenic impacts can influence abiotic variables which can alter biotic functional and structural characteristics of stream ecosystems (Lorenz *et al.*, 1997)(Figure 1.2). Anthropogenic impacts include those such as nutrient enrichment, organic pollution and alteration of riparian vegetation (Lorenz *et al.*, 1997) which affect water quality, as well as stream habitats and ecosystems (Vannote *et al.*, 1980). Vannote *et al*., (1980) suggest the RCC is a reliable framework for evaluating change in stream ecosystems from anthropogenic perturbations.

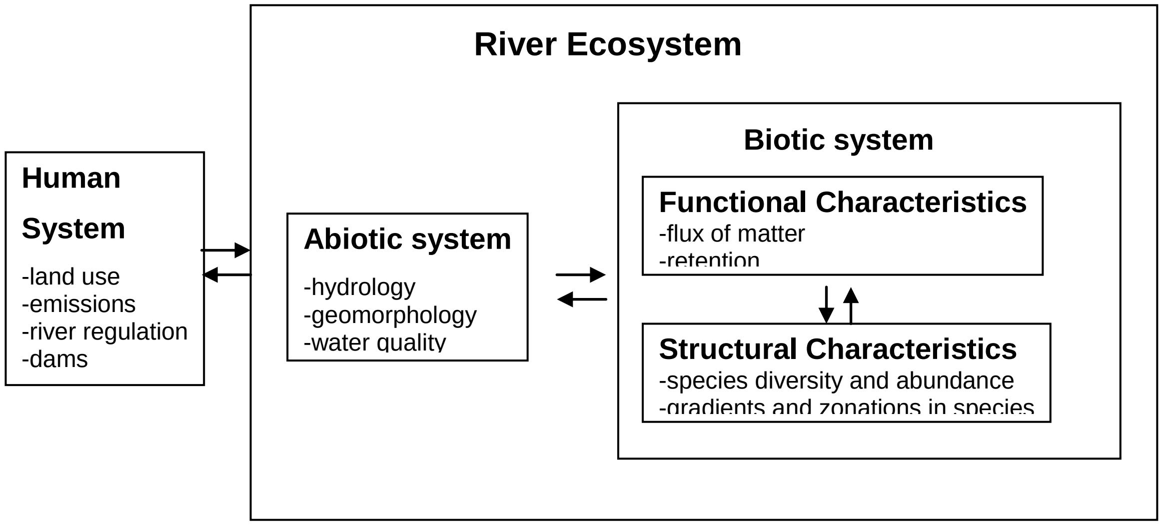

Figure 1.2. Interactions within and between riverine ecosystems and human systems (Lorenz *et al*., 1997).

The Serial Discontinuity Concept developed by Ward and Stanford (1983) addresses the effects of river regulation and reservoirs on stream ecosystems. Dams are discontinuities within the river continuum (Stanford and Ward, 2001) and cause upstream and downstream shifts in abiotic and biotic parameters and processes (Lorenz *et al*., 1997). For example, reservoirs alter water temperature, reduce connectivity between the stream and riparian zone and alter downstream CPOM fluxes, resulting in a shift towards collector biota, which primarily consume FPOM (Lorenz *et al*., 1997). The degree to which biological communities are impacted by the discontinuity is a function of distance from the discontinuity and the extent of departure from reference conditions (Stanford and Ward, 2001).

1.4.4 Effects of Reservoirs on Stream Ecosystems

There is increasing concern that reservoirs can alter downstream ecosystems (Stanford and Ward, 1979; Petts, 1984) by modifying the downstream flux of water, sediment and water temperature and creating barriers to upstream-downstream movement of organisms and nutrients (Poff and Hart, 2002). Some studies (Poff and Hart, 2002; Petts, 1984; Ward and Stanford, 1987) claim these fundamental alterations to the abiotic environment have significant ecological consequences (Table 1.2).

Table 1.2. Effects of regulation on abiotic and biotic components in downstream lotic reaches from dams (Poff and Hart, 2002; Ward and Stanford; Petts, 1984).

Abiotic and Biotic Variables	Observed Modification
Flow Management	▪ Reduce seasonal flow variability. ▪ Alter timing of annual extremes. ▪ Reduce flood magnitudes.
Suspended Particles	▪ Alter supply and transport of organic and inorganic particles.
Channel Morphology	▪ Alter cross-sectional area, downcutting, and lateral movements. ▪ Reduce fine particles and leave predominately course particles. ▪ Reduce base level of tributaries entering stream.
Chemical Conditions	▪ Alter seasonal patterns and reduce natural temporal variability in the chemistry of water. ▪ Alter dissolve gasses, especially dissolved Oxygen which range from anoxic to supersaturated.
Thermal Conditions	▪ Surface release reservoirs elevate summer temperatures and delay vernal warming and autumnal cooling. ▪ Deep-release storage reservoirs decrease annual and diel ranges, produce winter warm and summer cool conditions, and disrupt periodicity patterns.
Vegetation	▪ Encourage growth of attached algae and higher plants.

1.4.5 The Effects of Reservoirs on Benthic Macroinvertebrates

During the past four decades, many studies have documented the effects of reservoirs on benthic macroinvertebrate communities in stream ecosystems. In a review of thirteen studies from United States, Europe and South Africa, Stanford and Ward (1979) found a reduction in species-diversity of benthic macroinvertebrates downstream of the river impoundment, while the majority of reservoirs showed an overall increase in benthic macroinvertebrate abundance (Table 1.3).

More recently, numerous studies have documented the downstream effects of reservoirs on aquatic ecosystems using benthic macroinvertebrates (Table 1.4). Many studies have used the diversity and richness of the benthic macroinvertebrate community to examine the physical, chemical and biological alterations of regulated rivers (Petts, 1984; Gore and Petts, 1989; Hellawell, 1986). In general, benthic invertebrates display several responses, primarily through changes in abundance and diversity, to changes in downstream lotic environments resulting from flow management, water quality impacts, thermal alterations, and substrate and vegetation modification.

Table 1.3. Downstream effects of reservoirs on benthic macroinvertebrates (modified from Stanford and Ward, 1979).

Reference	Location	Observed Benthic Macroinvertebrate Changes
Tarzwell, 1939	Clinch River, Norris Dams, Tennessee Valley, USA	Reduced Trichoptera and Ephemeroptera by 30% and were replaced by chironomids and gastropods.
Briggs, 1948	Stevens Creek, Central California, USA	Biomass more than doubled.
Pfitzer, 1954	Tennessee Valley, South Holston Reservoir, USA	Increased populations of simuliids, Chironomidae, *Gammarus and Hydropsyche.*
Pearson *et al*., 1968	Green River Flaming Gorge Dam, USA	Number of taxonomic groups reduced.
Penaz et al., 1968	River Svratka, Vir Valley, Reservoir	Number increased up to 3.5 times and biomass up to 2.8 times.
Hilsenhoff, 1971	Mill Creek, Wisconsin, USA	Many species eliminated and the fauna became dominated by *Simulium* sp., Chironomidae and *Gammarus* sp.
Lehmkuhl, 1972	S. Saskatchewan River, Gardiner Dam, Canada	Marked reduction of macroinvertebrates downstream for over 100km. 15 species of Ephemeroptera were eliminated.
McClure and Stewart, 1976	Brazos River, Possum Kingdom Reservoir, USA	Increased zoobenthos diversity for 80km below the dam.
Mullan *et al*., 1976	Upper Colorado River, Navajo Dam, USA	Invertebrate densities increased from $820m^{-2}$ to $6727m^{-2}$ within a 13km reach.
Ward, 1976	S. Platte River, Cheesman Lake, USA	Reduced diversity but increased standing crop for 32km.
Young *et al*., 1976	Guadalupe River Canyon Reservoir, USA	Diverse macroinvertebrate community established 24 km downstream 5 years after dam closure.
Armitage, 1978	River Tees, Cow Green Reservoir, UK	Reduced diversity and increased biomass for only 400m below the dam.
Scullion *et al*., 1982	River Elan, Craig Goch Reservoir UK	Reduced abundance and diversity.

Table 1.4. Downstream effects of reservoirs on benthic macroinvertebrates.

Reference	Location	Observed Benthic Macroinvertebrate Changes
Munn and Brusven, 1991	Clearwater River, Idaho, USA	Found high abundance and low taxa richness. Dominated by Orthoclad chrinomids from 68-99%.
Al-Lami *et al*., 1998	Radica Lake, Iraq	Total mean density and abundance increased. Benthic community dominated by Oligochaeta.
Ogbeibu and Oribhabor, 2001	Ikpoba River, Nigeria, Africa	Abundance and density of benthic macroinvertebrates were significantly decreased.
Cereghino *et al*., 2002	River Oriege, France	Low abundance.
Cortes *et al*., 2002	Alto Lindoso and Touvedo Dam, Portugal	Decreased variation and diversity of benthic macroinvertebrates.
Mwaura *et al*., 2002	Eight small reservoirs Kenya, Africa	Low diversity and abundance of benthic invertebrates. Dominated by Lumbriculidae and Chironomidae.
Lessard and Hayes, 2003	Ten small dams Michigan, USA	Community composition of benthic macroinvertebrates shifted
Richardson *et al*., 2003	Peticodiac River New Brunswick	Increased abundance of resistant species: Chironomidae. Macroinvertebrates reduced downstream post drawdown.
Brandimarte *et al*., 2005	Mogi-Guacu River, Brazil	Reduction of taxa composition.
Michaletz *et al*., 2005	Thirty impoundments Missouri, USA	Ephemeroptera and Odonata abundance decreased. Diptera abundance increased.
Furey *et al*., 2006	Sooke Lake Reservoir, British Columbia	Biomass of benthic macroinvertebrates decreased post reservoir drawdown.
Moreno and Callisto, 2006	Iberite Reservoir, Brazil	Low values of richness and diversity. High densities of tolerant organisms.
Vallania, 2007	Grande River, Argentina	Filter-feeders, scrapers and predators increased and detrivores and shredders decreased.

1.4.5.1 Flow Management

The hydrological alterations and the reduction of natural high flows resulting from reservoirs used in flow management can alter the character abundance and diversity of downstream ecosystems and their aquatic organisms. For example, the life-cycles of many lotic organisms, including benthic macroinvertebrates, rely on seasonal variations in discharge, including high flows, which are important for respiratory, physiological and feeding requirements (Petts, 1984). Artificially low flows may favor flow-specific species and reduce the number of organisms that are adapted to fast-moving water (Petts, 1984), while artificially high flows may reduce those adapted to slow-flowing water (Gore and Petts, 1989). Fluctuating flows within impounded rivers often reduce benthic macroinvertebrate abundance and diversity downstream for few species are able to adapt (Petts, 1984).

Flow alterations can alter the drift behaviour of benthic macroinvertebrates. Drifting is defined as the downstream transport of benthic fauna by current in lotic waters (Rosenberg and Resh, 1993). Variations in drift are related to many factors including density of organisms in the benthos, the life-history stage, the biological activity and behaviour, as well as flow-velocity (Rosenberg and Resh, 1993). Most benthos drift throughout the night but artificially high or low flows can cause a massive number of organisms to drift during the day (Petts, 1984). As a result, this may cause a great reduction in the number of benthos because many can be consumed by benthivorous sight feeding fish. Artificial water-level fluctuations can affect the drift behaviour and alter the abundance and diversity of benthic macroinvertebrates (Petts, 1984).

1.4.5.2 Water Quality Impacts

Alterations in the abundance and diversity of benthic macroinvertebrates downstream of reservoirs have also resulted from significant changes in water quality. The release of water from an anoxic hypolimnion can have adverse consequences on the downstream benthic fauna as low dissolved oxygen waters are often transmitted to receiving streams (Petts, 1984). Spence and Hynes (1971) suggested that the release of poorly oxygenated water may have caused the elimination of three predatory species of Plecoptera from the Grand River below Shand Dam, due to their sensitivity to changes in dissolved oxygen levels.

Eutrophication of impoundments caused by excessive nutrients (phosphorus and nitrogen) increase the potential for the development of algae and presents a potential threat to the composition of the benthic community (Symons, 1969). Sephton *et al.*, (1983) determined that the observed changes in the chironomid community downstream of Laurel Creek Reservoir were associated with increased eutrophic and algal conditions. Spence and Hynes (1971) also reported a reduction in species diversity below Shand dam, due to mildly eutrophic conditions.

1.4.5.3 Thermal Alterations

The alteration of the thermal regime has been recognized as a critical factor influencing changes in the biotic community (Petts, 1984). For example, many life-cycle phenomena of benthic macroinvertebrates such as hatching, growth and emergence, depend on thermal cues (Rosenberg and Resh, 1993). The rapid vernal rise in temperature required by some species for maturation and emergence may not occur in the summer-cool waters below deep release dams (Ward and Stanford, 1987). Accordingly, Lehmkuhl (1972) observed a marked reduction of invertebrates, including the elimination of 19 species of Ephemeroptera downstream of the deep release reservoir, Gardiner dam. Reservoir management should consider the importance of the thermal regime upon all biotic components in downstream habitats.

1.4.5.4 Substrate Modification

The heterogeneity of substrate particle size is critical for varied microhabitats, which can support and maintain abundant and diverse fauna (Hynes, 1970; Hellawell, 1986). Moreover, the spaces between particles in the substrate are of vital importance for many organisms providing additional living space and an important refuge against high-flow velocities (Hynes, 1970). The downstream deposition of sediment below reservoirs can effectively fill the interstitial spaces of the substrate. This subsequently can seal off microhabitat while significantly reducing habitat heterogeneity by modifying the substrate particle size in downstream reaches (Petts, 1984).

1.4.5.5 Vegetation Modification

Reservoirs can modify the abundance and diversity of vegetation in regulated streams (Petts, 1984; Ward and Stanford, 1987). The reduction in the magnitude and frequency of floods can produce dense masses of algae. Beds of submerged angiosperms may also develop in regulated streams in regions where such plants are normally absent from lotic biotopes (Ward, 1976). In addition to changing the food base, the enhanced aquatic flora may eliminate clean rock surfaces, as well as provide sites that serve as refugia from the current. As a result, some species of benthic invertebrates are eliminated, while others, which were previously absent, can now invade the regulated segment of stream (Ward, 1976). In eutrophic waters, aquatic plants may reach nuisance proportions below dams, however, the additional habitat niches and the increased amount of food provided for aquatic insects may be viewed as a beneficial change in an unproductive system (Ward, 1984). Management schemes may control aquatic flora by regulating the intensity and frequency of flood events or by releasing water from reservoir strata with certain nutrient levels (Ward, 1984).

Other studies show that a downstream increase of plankton from an upstream reservoir outflow may account for the dense population of filter-feeding insects that can develop in stream reaches below dams (Ward, 1984). Spence and Hynes (1971) demonstrated that the outflow of organic matter, particularly zooplankton and phytoplankton, from the Shand Dam greatly influenced the community structure of downstream benthos and accounted for the increased abundance of detrivorous arthropods. Control of plankton concentrations released from the dam, based on monitoring the depth distribution of reservoir plankton, may also be an effective management strategy (Ward, 1984).

1.4.5.6 Reservoir Management Strategies

Reservoir management strategies, such as reservoir drawdown, can also generate a wide range of discharge patterns and environmental impacts (Furey *et al*., 2006; Richardson *et al*., 2003; Petts, 1984). Reservoir drawdown (declining water level) normally begins during late summer or early fall to capture the spring runoff and manage downstream river flows (Maul *et al*., 2004). During summer reservoirs can provide considerable storage volume for flood reduction and provide flow augmentation (Petts, 1984). The seasonal drawdown operation aims to capture high sediment and nutrient loads during the spring melt period and regulate reservoir levels so they are high during summer months and low during winter months (Pizzuto, 2002). An increasing number of studies have documented that reservoir drawdown can significantly alter the integrity of benthic macroinvertebrate communities.

In natural rivers that experience flows of high variability, a high level of production can be attained provided that the benthic community present is adapted to the frequency and magnitude of flow fluctuation (Petts, 1984). However, such adaptations require variable adjustment periods and the combination of severe water-level fluctuations can devastate invertebrate populations (Petts, 1984). Kroger (1973) concluded that the effects of a single large drawdown on productivity are harmful and that the net effects of multiple drawdowns may devastate macroinvertebrate populations. Furey *et al*., (2004) found that the drawdown of Sooke Reservoir, which experienced more than six meters of seasonal drawdown, drastically reduced the abundance and community composition of benthic macroinvertebrates.

1.4.6 Environmental Monitoring

Environmental monitoring is an important tool used in watershed planning and management to evaluate anthropogenic changes in order to protect the health of stream ecosystems (Parr, 1994). Previous assessments have largely been focused on physical and chemical assessments, while current trends are recognizing the importance of biological monitoring and the use of benthic macroinvertebrates as monitoring tools (Rosenberg and Resh, 1993). The following section reviews literature on environmental indicators, biological monitoring and benthic macroinvertebrates.

The health of a stream ecosystem is a combination of physical, chemical and biological processes (Leopold *et al*., 1964). While the ideal approach is to evaluate all chemical, physical and biological indicators of stream quality, this method of assessment is rarely practical due to time and budget constraints (Parr, 1994). Therefore, most studies concentrate on one approach to water quality monitoring. Currently, assessment of aquatic ecosystems is based largely upon chemical measurements of water quality (Metcalfe-Smith, 1994). However, chemical indicators alone cannot provide accurate information intended for the sound management of aquatic ecosystems because they do not directly measure the effects of disturbance on living organisms (Jones *et al*., 2004).

Biological indicators can improve the interpretation of water quality monitoring that was once based solely on chemical data (Lenat, 1988). Biological communities integrate the effects of various stressors and provide a measure of their cumulative impact (Barbour *et al*., 1999).

Whereas chemical data are typically discrete and require a large number of replicates for accurate assessment of disturbance, biological indicators integrate environmental impacts over time (De Pauw and Vanhooren, 1983). Karr (1991) states that it is important to recognize that not all impacts are chemical in nature and that some disturbances may be biological and or physical. Biological monitoring is especially powerful under conditions of toxic, intermittent or mild organic pollution and habitat alteration where changes in water quality are not easily detected by chemical analyses (Barton, 1996). Despite the limitation that biological data are highly variable because of differential sensitivity to pollution and patchy spatial and temporal distribution of organisms and environment (Chapman and Kimstach, 1996), biological indicators are often preferred in the monitoring of aquatic ecosystems as the direct impact of all changes to the ecosystem can be assessed (Parr, 1994; Metcalfe-Smith, 1994).

The biological assessment of water quality is largely based on ecological surveys, bioassays or chemical analyses of body tissues (Friedrich *et al*., 1996). Ecological surveys are typically used to assess the impact of perturbations on receiving water, whereas bioassays are used to determine these effects on specific organisms (Brabec *et al.*, 2002). Indicator organisms such as microbes, plants, invertebrates and vertebrates are often used in these surveys (Markert *et al*., 2003). Amongst these biological indicators, bacteria, algae, benthic macroinvertebrates and fish are most commonly used (Hellawell, 1986).

The most reliable biological indicator of water quality is a combination of various groups of organisms (Haslam, 1982), but it is unrealistic to conduct bioassessment on the entire aquatic ecosystem so most researchers focus on a particular component (Metcalfe-Smith, 1994). According to Hellawell (1986), a clear preference has emerged for benthic biomonitoring techniques. Benthic macroinvertebrates are very abundant is most aquatic habitats (Reynoldson, 1984) and have a wide range of tolerances in aquatic systems to various degrees of perturbations. Consequently, their relatively long life cycles can be used to integrate the effects of disturbances over time (Pratt and Coler, 1976). They are good indicators of local conditions because of their sedentary lifestyle and direct relationship with stream substrate (Hellawell, 1986). Moreover, the use of benthic macroinvertebrates as monitors of water quality is convenient and economical (Olive *et al*., 1988) for qualitative sampling and analysis can be conducted using simple inexpensive equipment, and the taxonomy of many groups is well known and identification keys are available (Loeb and Spacie, 1994). Lastly, benthic macroinvertebrates are well suited to experimental approaches to biomonitoring and many methods of data analysis have been developed (Rosenberg and Resh, 1993).

There are some disadvantages to using benthic macroinvertebrates as an environmental indicator. Hellawell (1986) states that certain benthic macroinvertebrate groups can be taxonomically difficult to identify and Rosenberg and Resh (1993) argue that benthic invertebrates do not respond to all impacts. In addition, the distribution and abundance of benthic macroinvertebrates can be affected by factors other than water quality (Loeb and Spacie, 1994). Moreover, drift behaviour can carry macroinvertebrates into areas which they do not normally occur (Rosenberg and Resh, 1993). Quantitative sampling can be time consuming and seasonal variation in abundance and distribution may further create additional misrepresentations (Rosenberg and Resh, 1993; Loeb and Spacie, 1994). However, Reynoldson (1984) states that sampling errors can be reduced through replication and Barbour *et al*. (1999) suggest that sampling a single habitat, particularly riffles or runs, will aid in standardizing assessment and provide representative sampling that can be compared to other stream reaches or to other streams.

Various methods have been used to collect benthic macroinvertebrates in stream riffle environments. Included among these are active sampling methods (grab samplers and nets) and passive methods (colonization samplers) (APHA, 2005). Nets, such as the hand net, kick net and Surber samplers, are normally used up to a water depth of one meter, predominately intended for riffle areas (Barbour *et al*., 1999). Grab samplers such as the Ekman grab or core sampler can be used in areas where the substrate is soft, such as pool habitats, not in cobble-bottomed riffles (Hellawell, 1986). The T-sampler is suitable for collecting grab samples in stream environments with shallow and fast moving water (Jones *et al*., 2004).

Once benthic invertebrate samples are collected and preserved, taxonomic identification is generally carried out to the family, genus or species level. The genus and species identifications provide a more refined evaluation of environmental impairment, but are more difficult to conduct, whereas family level identification provides more precise identification, requires less expertise and accelerates assessment (Barbour *et al*.,1999). In either case, taxonomic identification level should be consistent among all samples (Barbour *et al*., 1999) but the information revealed by the benthic fauna increases with sampling effort and closer identification of the animals (Barton, 1996).

After sampling and identification of benthic macroinvertebrates, data are summarized using a number of indices. Traditionally, diversity, biotic, community comparison, and feeding measure indices have been used to summarize invertebrate data (Table 1.6).

Table 1.5. Benthic macroinvertebrate indices (modified from Barbour *et al*., 1999).

Diversity	Biotic	Community Comparison	Feeding Measures
▪ No. of individual organisms ▪ No. of taxa	▪ EPT ▪ HBI	▪ % Ephemeroptera ▪ % Chironomidae ▪ % Isopoda ▪ % Dominant taxa	▪ % Filter-feeders ▪ % Scrapers ▪ % Shredders ▪ % Predators ▪ % Detrivores

Diversity indices combine the abundance (number of individual organisms), richness (number of taxa) and evenness (distribution of individuals among taxa) into a single mathematical expression. Diversity indices are used because they are predominately quantitative and contain no subjective assumptions regarding habitat and water quality tolerances (Metcalfe-Smith, 1994). However, diversity indices have often been criticized because they give equal weight to both pollution tolerant species and pollution sensitive ones (Metcalfe-Smith, 1994).

Biotic indices, such as the Hilsenhoff's Biotic Index (HBI) used to detect organic enrichment (Hilsenhoff, 1987), combine quantitative measures of diversity with qualitative water quality tolerance scores for key taxa. Community comparison indices are used to evaluate the composition of benthic macroinvertebrates (Metcalfe-Smith, 1994). The community comparison indices are strictly quantitative unlike the biotic indices and are sensitive to anthropogenic perturbations (Metcalfe-Smith, 1994). Feeding measure indices provides information on the balance of feeding strategies (food acquisition and morphology) in the benthic macroinvertebrate community (Barbour *et al*., 1999). Feeding measure indices are qualitative and have been criticized due to difficulties with the proper assignment of benthic fauna to single functional feeding groups (Karr and Chu, 1997).

1.4.7 Environmental Monitoring of Reservoirs Using Benthic Macroinvertebrates in the Grand River Watershed

Although there have been many studies over the past four decades of environmental monitoring of reservoirs using benthic macroinvertebrates, few studies have been reported in the Grand River watershed (Sephton *et al*., 1983; Spence and Hynes, 1971; Paterson and Fernando, 1970) (Table 1.7). Studies on reservoirs in the Grand River watershed by Sephton *et al*. (1983), Spence and Hynes (1971), and Paterson and Fernando (1970) reported numerous downstream changes in benthic macroinvertebrate abundance and diversity resulting from the upstream impoundment. However, current research in the Grand River watershed is needed for land patterns have changed over the past three decades. For example, from 1971 to 1996, Canada's cities and towns have expanded steadily resulting in a 76% increase in urban land cover over the 25 year period (Hofmann, 2001). In addition, few or no studies have compared the management strategies of both deep release and surface release reservoirs on downstream aquatic ecosystems and benthic macroinvertebrate composition. Clearly, further research for the planning and management of the Grand River watershed is needed.

Table 1.6. Benthic macroinvertebrate studies on reservoirs in the Grand River watershed.

Reference	Location	Reservoir Type	Observed Benthic Macroinvertebrate Changes
Sephton *et al*., 1983	Laurel Creek Reservoir, Waterloo, ON	Surface release	▪ Tanypodinae became dominant (60%) and Chironomini decreased (38%). ▪ Decreased standing stock of Chironomidae over a 13 year study was associated with trophic conditions at the substrate and bottom oxygen levels of the reservoir.
Spence and Hynes, 1971	Shand Dam, Fergus, ON	Deep release	▪ Profound differences were found in the macroinvertebrate riffle fauna upstream and downstream. ▪ Plecoptera were absent and number of species of *Stenonema* was reduced. ▪ Number of Chironomidae, Simuliidae, *Optioservus* (Coleoptera), Hydropsychidae, and *Hyalella azteca* (Amphipoda) increased downstream. ▪ Changes were associated with increased availability of detritus and lower water temperatures downstream.
Paterson and Fernando, 1970	Laurel Creek Reservoir, Waterloo, ON	Surface release	▪ Euryoxybiontic chironomids declined and polyoxybiontic species increased in abundance. ▪ Changes in the dominance of chironomid fauna were associated with a partial loss of the rich deposits of organic debris by siltation and decomposition. ▪ Dominant species were adapted to a wider range of environmental conditions and modifications of the reservoir habitat.

1.4.8 Summary

Reservoirs alter downstream environments, which can subsequently influence the integrity of aquatic stream ecosystems and the abundance and diversity benthic macroinvertebrates. Biological indicators and benthic macroinvertebrates are effective indicators used to monitor environmental change according to many researchers because they show cumulative impacts and display long-term changes in water habitat and quality. The broad ecological perspective essential for sound management of aquatic habitats is needed to effectively manage and mitigate the impacts of impoundments on lotic ecosystems. Furthermore, there is a need for current research to assess the impacts of reservoir management strategies of deep release and surface release reservoirs on benthic macroinvertebrates in the Grand River watershed. The impacts of river regulation and reservoir management research can provide valuable information to Grand River watershed planning and stream ecology, as well as to advocate evidence of a direct relationship between reservoir impacts and stream ecosystems.

Chapter 2: Methods

2.1 Experimental Design

Five reservoirs located in the Region of Waterloo and Wellington County, Ontario were selected for this research. Based on methods by Green (1979), a reference sampling station (A) was situated upstream of the reservoir and a second sampling station (B) was established downstream of each reservoir for impact evaluation. All sampling stations were situated in stream riffle transects, which are defined as areas in streams of fast flow over boulders and cobbles, which break the water surface (Jones *et al*., 2004) (Figure 2.1). Upstream and downstream distances between stations and reservoirs varied based on differences between reservoir size and site accessibility. All stations were located in areas with road or walking accessibility to the stream, usually by access to a bridge.

Benthic macroinvertebrates were collected using a T-sampler, which is adapted to make grab samples in shallow (>1m) and fast moving water (Jones *et al*., 2004). At each sampling station, a minimum of six samples was collected at equal intervals across the transects to provide a measure of spatial variability in the abundance and diversity of organisms at each site (Figure 2.2). During the study period a total of 180 samples was collected. Invertebrates were sampled on three separate occasions (spring, summer and fall after drawdown) across each stream riffle transect in a line perpendicular to the direction of flow. Samples were preserved in the field with 10% formalin and identified in the laboratory with a dissecting microscope.

Indices were calculated to summarize benthic macroinvertebrate community abundance and diversity at each site: abundance (number per sample), taxa richness, Ephemeroptera, Plecoptera and Trichoptera (EPT) taxa richness (number of individuals), Hilsenhoff's Biotic Index (HBI), relative dominance, percent Ephemeroptera, Isopoda, Chironomidae and percent filter-feeders.

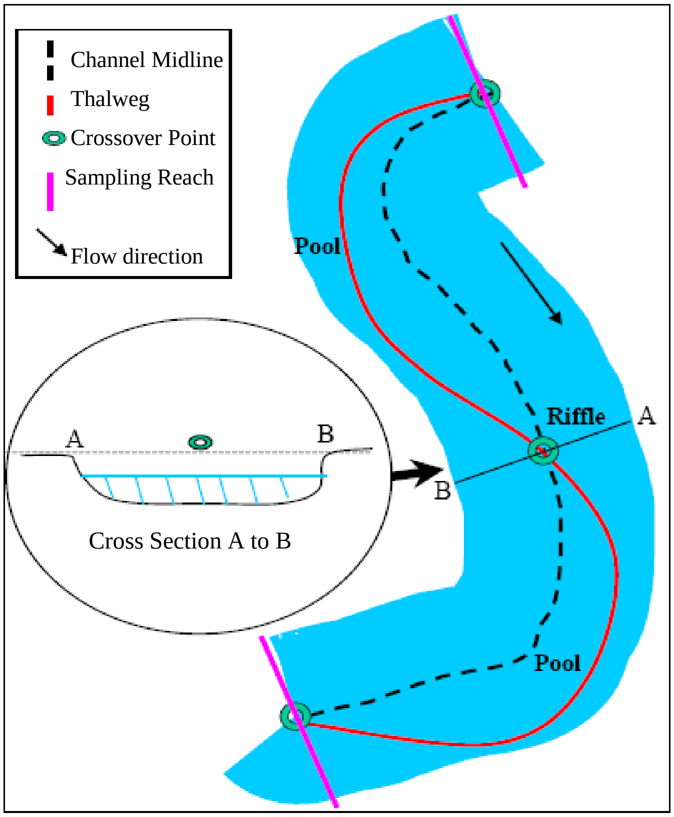

Figure 2.1. Stream riffle transect (Jones *et al*., 2004).

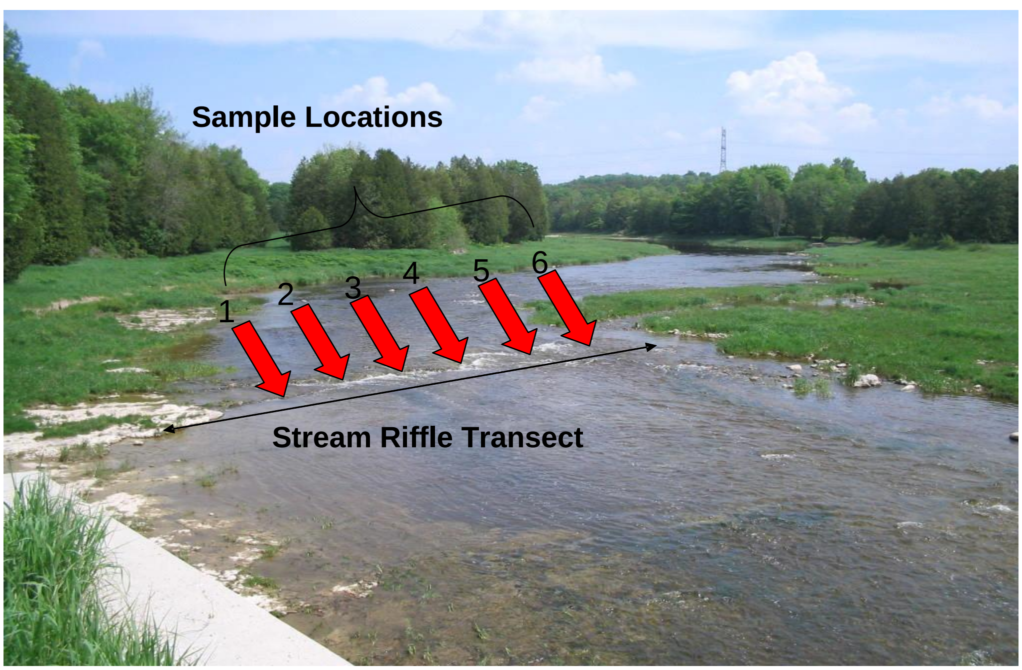

Figure 2.2. **Sampling collection at individual stations.**

2.2 Study Area Selection and Description

The Grand River drains an area of 6800 km^2 in southern Ontario and is the largest Canadian tributary flowing into Lake Erie. Flow of this river is regulated by seven multipurpose reservoirs and 25 smaller dams operated by the Grand River Conservation Authority, plus an additional 100 privately owned dams (Shantz *et al*., 2004) (Figure 2.3). Water levels in many of the deep release reservoirs are slowly lowered near the end of summer primarily for flow augmentation, while most surface release reservoirs are lowered in a shorter period of time in the fall (Boyd *et al*., 2000; Shantz *et al*., 2004)(Appendix C). Specific reservoirs were selected for this study because they provide flow management (flood control and or flow augmentation) in the Grand River watershed. After consultation with engineers at the GRCA, five reservoirs were selected based on similarity in characteristics such as function, operation (deep release or surface release), drawdown and storage capacity size (Table 2.1). All reference sites selected upstream of the reservoirs (Station A) were located in areas with little residential development.

Table 2.1. Study site selection.

Reservoir Function
• Serve as multipurpose reservoir (provides flood control and or flow augmentation) in the Grand River watershed.
Reservoir Type
• Reservoir operates as either a surface-release or deep-release dam to compare impacts.
Reservoir Drawdown
• Reservoir is seasonally drawn down either slowly in the summer or quickly in the fall to compare impacts
Reservoir Storage Capacity
• Reservoir storage capacity size is very large or very small to compare impacts.

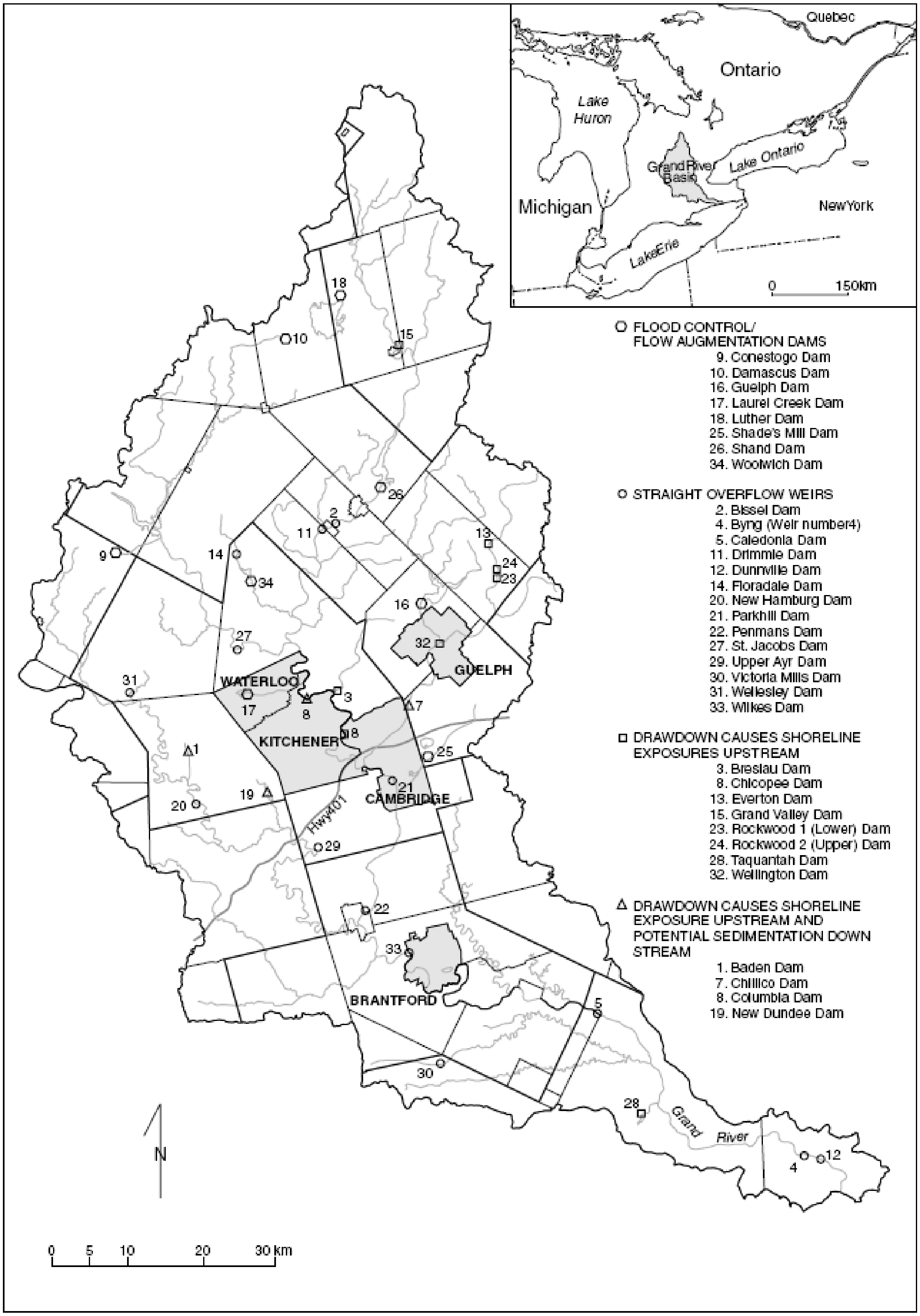

Figure 2.3. Grand River watershed dam locations (Shantz *et al*., 2004).

The five reservoirs evaluated in this study are located near the Villages of Wallenstein and Drayton, Town of Fergus and Cities of Guelph, Cambridge and Waterloo (Figure 5). Reservoir characteristics including age, height, drainage area and storage capacity were obtained from the GRCA and are presented in Table 2.2 (Boyd *et al*., 2000). Detailed descriptions of each site are presented in Table 2.3 and Appendix A contains photographs of each station. Soil type was determined using geospatial data from the Ontario Ministry of the Agriculture, Food and Rural Affairs (OMAFRA, 2000) and channel width, characteristics of the riparian buffer zone and stream substrate type were determined in the field during sampling.

Table 2.2. Reservoir characteristics (Boyd *et al*., 2000).

Reservoir	Age (years)	Multi-Purpose	Height (m)	Upstream Drainage (Km^2)	Storage Capacity (m^3)	Reservoir Type	Drawdown (m)
Shand	65	Flood control, flow augmentation	22.5	802	63,874,000	Deep release	7.13
Conestogo	49	Flood control, flow augmentation	23.1	563	59,457,000	Deep release	8.37
Guelph	31	Flood control, flow augmentation, recreation	14.3	242	22,387,000	Deep release	1.89
Shade's Mills	34	Flood control, induced infiltration, recreation	9.8	97.7	3,240,000	Surface release	1.57
Laurel Creek	39	Flood control, recreation	5.6	31.3	1,540,000	Surface release	3.03

Table 2.3. Characteristics of study areas.

Reservoir	Reservoir Number	Sampling Station	Location	Distance to Dam (km)	Channel Width (m)	Substrate Type	Riparian Width (m)	Riparian Type	Soil Type
Shand	1	A	Town of Fergus	12.51	3.9	gravel to cobble	25m+	Trees	Sandy Loam to Loam
		B	Town of Fergus	1.23	4.6	cobble to boulders	25m+	Trees	Loam
Conestogo	2	A	Village of Drayton	9.50	2.9	sand to gravel	25m+	Mixed	Loam
		B	Village of Wallenstein	9.75	3.8	gravel to boulders	25m+	Mixed	Loam
Guelph	3	A	City of Guelph	2.80	2.1	sand to cobble	25m+	Trees	Fine Sandy Loam to Loam
		B	City of Guelph	1.03	2.5	Sand to boulders	25m+	Trees	Sandy Loam to Loam
Shade’s Mills	4	A	City of Cambridge	10.80	2.0	sand to gravel	25m+	Mixed	Loam
		B	City of Cambridge	0.83	2.3	gravel to boulders	10m	Mixed	Loam
Laurel Creek	5	A	City of Waterloo	0.72	0.9	gravel	25m+	Trees	Loam
		B	City of Waterloo	0.05	1 .0	gravel	25m+	Grasses	Loam

2.2.1 Belwood Lake

Belwood Lake (Figure 2.4), located near the Town of Fergus in the Wellington County, is a 14 km long lake, which was created in 1942 with the construction of the Shand Dam (reservoir study site one) and is one of the first dams built in Canada for flood control purposes (Boyd *et al*., 2000).

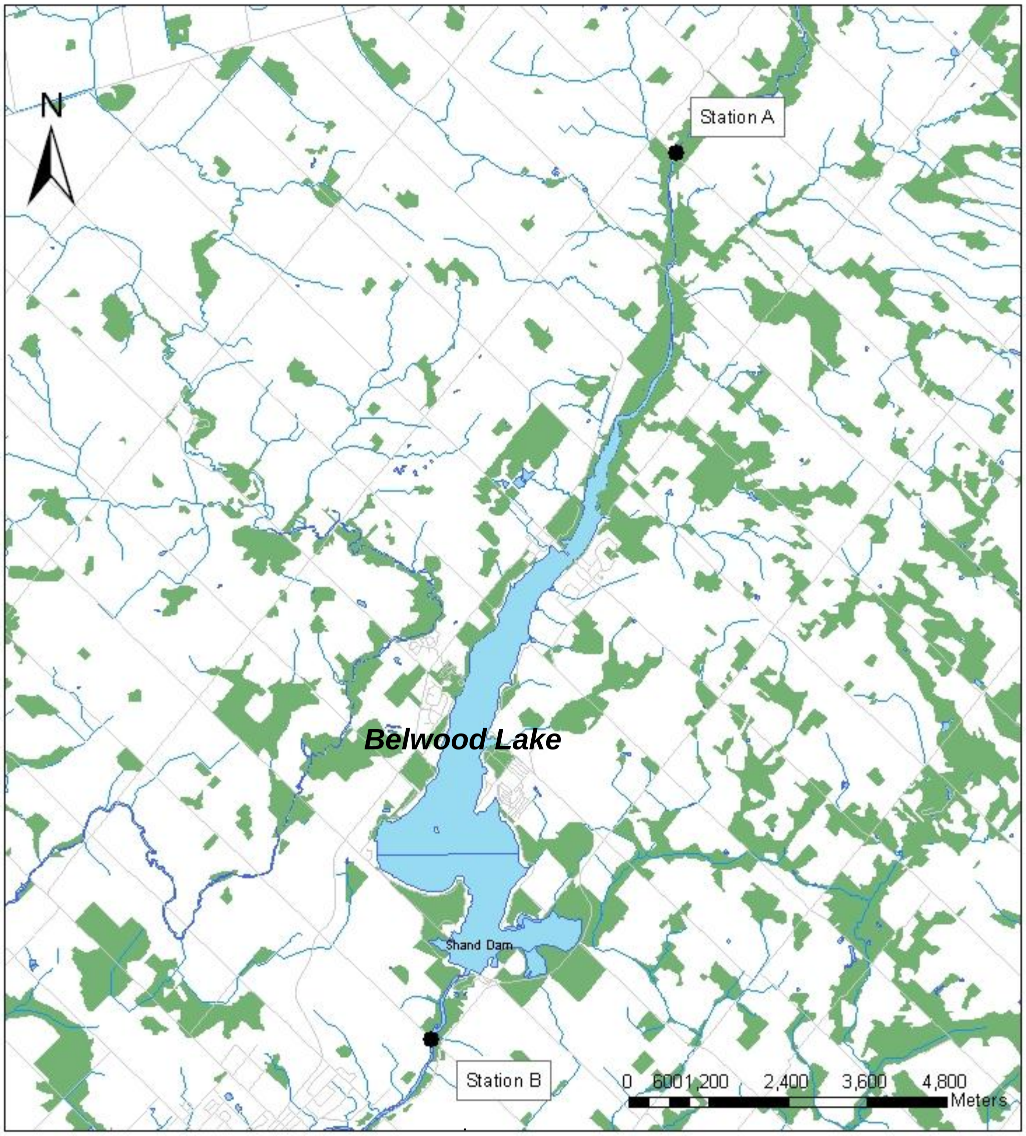

Figure 2.4. Study site one (Belwood Lake, Shand Dam).

Today it is managed for flood control and flow augmentation. The Shand Dam is a deep release reservoir that empties into the Grand River (Boyd *et al*., 2000). The reservoir is drawn down gradually from midsummer to the fall by a total of 7.13 meters and is refilled in the spring (Boyd *et al*., 2000). Reference station, 1A, is located 12.51 km upstream of the reservoir and accessed by a bridge on the 11th Line, approximately 0.03 km east of Highway 5. Station 1B is located 1.23 km downstream of the impoundment and accessed by way of a bridge on 2nd Line, 0.01 km west of Highway 18.

2.2.2 Conestogo Lake

Conestogo Dam (reservoir study site two) was built in 1958 on the Conestogo River (Boyd *et al*., 2000) and creates a y-shaped lake (Figure 2.5) which has two arms that stretch 6 km each. The impoundment is a multipurpose reservoir managed today for both flood control and flow augmentation (Boyd *et al*., 2000). Conestogo Dam is a deep release reservoir that is gradually released through the summer and early fall by a total of 8.37 m and refilled in the spring (Boyd *et al*., 2000). Reference station, 2A, is located 9.50 km upstream of the reservoir and accessed by River Run Road off Highway 11 in the Village of Drayton. The site is located behind a residential area in an undeveloped forest and is undisturbed from agricultural practices which dominate the surrounding area. Station 2B is located 9.75 km downstream of the impoundment and accessed by way of a bridge on Highway 86 in the Village of Wallenstein.

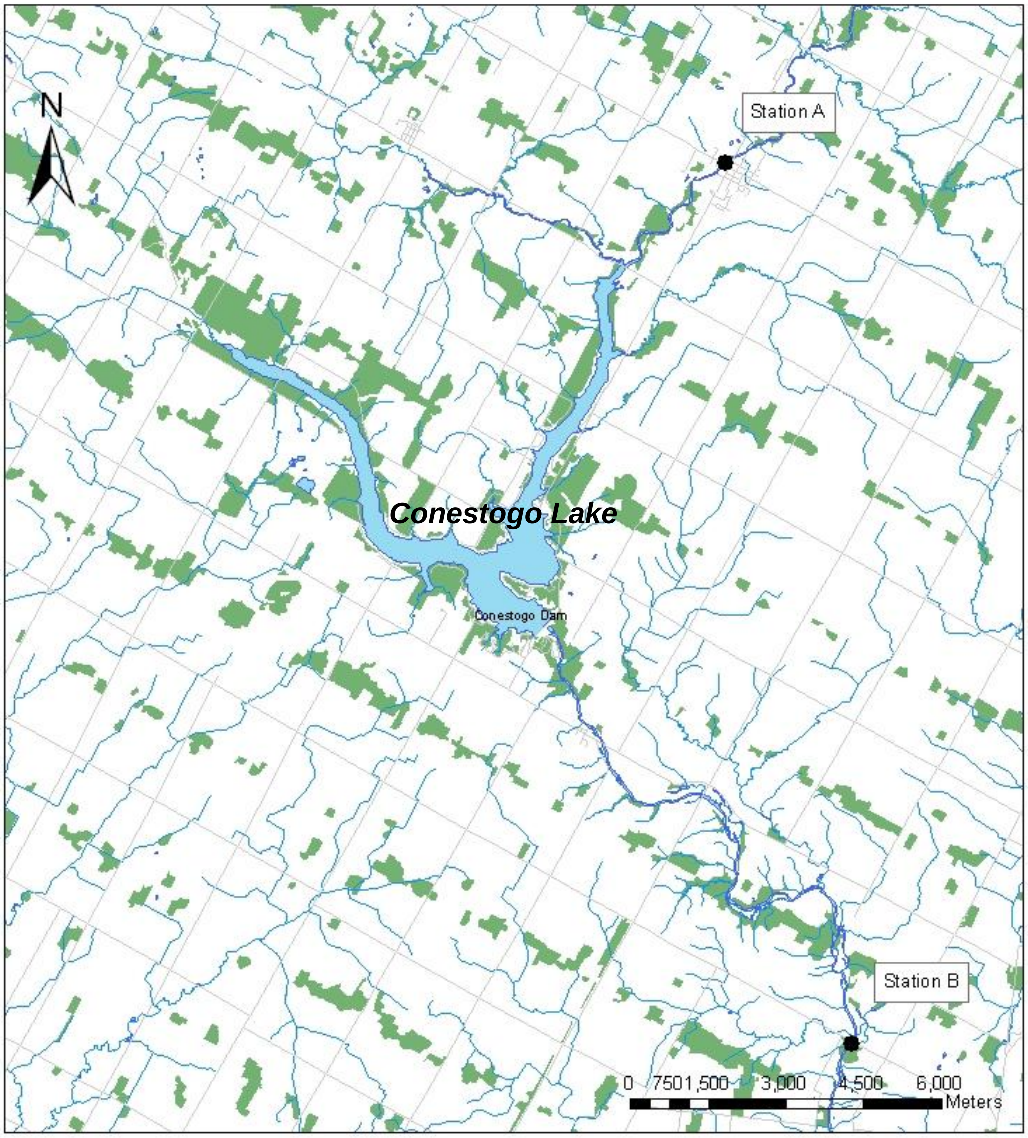

Figure 2.5. Study site two (Conestogo Lake, Conestogo Dam).

2.2.3 Guelph Lake

Guelph Lake (Figure 2.6), located on the northeast edge of the City of Guelph, Wellington County, was created in 1972 with the construction of Guelph Dam (reservoir study site three) (Boyd *et al*., 2000). This deep release reservoir, which flows into the Speed River, is managed today for both flood control and flow augmentation. It is also used for recreational purposes with the adjacent 3971 ha conservation area (Boyd *et al*., 2000). Guelph Lake is slowly drawn down in the summer and early fall by 1.89 m and filled in the spring (Boyd *et al*., 2000). Reference station, 3A, is located 2.80 km upstream of the reservoir and accessed by a bridge on Mill Rd, 0.03 km south of Jones Baseline, in a residential development located outside the city centre. Station 3B is located 1.03 km downstream of the impoundment and accessed by a bridge on Victoria Road North, 0.02 km south of Conservation Road.

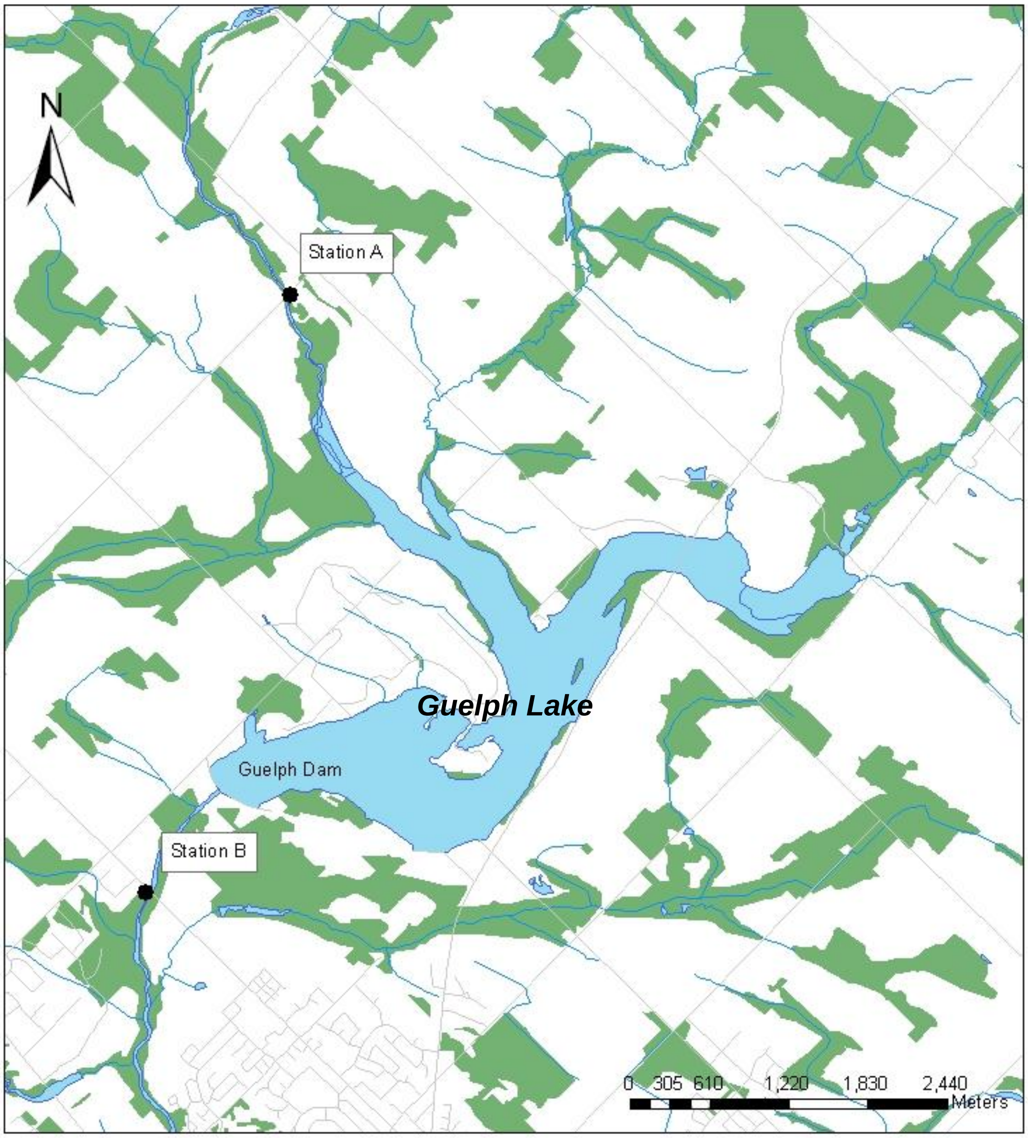

Figure 2.6. Study site three (Guelph Lake, Guelph Dam).

2.2.4 Mill Creek

Mill Creek (Figure 2.7) is a small tributary of the Grand River that flows from Puslinch Township south of the City of Guelph and southwest to the City of Cambridge (Boyd *et al*., 2000).

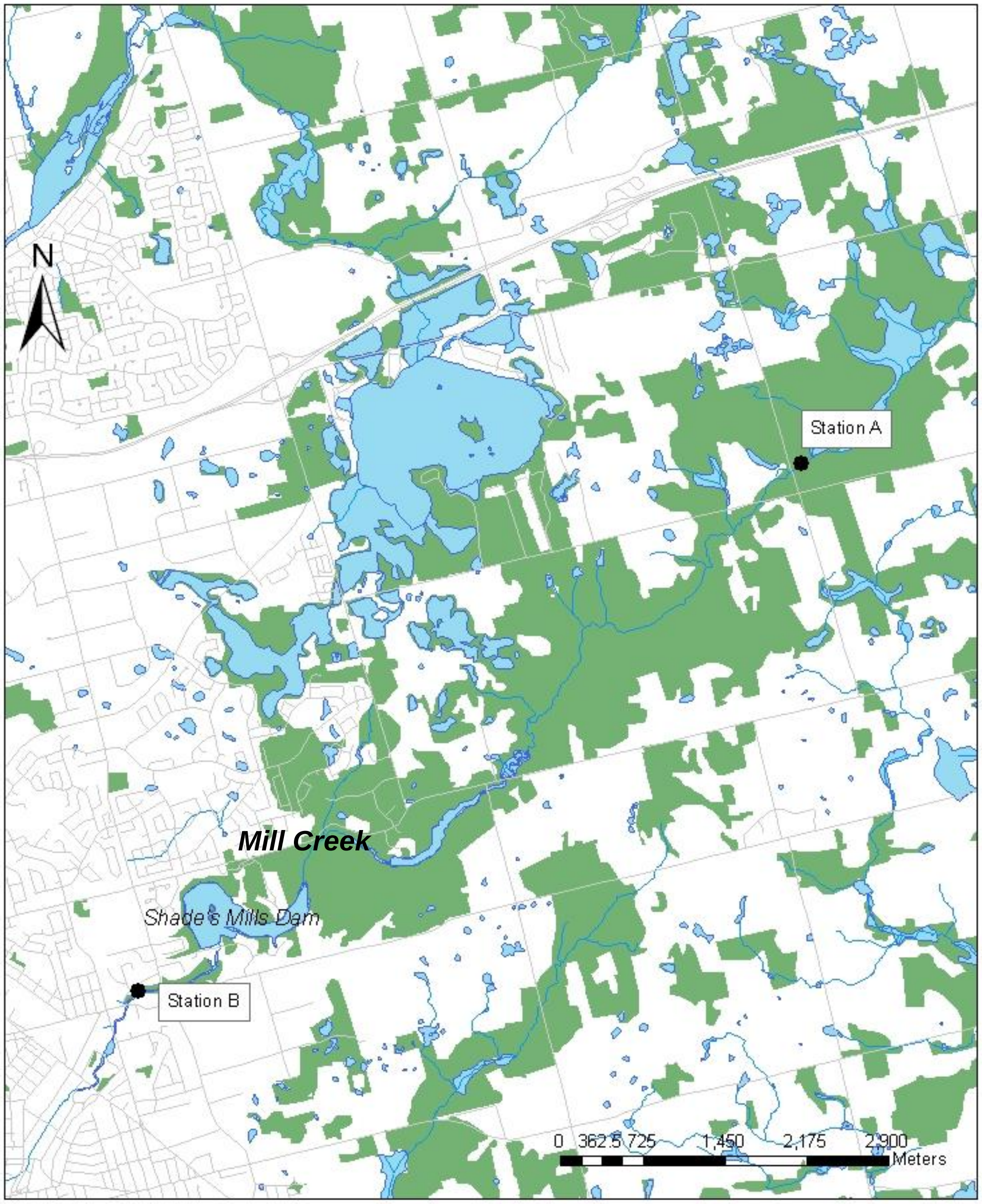

Figure 2.7. Study site four (Shade's Mills Dam).

Mill Creek is regulated by the Shade's Mills dam (reservoir study site four) (Figure 2.7), located on the eastern edge of the City of Cambridge in the Waterloo Region, and is currently managed for flood control (Boyd *et al*., 2000). Shade's Mills dam was originally constructed in the 1800's but has since then been reconstructed in 1966. This 36 ha surface release reservoir is drawn down in the fall (November) by 1.57 meters and refilled in the spring (April) depending on precipitation (Boyd *et al*., 2000). Station 4A is located 10.8 km upstream of the reservoir, in a residential area north of the city centre on Sideroad 10 South 0.01km north of Concession 1. Station 4B is located 0.83 km downstream of the impoundment and accessed by way of a bridge on Clyde Rd 0.03 km west of Franklin Blvd.

2.2.5 Laurel Creek

Laurel Creek (Figure 2.8) is located in the northwest portion of the City of Waterloo in the Waterloo Region and flows through the centre of the city before emptying into the Grand River (Barton *et al*., 2000). Laurel Creek Reservoir (study site five), was built in the 1830's but has since been reconstructed in 1967 by the GRCA for flood control purposes (Boyd *et al*., 2000). Moreover, Laurel Creek has two smaller downstream impoundments, Columbia Lake, 1967 and Silver Lake, constructed in 1960, which are mainly used for recreation (GRCA, 2004). Laurel creek reservoir has a surface area of 67 ha and a mean depth of 1.3 m (Sephton *et al*., 1983). The reservoir is drawn down in the fall to the original stream channel (3.03 meters) and refilled in the spring (Boyd *et al*., 2000). Reference station, 5A, is located 1.0 km upstream near the Laurel Creek Nature Centre, where Laurel Creek runs through the 47 ha property. Station B, is located 0.05 km downstream of the impoundment.

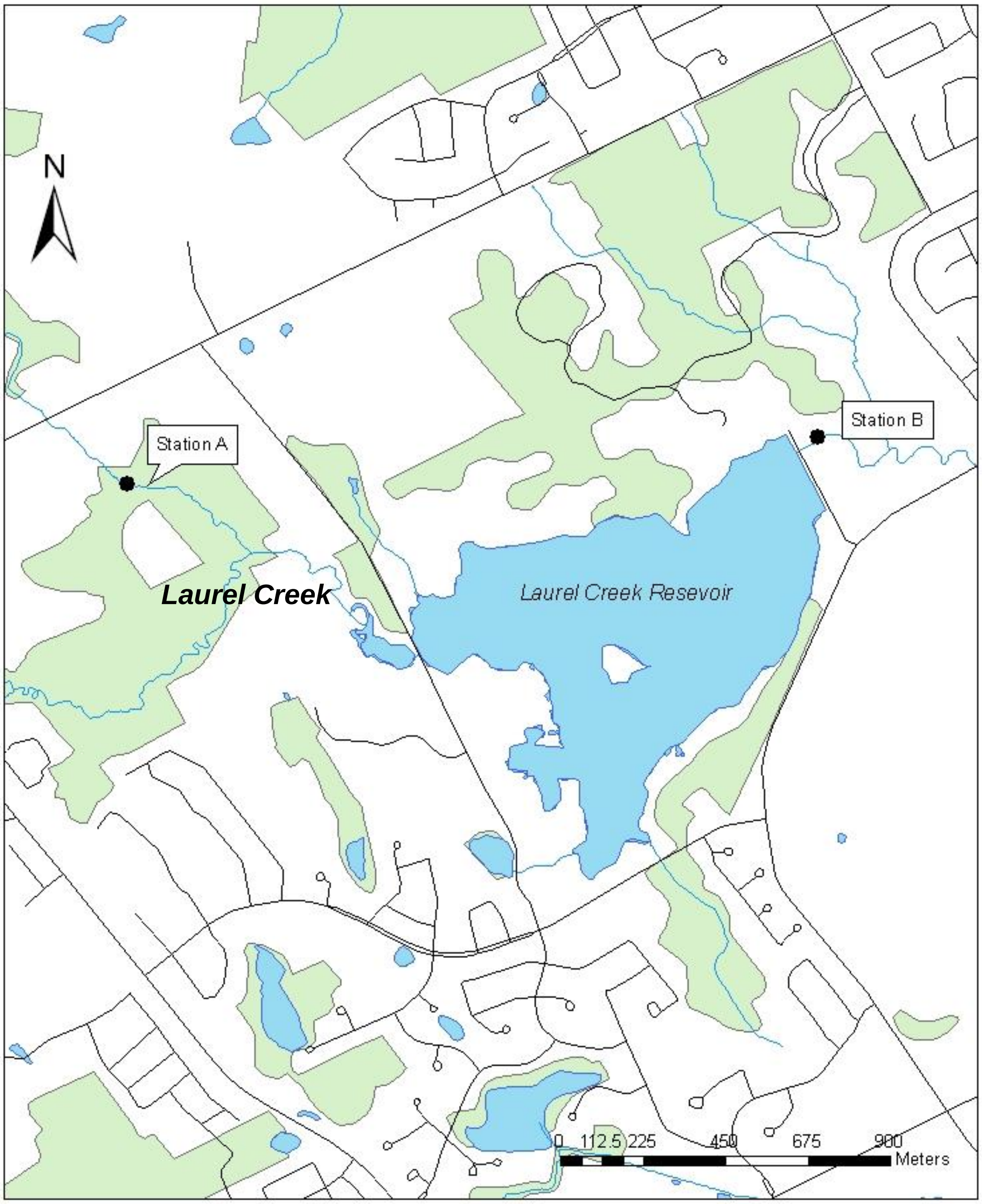

Figure 2.8. Study site five (Laurel Creek Reservoir).

2.3 Field Methods

2.3.1 Rainfall, Velocity and Water Quality

Field work was conducted during three sampling periods: from May 30 to June 12, 2006 (sampling period 1), from August 9-11, 2006 (sampling period 2), and following reservoir drawdown, (sampling period 3), from November 6-9, 2006. All sampling took place between 10:00 and 17:00 hours. There were no major rainfall events 14 days prior to sampling. Precipitation data were obtained from the University of Waterloo Weather Station (43^o 28'N, 80^o 33'W). Rainfall is measured automatically at 15 minute intervals using a tipping bucket (Texas Electronics® Model TE525).

Average stream velocity was measured using a Sigma® Portable Velocity Meter and depth was recorded using a meter stick. Average velocity and depth were recorded when invertebrate samples were collected and a total of six velocity and depth measurements was taken at each transect. Substrate type and predominant types of riparian vegetation (grasses, shrubs or trees) were recorded at each station. In addition, distance from the stream bank to the outer edge of the riparian buffer was measured.

Air temperature, water temperature and electrical conductivity were measured using an Orion® Model 105 conductivity and temperature meter. Conductivity and temperature were used to calculate total dissolved solids (TDS) concentration using the following equation (APHA, 2005):

$$\text{TDS (mgL}^{-1}) = \left(\frac{\text{Initial Conductivity } (\mu\text{scm}^{-1})}{1+ [0.02 \text{ (cell Temperature } -25^o\text{C)}]} \right) * 0.666$$

Dissolved oxygen (DO) levels were recorded using a YSIR Model 57 oxygen meter and pH was measured using an Orion R Model 720A pH meter. All meters were calibrated in the laboratory each day before sampling.

2.4 Benthic Macroinvertebrate Sampling

Benthic macroinvertebrate samples were collected in spring, summer and fall, upstream and downstream of the impoundments using a T-sampler, with an area of 103.8 cm^2 and mesh size of 250µm, in stream riffles (Jones *et al.*, 2004). Although there are several advantages and disadvantages of benthic sampling at different times of year (Barton, 1996; Barbour *et al*., 1999; Jones *et al*., 2004**)** (Table 2.4), a minimum of three replicates (sampling periods) is recommended to strengthen confidence in the estimates of means (Jones *et al*., 2004).

Table 2.4. **Advantages and disadvantages of benthic sampling at different times of year (Barton, 1996; Barbour *et al*., 1999; Jones *et al*., 2004).**

Season	Advantages	Disadvantages
Winter (January-February)	▪ High richness. ▪ Animals are large and easily identified.	▪ Difficult or unsafe conditions prevail. ▪ Community composition may not reflect water quality.
Spring (May-June)	▪ High richness. ▪ Animals are large and easily identified.	▪ Short sampling period between spring freshet or ice-out.
Summer (July-August)	▪ Most stressful season due to high water temperature and low oxygen levels. ▪ Invertebrates are likely to show a response to impacts.	▪ Variable richness. ▪ Drought conditions.
Fall (October-November	▪ High richness. ▪ Composition may reflect summer impacts.	▪ Prevalence of small juveniles (difficult to identify). ▪ Community composition may not reflect summer quality.

At each station, upstream and downstream from the reservoir, six samples were collected along a stream riffle transect perpendicular to the direction of flow. Following collection, the benthic macroinvertebrate samples were washed into 120mL Starplex Scientific® sterile sample jars and preserved in 10% formalin (Kilgour and Barton, 1999).

2.5 Laboratory Methods

Approximately 48 hours after collection, samples were transferred to 70% ethanol (Jones *et al*., 2004). Samples were sorted with a dissecting scope (Wild Heerbrugg ® Model 5A) and illuminator (Chiu Technical Corporation ® Lumina Model F0-150). Invertebrates were removed from the sediment and placed in 10mL vials containing 70% ethanol, then identified using the dissecting scope and illuminator following dichotomous keys found in Merrit and Cummins (1996), Thorp and Covich (2001) and Mackie (2000). Each organism was identified to the lowest practical taxonomic level, typically genus or species, with the exception of Nematoda identified to phylum, Hydrachnida to Order, Oligochaeta to family and Chironomidae identified to subfamily or tribe. Despite variation in the level of identification, samples are comparable because levels of identification were consistent among all samples (Barbour *et al*., 1999).

2.6 Summary Indices

The benthic macroinvertebrate data were summarized into nine indices that have previously been used in various studies to assess water quality and river regulation impacts on benthic macroinvertebrates (Barbour *et al*., 1999; Jones *et al*., 2004). The indices include: abundance (number per sample), taxa richness, EPT taxa richness (number of individuals), HBI, relative dominance, percent Ephemeroptera, Isopoda, and Chironomidae and percent Filter-Feeders. The literature indicates that each index is expected to vary (increase or decrease) in a predictable manner in response downstream reservoir outflow (Table 2.5).

The benthic macroinvertebrate summary indexes can be used to detect environmental changes downstream of reservoirs. Changes in habitat diversity and water quality imposed by impoundments often reduce taxa richness (Kilgour and Barton, 1999), while changes in organic matter content and vegetation can encourage benthic macroinvertebrate abundance downstream (Spence and Hynes, 2001). EPT taxa richness is a good indicator of water quality for these sensitive species of benthic macroinvertebrates have a high demand for DO and often decrease if the water quality and DO levels are low (Lenat, 1988). The Hilsenhoff’s Biotic Index is a good indicator of detecting organic enrichment (Hilsenhoff, 1987), while Isopoda are particularly responsive to organic loading (Whitehurst, 1991). Chironomidae are more tolerant to habitat and water quality alterations and through competitive interactions, are often more abundant following these impacts (Hynes, 1960). Filter-feeders are more responsive to changes in the production of FPOM, which often increase downstream of reservoirs (Kerans and Karr, 1994).

Table 2.5. Benthic macroinvertebrate summary indices and predicted responses downstream from reservoir outflow.

Benthic Macroinvertebrate Summary Index	Predicted Response to Increasing Perturbation	Source
Abundance	Increase or Decrease	Paul and Meyer, 2001
Taxa Richness	Increase or Decrease	Kilgour and Barton, 1999
Ephemeroptera, Plecoptera, Trichoptera (EPT) taxa richness	Decrease	Lenat, 1988
Hilsenhoff's Biotic Index (HBI)	Increase	Hilsenhoff, 1987; Hilsenhoff, 1988
Relative abundance of the dominant taxon	Increase	Barbour *et al*., 1999
Percent Ephemeroptera	Decrease	Hynes, 1970
Percent Isopoda	Increase	Whitehurst, 1991
Percent Chironomidae	Increase (or Decrease)	Hynes; 1970; Barbour *et al*., 1999
Percent Filter-Feeders	Increase	Kerans and Karr, 1994; Barbour *et al*., 1999

2.7 Statistical Methods

Statistical analyses were performed using SPSS® for Windows (Version 12.0.1). Tests of normality (Q-Q plots using studentized residuals) indicated that the data were not normally distributed. In order to meet the assumption of normality for ANOVA, data were natural log transformed (Harvey, E., pers. Com., 2007). A repeated measures ANOVA was used to determine if there were significant differences in each index between reservoirs, sampling stations and sampling periods.

Chapter 3: Results

3.1 Introduction

The following chapter presents the results and trends of environmental, water quality and benthic macroinvertebrate data recorded and collected during three sampling periods (May 30-June 12, 2006; August 9-11, 2006; November 6-9, 2006), hereafter referred to as spring, summer and fall respectively. These data will be further discussed in detail in Chapter 4.

3.1.1 Temperature and Precipitation

Mean daily temperature and total precipitation (Figures 3.1-3.3) were measured at the University of Waterloo Weather Station during spring, summer and fall sampling. While taking precipitation measurements at each station, above and below reservoirs, would have been more accurate than using the Weather Station data, it was not feasible to do so in the present study. Therefore, although the meteorological data are suitable for Laurel Creek Reservoir, located in Waterloo, they are an approximation of the changes in temperature and precipitation for reservoirs, Shand Dam, Conestogo Dam, Guelph Dam and Shade's Mills. Mean temperature was highest during summer (11.1 ^{o}C to 25.3 ^{o}C) and was lowest during fall (6.0 ^{o}C to 9.8 ^{o}C). Spring mean temperature ranged from 16.6 ^{o}C to 20.2 ^{o}C. Total precipitation was highest during spring which accumulated 50.0 mm and lowest during summer with 0.0 mm, while fall precipitation accumulated 11.2 mm. There were no major rainfall events during spring and summer, however, a major rainfall (33.8 mm) occurred in spring which delayed sampling because of high stream discharges.

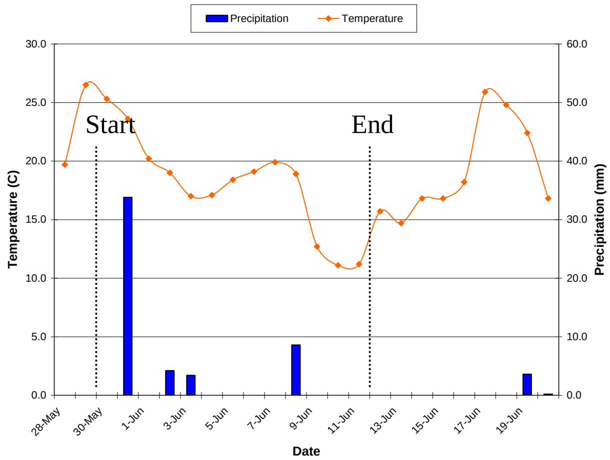

Figure 3.1. Mean temperature and total precipitation: spring.

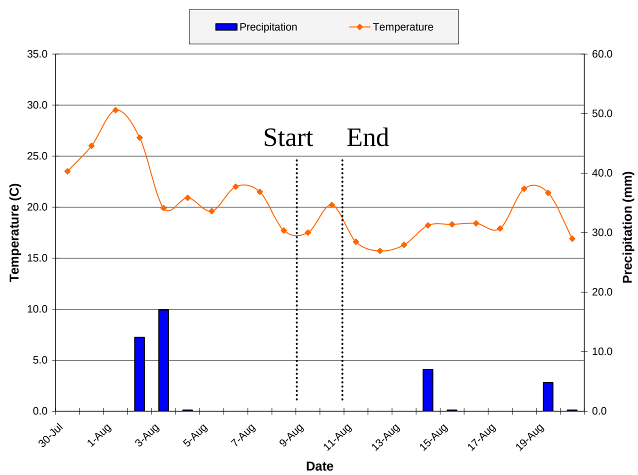

Figure 3.2. Mean temperature and total precipitation: summer.

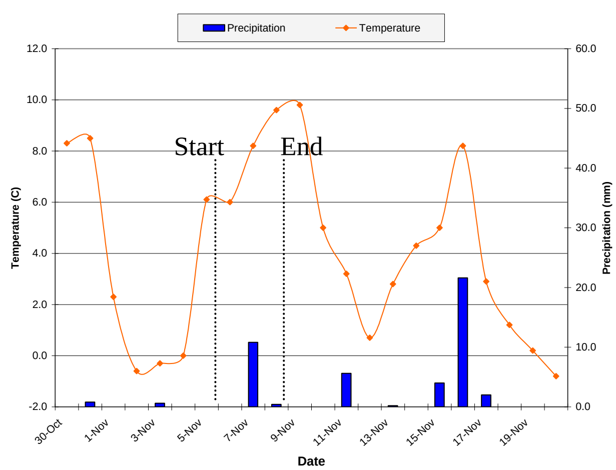

Figure 3.3. Mean temperature and total precipitation: fall.

3.1.2 Stream Velocity and Depth

Stream velocity and depth were recorded during benthic sampling above and below the five reservoirs (Figures 3.4-3.13; Table 3.1). During spring, mean velocities ranged from 0.29 ms^{-1} at station 5B to 0.813 ms^{-1} at station 2A. Mean velocity ranged from 0.24 ms^{-1} at station 2B to 0.70 ms^{-1} at station 3B during summer and from 0.27 ms^{-1} at station 5B to 0.88 ms^{-1} in station 2B during fall. The highest velocity (1.10 ms^{-1}) was recorded during fall and was lowest in summer (0.09 ms^{-1}). Velocity was generally higher for station A compared to station B for all sampling periods.

Mean water depth ranged from a low of 0.151 m at station 1B to a high of 0.294 m at station 3A during spring. In summer, the mean depth ranged from 0.111 m in station 5A to 0.217 m in station 4B, while the mean depth ranged from 0.275 m at station 3A to 0.434 m at station 2A during fall. The mean depth was highest in fall and lowest in summer. Mean depth was on average higher at station B compared to station A during spring and summer. However, mean depth was lower at station B during fall.

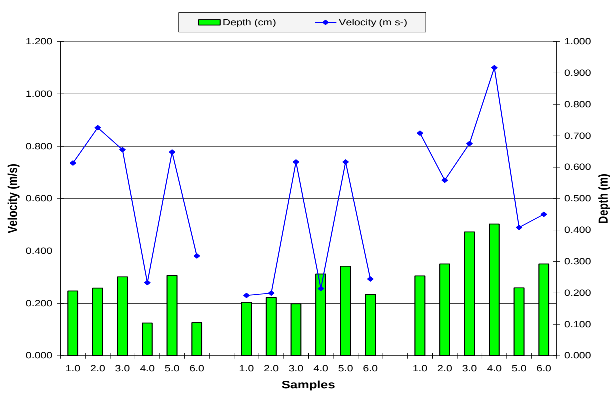

Figure 3.4. Velocity and depth measurements recorded in spring, summer and fall: 1A.

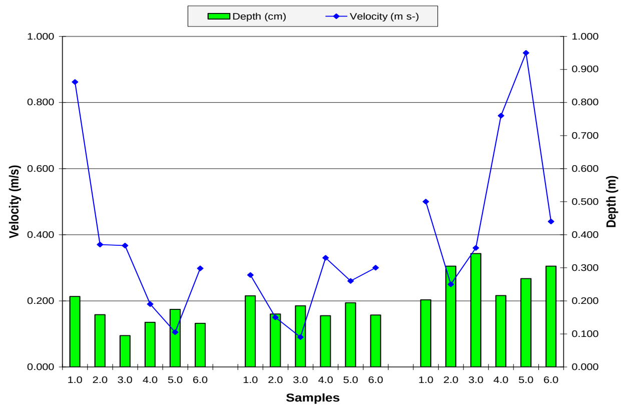

Figure 3.5. Velocity and depth measurements recorded in spring, summer and fall: 1B.

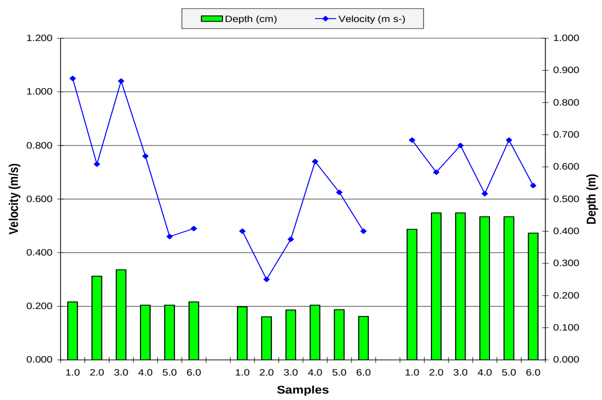

Figure 3.6. Velocity and depth measurements recorded in spring, summer and fall: 2A.

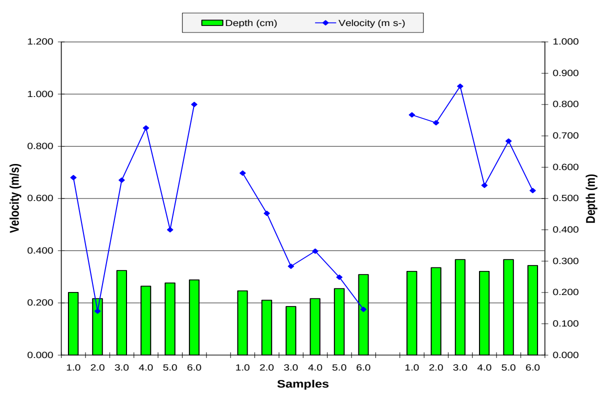

Figure 3.7. Velocity and depth measurements recorded in spring, summer and fall: 2B.

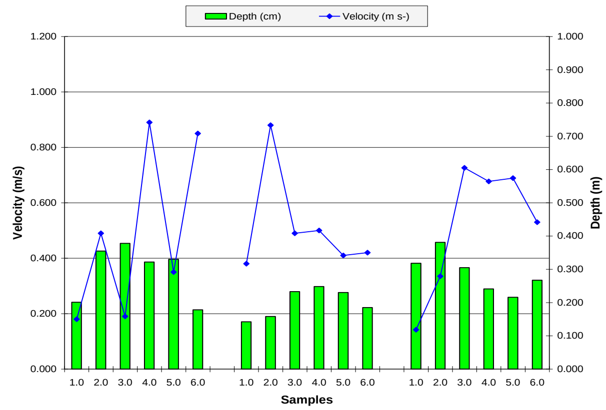

Figure 3.8. Velocity and depth measurements recorded in spring, summer and fall: 3A.

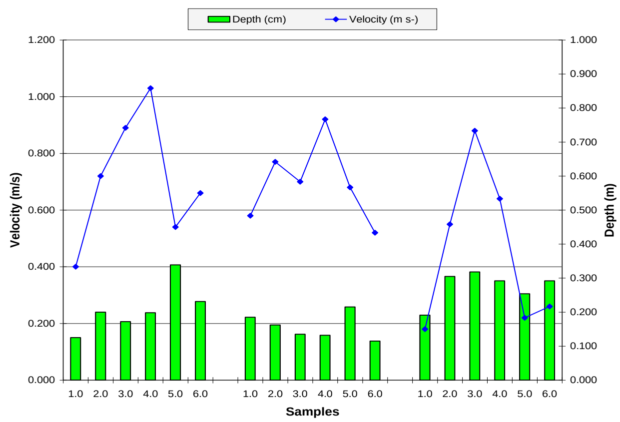

Figure 3.9. Velocity and depth measurements recorded in spring, summer and fall: 3B.

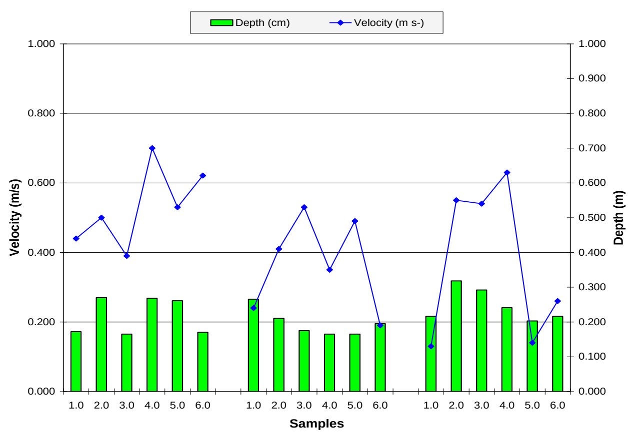

Figure 3.10. Velocity and depth measurements recorded in spring, summer and fall: 4A.

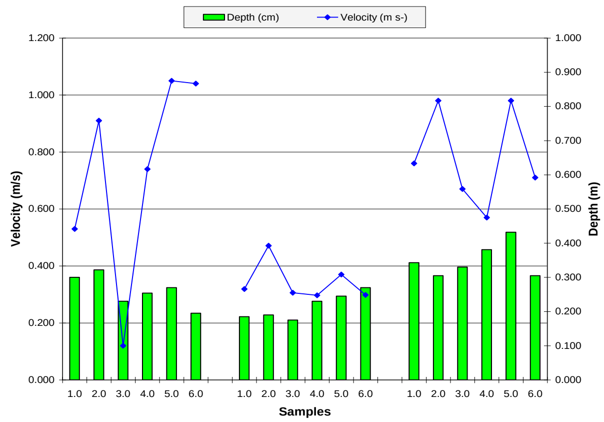

Figure 3.11. Velocity and depth measurements recorded in spring, summer and fall: 4B.

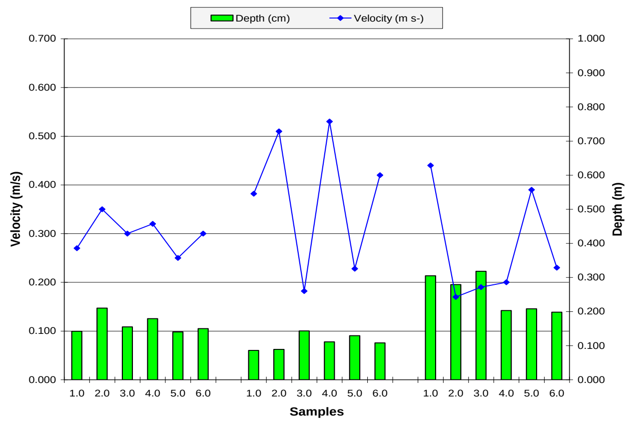

Figure 3.12. Velocity and depth measurements recorded in spring, summer and fall: 5A.

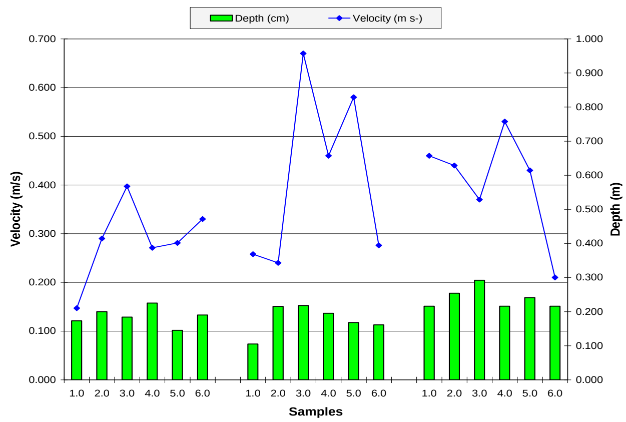

Figure 3.13. Velocity and depth measurements recorded in spring, summer and fall: 5B.

Table 3.1. Mean velocity and depth values recorded during sampling.

Reservoir	Sampling Period	Station	Mean Velocity (ms^{-1})	Mean Depth (m)
1 Shand	1	A	0.64	0.189
		B	0.37	0.151
2 Conestogo	1	A	0.81	0.207
		B	0.64	0.223
3 Guelph	1	A	0.49	0.294
		B	0.75	0.211
4 Shade's Mills	1	A	0.53	0.218
		B	0.77	0.262
5 Laurel Creek	1	A	0.30	0.163
		B	0.29	0.186
1 Shand	2	A	0.42	0.210
		B	0.24	0.178
2 Conestogo	2	A	0.51	0.153
		B	0.41	0.197
3 Guelph	2	A	0.51	0.199
		B	0.70	0.157
4 Shade's Mills	2	A	0.37	0.196
		B	0.34	0.217
5 Laurel Creek	2	A	0.38	0.111
		B	0.41	0.177
1 Shand	3	A	0.72	0.311
		B	0.54	.0.273
2 Conestogo	3	A	0.74	0.434
		B	0.79	0.285
3 Guelph	3	A	0.52	0.288
		B	0.46	0.295
4 Shade's Mills	3	A	0.38	0.248
		B	0.78	0.349
5 Laurel Creek	3	A	0.88	0.285
		B	0.27	0.251

3.1.3 Water Quality Measurements

Air and water temperatures, pH, dissolved oxygen and conductivity were recorded in the field (Table 3.2). Water temperatures in spring ranged from 14.8^{o}C at station 5B on June 12, 2006 to 25.1^{o}C at station 1A on May 30, 2006. During summer, water temperatures ranged from a minimum of 17.4^{o}C at station 4A on August 11, 2006 to a maximum of 27.3^{o}C at station 1A on August 11, 2006. Water temperatures in fall ranged from 4.7^{o}C at station 3A on November 11, 2006 to 6.9^{o}C at station 1A on November 8, 2006. During spring and summer, water temperatures at station B, downstream of deep release reservoirs (Shand Dam, Conestogo Dam, Guelph Dam) were on average 5.3^{O}C lower than Station A during spring and 1.7^{o}C lower than Station A during summer. However, water temperatures at station B below the surface release reservoirs (Shade's Mills Dam, Laurel Creek Reservoir) were on average 4.0^{o}C higher than at station A during spring and 6.4^{o}C higher than station A during summer. In contrast, water temperatures at Station B for all deep release reservoirs in fall after drawdown were on average 0.6^{o}C higher than station A, and water temperatures in surface release reservoirs were on average 0.2^{o}C lower at station B in comparison to Station A.

In spring, pH ranged from 7.19 at station 4A on June 7, 2006 to 8.45 at station 3B on June 7, 2006. In summer pH ranged from 7.89 at station 4A on August 8, 2006 to 8.70 at station 1B on August 10, 2006 and during fall, pH ranged from a low of 7.59 at station 5A on November 7, 2006 to a maximum of 8.59 at station 3B on November 6, 2006. On average, the pH was higher at station B than station A in all sampling periods.

Dissolved oxygen (DO) levels ranged from 10.23 mgL^{-1} at station 1A on May 30, 2006, to 15.67 mgL^{-1} at station 2B on June 12, 2006 in spring. In summer, DO levels ranged from 6.85 mgL^{-1} at station 5B on August 9, 2006 to 13.58 mgL^{-1} at station 2A on August 11, 2006, and in fall DO levels ranged from 9.41 mgL^{-1} at station 5B on November 11, 2006 to 16.29 mgL^{-1} at station 3A on November 6, 2006. In all stations, with the exception of deep release reservoir, Shand Dam (4), DO levels were higher at station A in comparison to station B for all sampling periods.

Table 3.2. Air and water temperatures, pH, DO, conductivity and TDS measurements.

Reservoir	Sampling Period	Date	Station	Air (°C)	Water (°C)	pH	Dissolved Oxygen (mgL^{-1})	Conductivity (μS)	TDS (mgL^{-1})
1 Shand	1	30-05-06	A	30.9	25.1	7.80	10.23	414	275
			B	28.5	15.2	8.13	11.16	307	254
2 Conestogo	1	12-06-06	A	23.5	19.7	7.24	15.48	420	313
			B	22.0	18.4	8.14	15.20	367	282
3 Guelph	1	07-06-06	A	24.0	21.2	7.70	12.08	478	345
			B	23.5	16.6	8.45	11.68	381	305
4 Shade's Mills	1	07-06-06	A	22.5	17.2	7.19	11.96	534	421
			B	22.5	18.7	7.98	10.95	536	408
5 Laurel Creek	1	12-06-06	A	19.5	14.8	7.95	10.26	427	357
			B	23.0	21.2	8.41	9.29	304	219
1 Shand	2	10-08-06	A	26.6	25.6	8.46	10.32	478	315
			B	26.5	22.6	8.70	10.84	390	273
2 Conestogo	2	11-08-06	A	27.5	27.3	8.40	13.58	503	320
			B	27.0	21.7	8.60	12.51	411	293
3 Guelph	2	10-08-06	A	25.9	24.5	8.12	11.51	503	352
			B	25.8	22.6	8.24	11.09	442	303
4 Shade's Mills	2	11-08-06	A	24.5	17.4	7.89	9.57	581	456
			B	23.0	21.8	8.14	7.39	605	430
5 Laurel Creek	2	09-08-06	A	25.8	17.9	8.26	8.63	524	407
			B	27.0	26.3	8.32	6.85	411	267
1 Shand	3	08-11-06	A	10.5	6.9	8.28	12.07	595	621
			B	10.7	7.1	8.49	12.33	485	537
2 Conestogo	3	09-11-06	A	9.1	6.2	8.44	14.84	629	671
			B	8.5	6.5	8.49	14.25	622	662
3 Guelph	3	06-11-06	A	7.4	4.7	8.57	16.29	550	617
			B	7.5	5.9	8.59	14.79	486	524
4 Shade's Mills	3	06-11-06	A	8.2	5.7	8.31	12.48	593	643
			B	7.3	5.5	8.34	9.55	557	606
5 Laurel Creek	3	07-11-06	A	8.1	5.9	7.59	12.30	465	501
			B	8.2	5.8	8.10	9.41	607	620

Conductivity measurements ranged from 536 µS at station 4B on June 7, 2006 to 304 µS at station 5B on June 12, 2006 in spring. In summer, conductivity ranged from a low of 390 µS at station 1B on August 10, 2006 to a high of 605 µS at station 4B on August 11, 2006, while conductivity ranged from 465 µS at station 5A on November, 2006 to 629 µS at station 3A on August 29, 2006 during fall. On average, conductivity was higher in station A compared to station B in all sampling periods.

Total dissolved solids (TDS) concentration ranged from 219 mgL^{-1} at station 5B on June 12, 2006 to 421 mgL^{-1} at station 4A on June 7, 2006 in spring. During summer TDS ranged from a minimum of 267 mgL^{-1} at station 5B on August 8, 2006 to a maximum of 456 mgL^{-1} at station 4A on August 11, 2006 and during fall TDS ranged from 507 mgL^{-1} at station 5A on November 7, 2006 to 671 mgL^{-1} at station 3A on November 9, 2006. On average in all sampling periods, TDS concentrations were higher at station A compared to Station B.

3.2 Benthic Macroinvertebrate Communities

3.2.1 Sample Size

One hundred and eighty samples were collected from ten stations throughout three sampling periods and benthic macroinvertebrate indices for each station are presented in Table 3.3. The values reported represent the means at each sampling station and results are discussed in more detail in the following sections. A complete taxonomic list for each sample is listed in Appendix B.

Table 3.3. Mean index values for benthic macroinvertebrates communities in stations upstream (A) and downstream (B) of reservoirs.

Reservoir	Sampling Period	Station	Abundance (number per sample)	Taxa Richness	EPT Taxa	HBI	% Dominant Taxa	% Ephemeroptera	% Isopoda	% Chironomidae	% Filter-Feeders
1 Shand	1	A	129.2	17.2	6.7	4.2	38.9	38.9	0.0	18.1	8.4
		B	270.3	12.5	2.8	5.5	45.8	0.4	14.1	45.8	27.7
2 Conestogo	1	A	283.7	18.5	6.0	4.5	28.5	16.5	0.9	27.7	23.7
		B	332.0	16.8	3.7	4.8	43.1	4.8	20.7	43.1	12.1
3 Guelph	1	A	102.3	14.7	6.0	4.1	70.7	70.7	0.1	7.8	6.7
		B	239.2	12.7	2.3	5.5	41.5	0.1	33.8	41.5	2.5
4 Shade's Mills	1	A	161.0	16.5	5.0	4.1	49.0	12.0	0.0	49.0	4.7
		B	175.5	14.0	3.5	4.7	43.4	2.9	1.5	19.1	52.5
5 Laurel Creek	1	A	201.8	12.5	3.7	4.3	68.6	1.1	0.0	68.6	10.7
		B	394.8	11.3	1.2	5.8	71.6	0.0	0.0	71.6	20.8
1 Shand	2	A	165.0	19.5	8.3	4.2	27.6	21.1	0.5	27.6	13.5
		B	296.3	12.5	3.3	5.8	67.2	0.8	67.2	8.9	4.0
2 Conestogo	2	A	378.3	20.8	8.2	4.5	28.7	27.2	0.0	17.0	14.9
		B	401.2	17.0	5.3	5.0	32.3	2.7	32.3	18.7	13.4
3 Guelph	2	A	186.8	18.3	6.0	4.3	39.3	39.3	0.1	27.6	3.7
		B	350.5	13.7	2.7	5.6	65.1	0.2	65.1	7.3	3.4
4 Shade's Mills	2	A	462.2	20.2	7.8	4.4	55.0	13.0	0.1	55.0	8.2
		B	530.2	16.3	5.2	4.7	31.0	7.6	5.2	4.1	30.2

Table 3.3. (Continued) Mean index values for benthic macroinvertebrates communities in stations upstream (A) and downstream (B) of reservoirs.

Reservoir	Sampling Period	Station	Abundance (number per sample)	Taxa Richness	EPT Taxa	HBI	% Dominant Taxa	% Ephemeroptera	% Isopoda	% Chironomidae	% Filter-Feeders
5 Laurel Creek	2	A	303.7	16.5	5.8	3.7	53.2	3.8	0.0	53.2	22.1
		B	401.2	11.2	2.2	5.4	66.2	0.3	0.0	66.2	5.4
1 Shand	3	A	224.5	15.2	6.5	4.3	29.1	20.7	0.0	27.2	20.2
		B	80.8	14.0	2.5	5.3	31.5	0.6	22.8	31.5	6.7
2 Conestogo	3	A	407.0	19.5	7.8	4.4	68.2	11.0	0.0	68.2	10.8
		B	100.7	14.8	4.8	4.9	29.6	2.9	16.0	27.7	34.2
3 Guelph	3	A	278.5	21.0	8.7	3.4	37.8	37.8	0.0	23.0	6.1
		B	75.0	11.0	1.7	6.0	29.0	2.5	29.0	24.8	14.9
4 Shade's Mills	3	A	221.2	17.2	7.7	4.7	53.0	23.4	0.0	6.4	47.2
		B	175.0	14.0	3.8	5.0	44.3	3.9	1.5	17.5	58.3
5 Laurel Creek	3	A	337.0	16.2	6.3	3.8	45.9	11.6	0.0	10.4	48.4
		B	298.2	11.8	2.2	6.0	66.6	1.0	0.0	66.6	9.4

3.2.2 Abundance and Richness

In total, 47, 780 invertebrates were collected from 180 samples during three sampling periods. On average, 265.4 ±11.6 (mean ± 1 standard error of the mean [SEM]) organisms were collected per sample. Abundance per sample ranged from 19 invertebrates from sample 5B1 in fall to 914 invertebrates from sample 4A4 in summer. Mean abundance ranged from 75.0 ±6.6 at station 3B in fall to 530.2 ±16.1 at station 4B in summer (Figures 3.14-3.16). Although mean abundance per sample in all stations varied among sampling periods, mean abundance was higher at station B compared to station A in spring and summer. Mean abundance was lowest at station B at all stations during fall following reservoir drawdown.

In total sixty-eight taxa were collected and number of taxa per sample ranged from a low of 8 from sample 5B6 in spring, sample 5B3 in summer and sample 3B5 in fall, to a high of 26 from sample 3A2 in the fall. The average taxa richness ranged from 11 ±0.7 at station 3B to 21 ±1.0 at station 3A during fall (Figures 3.17-3.19). Mean taxa richness in all stations were highest at station A in all sampling periods.

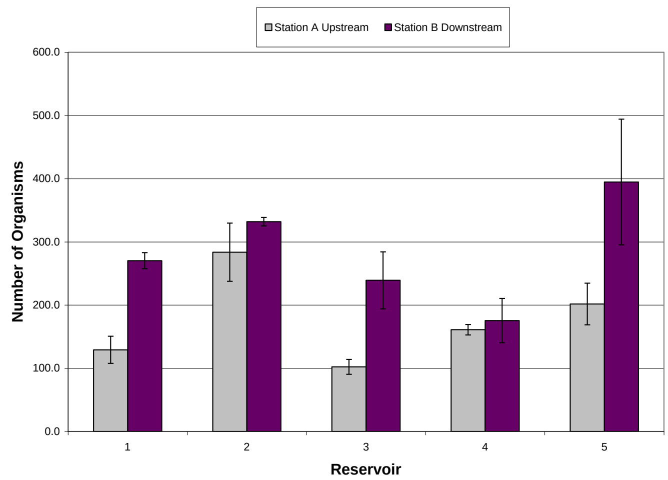

Figure 3.14. Mean (±SEM) abundance of benthic macroinvertebrates: spring.

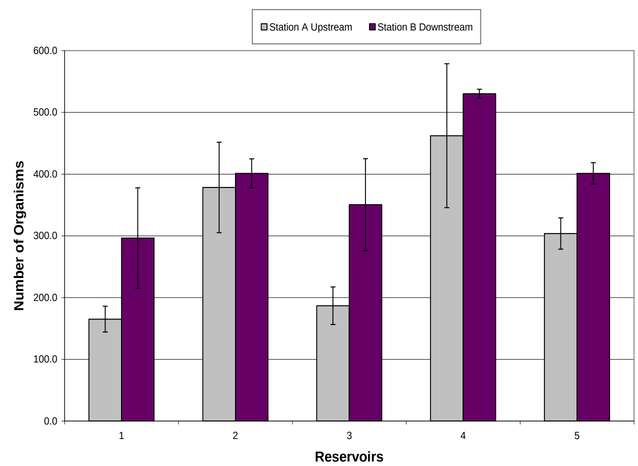

Figure 3.15. Mean (±SEM) abundance of benthic macroinvertebrates: summer.

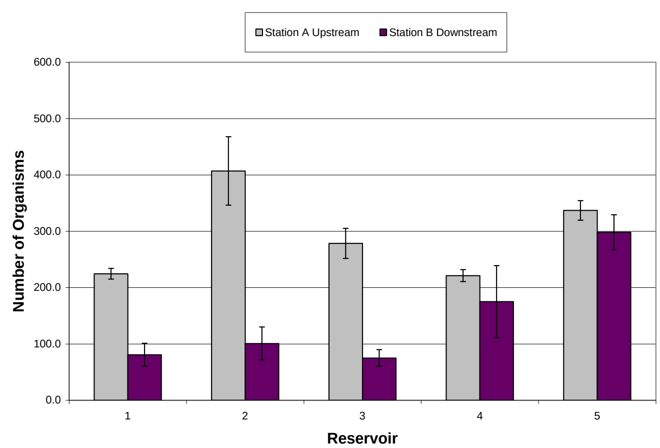

Figure 3.16. Mean (±SEM) abundance of benthic macroinvertebrates: fall.

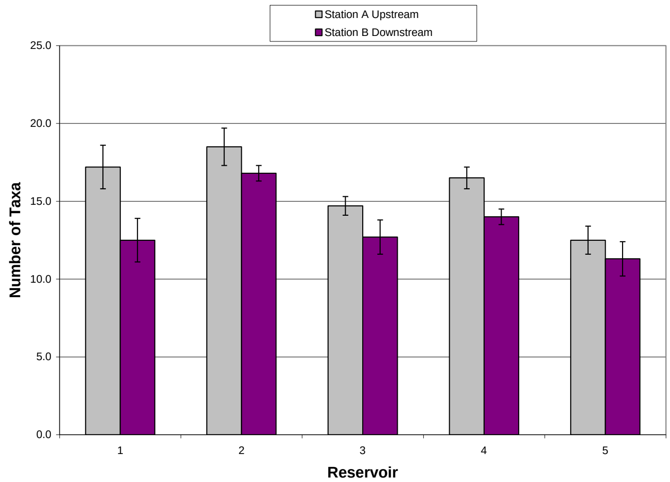

Figure 3.17. Mean (±SEM) benthic macroinvertebrate taxa richness: spring.

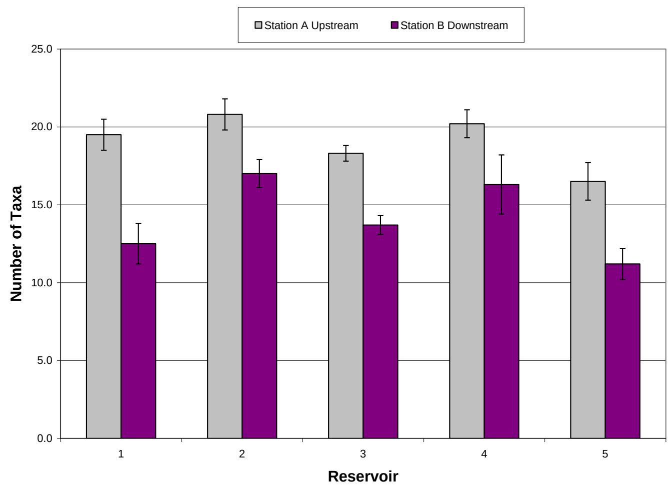

Figure 3.18. Mean (±SEM) benthic macroinvertebrate taxa richness: summer.

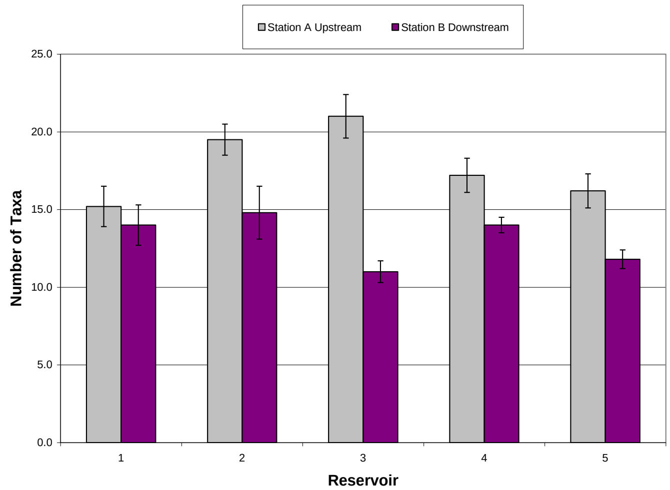

Figure 3.19. Mean (±SEM) benthic macroinvertebrate taxa richness: fall.

3.2.3 Biotic Indices

Average EPT taxa richness (number of individuals) ranged from 1.2 ±0.6 at station 5B in spring to 8.7 ±0.3 at station 3A in fall (Figures 3.20-3.22). In all stations EPT taxa richness was higher at station A in all sampling periods.

Mean values for Hilsenhoff's Biotic Index (HBI) ranged from a minimum of 3.4 ±0.2 at station 3A to a maximum of 6.0 ±0.3 at stations 3B and 5B during fall (Figures 3.23-3.25). Compared with station A, HBI values were moderately higher at station B in all three sampling periods and differences were greatest at Guelph Dam (3) and Laurel Creek Reservoir (5).

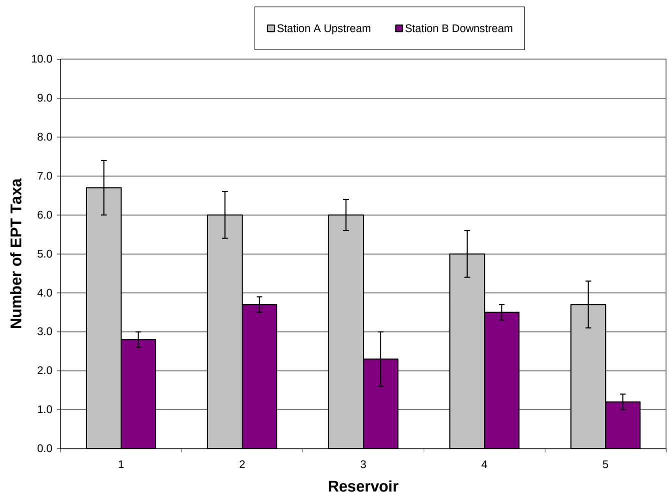

Figure 3.20. Mean (±SEM) EPT taxa richness: spring.

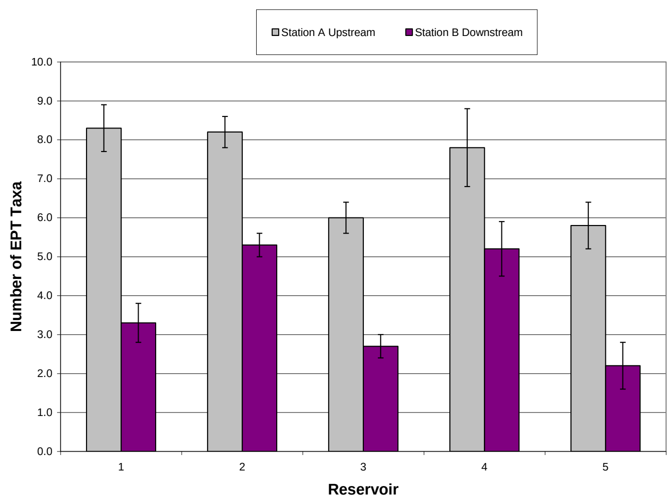

Figure 3.21. Mean (±SEM) EPT taxa richness: summer.

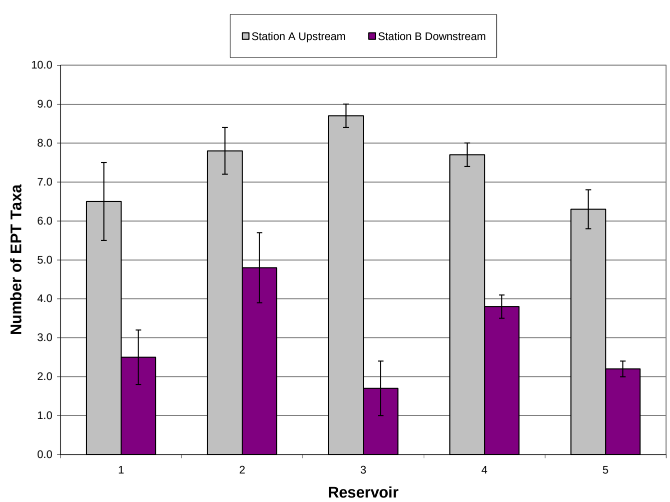

Figure 3.22. Mean (±SEM) EPT taxa richness: fall.

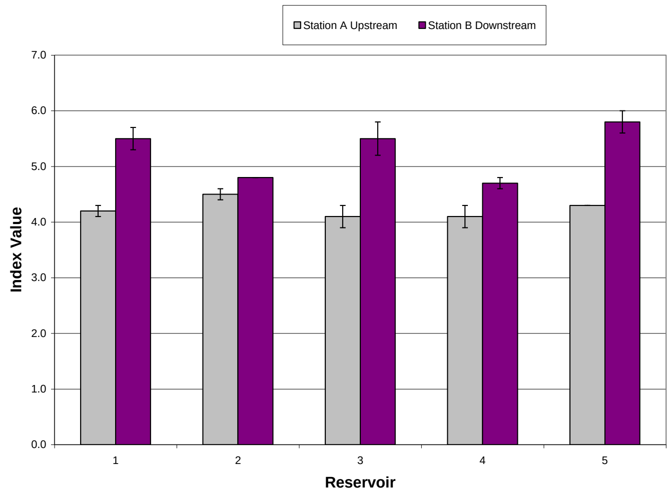

Figure 3.23. Mean (±SEM) values for Hilsenhoff's Biotic Index: spring. (* SEM <0.1)

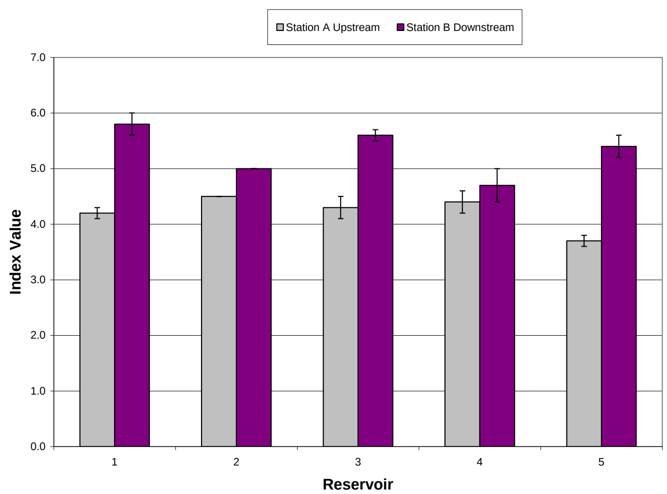

Figure 3.24. Mean (±SEM) values for Hilsenhoff's Biotic Index: summer. (* SEM <0.1)

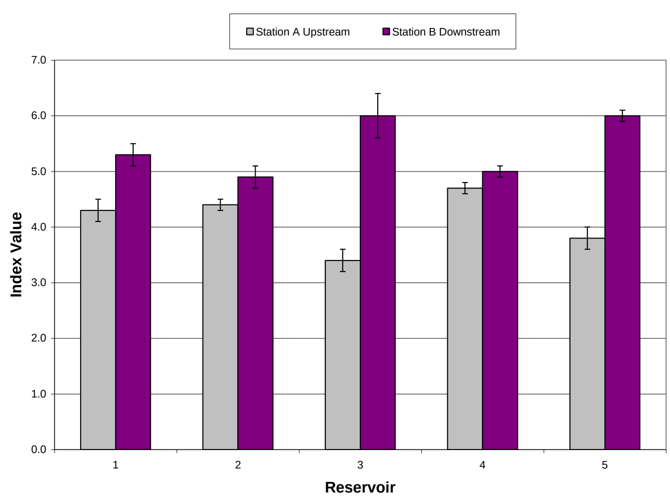

Figure 3.25. Mean (±SEM) values for Hilsenhoff's Biotic Index: fall. (* SEM <0.1)

3.2.4 Relative Abundance Indices

Mean percent dominance of the single most abundant taxon at each station varied from 27.0% ±4.2 at station 1A in spring to 72.4% ±5.1 at station 2A in fall (Figures 3.26-3.28). Mean percent dominance of the most abundant taxon varied between stations and sampling periods. On average, taxon abundance at station B was higher compared to station A during spring and summer, while taxon abundance at station A was higher than station B during fall.

Dominant invertebrate groups were determined based on the highest percentage of taxon at each station (Table 3.4). Dominant taxon invertebrate groups include Isopoda, Ephemeroptera, Trichoptera, Coleoptera, Diptera (non Chironomidae), and Chironomidae, while non dominant taxon invertebrate groups include Nematoda, Dugesia, Hydrachnida, Oligocheata, Mollusca, Amphipoda, Decapoda, Lepidoptera, Odonata, Plecoptera and Megaloptera. Isopoda were abundant downstream at stations 1B, 2B and 3B in summer, and 3B in fall. Ephemeroptera were abundant upstream at stations 1A and 3A in spring and station 3A during summer and fall. Trichoptera were abundant at station 2A and 4B in spring and stations 1A, 2B, 4A and 5A in fall. Coleoptera were abundant at stations 2A and 4B in summer and Diptera were abundant at station 4B during fall. The percent dominance of Chironomidae varied between stations and sampling periods, but on average the taxon were most abundant at station B in all sampling periods.

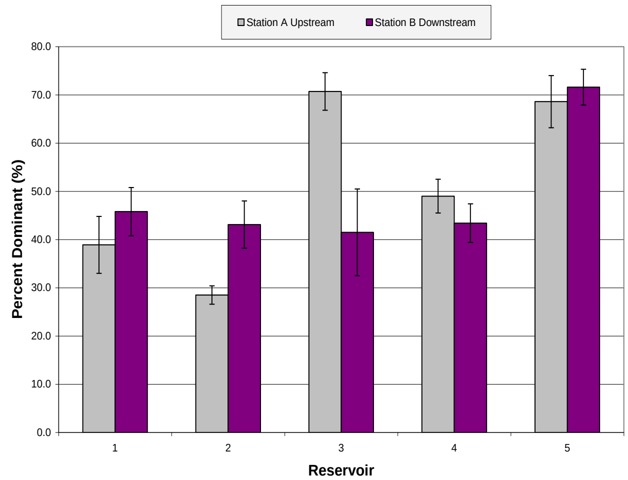

Figure 3.26. Mean (±SEM) relative abundance of the single most abundant taxon: spring.

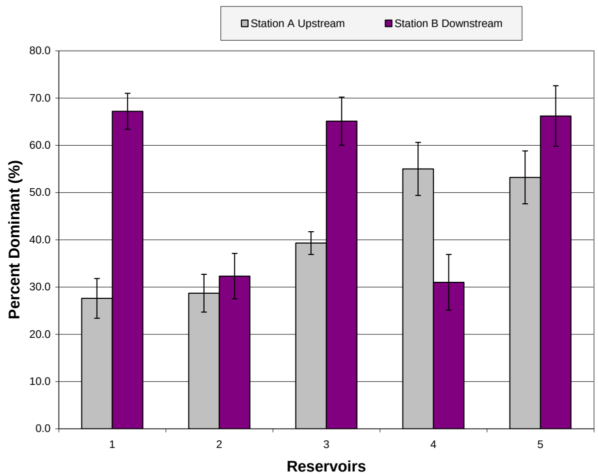

Figure 3.27. Mean (±SEM) relative abundance of the single most abundant taxon: summer.

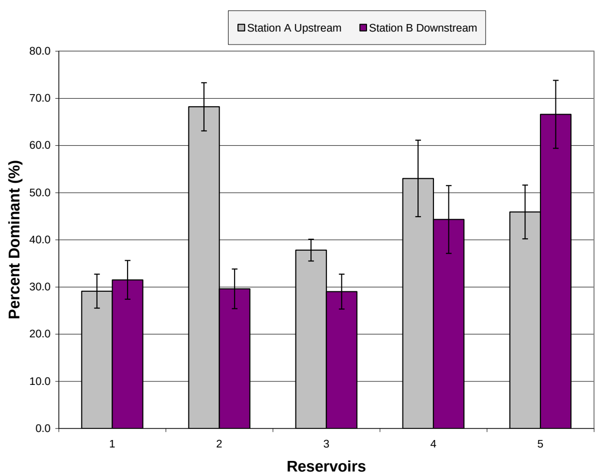

Figure 3.28. Mean (±SEM) relative abundance of the single most abundant taxon: fall.

Table 3.4. Mean percent composition of the benthic macroinvertebrates community.

Reservoir	Sampling Period	Station	Dugesia	Oligocheata	Mollusca	Isopoda	Ephemeroptera	Plecoptera	Trichoptera	Coleoptera	Diptera	Chironomidae	Other*
1 Shand	1	A	0.0	1.5	2.6	0.0	38.9	0.2	11.2	24.1	2.7	18.1	0.7
		B	3.3	5.4	0.0	14.1	0.4	0.0	17.0	0.6	12.5	45.8	0.9
2 Conestogo	1	A	4.0	6.0	0.7	0.9	16.5	0.0	28.5	12.5	1.5	27.7	1.7
		B	3.5	2.1	0.4	20.7	4.8	0.0	11.4	7.0	0.8	43.1	6.2
3 Guelph	1	A	0.3	1.9	0.6	0.1	70.7	0.5	7.5	8.6	0.1	7.8	1.9
		B	1.4	13.4	1.2	33.8	0.1	0.0	4.5	2.0	0.6	41.5	1.5
4 Mills	1	A	0.0	0.9	1.3	0.0	12.0	1.3	4.9	25.6	1.6	49.0	3.4
		B	1.3	0.4	1.7	1.5	2.9	0.0	43.4	20.6	8.1	19.1	1.0
5 Laurel	1	A	0.0	0.1	1.9	0.0	1.1	0.0	17.1	5.7	4.8	68.6	0.7
		B	3.6	1.6	0.4	0.0	0.0	0.0	16.9	0.5	3.8	71.6	1.6
1 Shand	2	A	0.0	5.1	3.2	0.5	21.1	0.9	20.7	19.1	0.2	27.6	1.6
		B	6.9	4.6	0.4	67.2	0.8	0.0	5.7	0.3	0.0	8.9	5.2
2 Conestogo	2	A	3.4	0.2	1.7	0.0	27.2	0.0	18.1	28.7	0.1	17.0	3.6
		B	9.2	7.5	5.3	32.3	2.7	0.0	9.6	12.0	1.5	18.7	1.2
3 Guelph	2	A	1.3	0.5	0.3	0.1	39.3	0.0	5.5	20.2	0.5	27.6	4.7
		B	12.4	7.3	2.3	65.1	0.2	0.0	1.0	2.6	0.8	7.3	1.0
4 Mills	2	A	0.1	2.5	0.7	0.1	13.0	0.1	11.6	11.1	1.3	55.0	4.5
		B	8.3	6.9	7.6	5.2	7.6	0.0	22.3	31.0	0.7	4.1	6.3
5 Laurel	2	A	0.0	0.6	4.0	0.0	3.8	0.0	24.3	6.4	2.8	53.2	4.9
		B	5.4	19.9	1.4	0.0	0.3	0.0	4.1	0.7	0.2	66.2	1.8
1 Shand	3	A	3.6	2.0	3.7	0.0	20.7	1.2	29.1	9.6	0.3	27.2	2.6
		B	4.4	15.7	5.4	22.8	0.6	0.0	7.9	1.7	6.5	31.5	3.5
2 Conestogo	3	A	0.8	0.2	0.9	0.0	11.0	0.6	12.3	4.1	1.0	68.2	0.9
		B	1.4	7.0	2.4	16.0	2.9	0.7	29.6	6.8	5.1	27.7	0.4
3 Guelph	3	A	1.3	1.4	0.7	0.0	37.8	12.3	6.8	13.6	0.8	23.0	2.3
		B	10.2	7.3	12.4	29.0	2.5	0.0	2.9	1.4	1.6	24.8	7.9
4 Mills	3	A	0.0	6.0	1.9	0.0	23.4	1.1	53.0	4.8	1.7	6.4	1.7
		B	1.2	0.0	4.6	1.5	3.9	0.9	9.5	11.5	44.3	17.5	5.1
5 Laurel	3	A	0.0	0.5	9.2	0.0	11.6	0.0	45.9	16.2	3.8	10.4	2.4
		B	1.7	17.2	0.3	0.0	1.0	0.0	5.5	0.0	4.7	66.6	3.0

*Other invertebrate groups include: Nematoda, Hydrachnida, Amphipoda, Decapoda, Lepidoptera, Odonata and Megaloptera

Mean relative abundance (percent) of Ephemeroptera in stations ranged from a low of 0.0% at station 5B to a high of 70.7% ±3.9 at station 3A during spring (Figures 3.29-3.31). Mean abundance (number of individuals) of Ephemeroptera in stations ranged from a low of 0.2 ±0.2 at station 5B during spring to a high of 106.7 ±18.6 during fall (Figures 3.32-3.34). In all stations, mean relative abundance of Ephemeroptera and mean abundance of Ephemeroptera was distinctively higher at station A compared to station B in all sampling periods. Overall mean relative abundance and mean abundance of Ephemeroptera was highest at station A in spring. Station A was by far higher than station B downstream of deep release reservoir, Guelph Dam (3), in all sampling periods.

The mean percent contribution of Isopoda to the benthic macroinvertebrate community ranged from a low of 0.0% at stations 1A, 4A and 5A in spring, stations 2A, 4A and 5A in summer, and all stations A in fall, to a maximum of 69.8% ±3.8 in station 2B in summer (Figures 3.35-3.37). Mean abundance of Isopoda ranged from 0.0 at stations 1A, 4A, 5A and 5B spring, stations 2A, 5A and 5B in summer, and all stations A and station B in fall to a maximum of 239.2 ±74.1 at station 3B in summer (Figures 3.38-3.40). With the exception of surface release reservoirs, Shade's Mills Dam (4) and Laurel Creek Reservoir (5), the percent contribution of Isopoda and the mean abundance of Isopoda in deep release reservoirs, Shand Dam (1), Conestogo Dam (2) and Guelph Dam (3), were notably higher at station B compared to station A in all sampling periods and were most noticeable during summer.

The mean percent contribution of Chironomidae to the benthic macroinvertebrate community varied from 5.8% ±0.6 at station 4A to 72.4% ±5.1 at station 2A during fall (Figures 3.41-3.43). Mean abundance of Chironomidae varied from a low of 8.0 ±1.7 at station 3A during spring to a high of 294.5 ±62.6 at station 2A during fall (Figures 3.44-3.46). Mean percent Chironomidae and mean abundance of Chironomidae varied between stations and sampling periods. During summer, mean percent Chironomidae was higher at station A compared to station B, while during spring and fall, mean percent Chironomidae was higher at station B. Mean abundance of Chironomidae was higher at station A during summer and fall, while mean abundance of Chironomidae was higher at station B during spring.

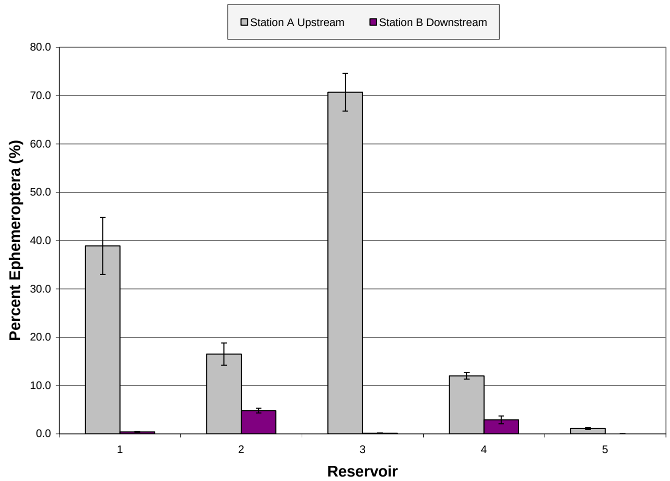

Figure 3.29. Mean (±SEM) relative abundance of Ephemeroptera: spring.

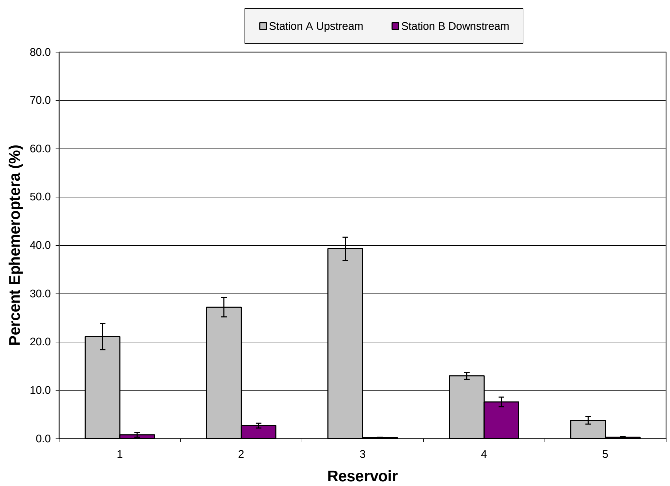

Figure 3.30. Mean (±SEM) relative abundance of Ephemeroptera: summer.

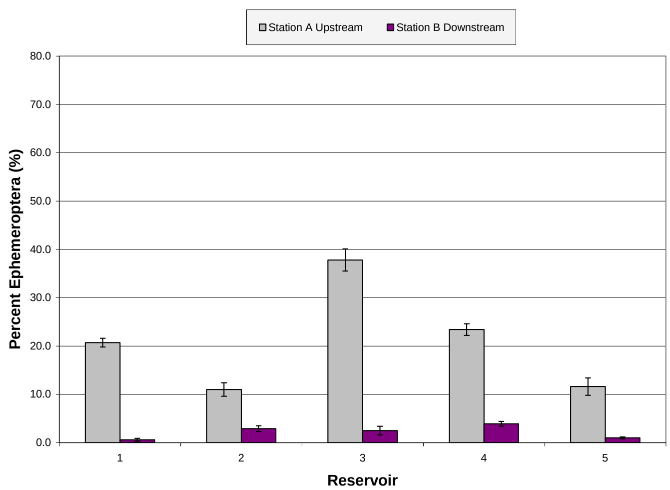

Figure 3.31. Mean (±SEM) relative abundance of Ephemeroptera: fall.

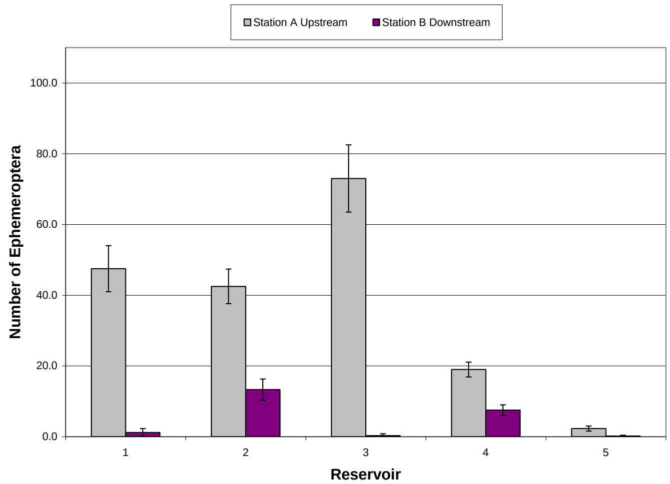

Figure 3.32. Mean (±SEM) abundance of Ephemeroptera: spring.

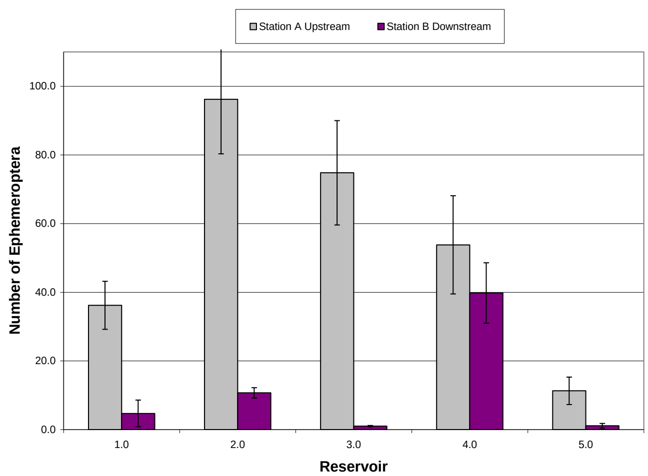

Figure 3.33. Mean (±SEM) abundance of Ephemeroptera: summer.

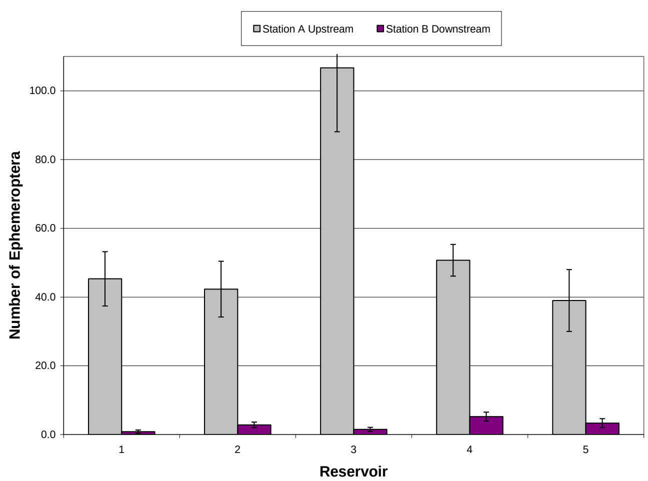

Figure 3.34. Mean (±SEM) abundance of Ephemeroptera: fall.

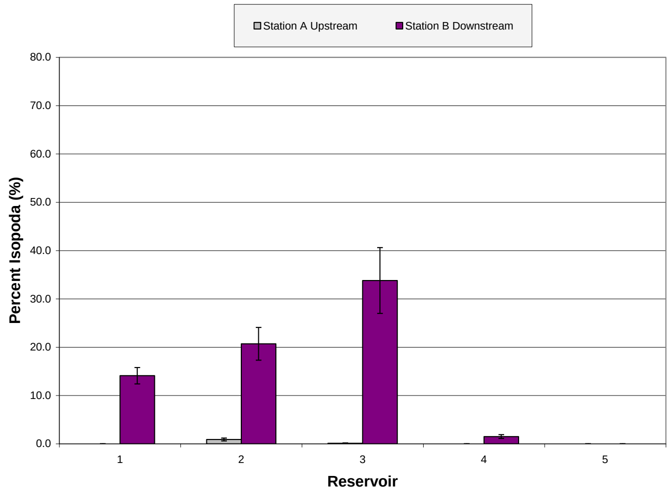

Figure 3.35. Mean (±SEM) relative abundance of Isopoda: spring.

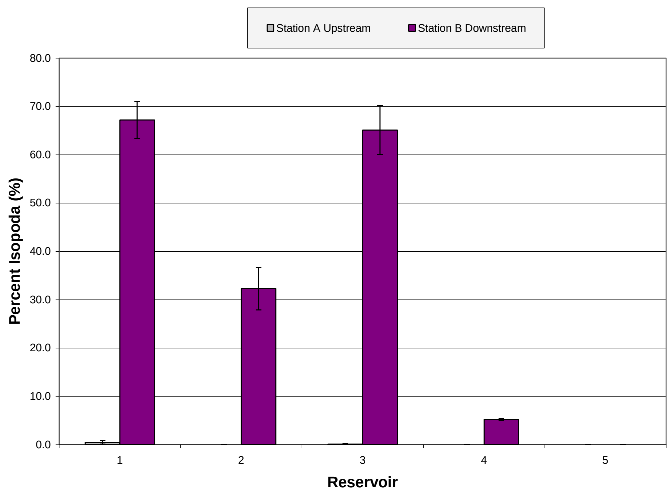

Figure 3.36. Mean (±SEM) relative abundance of Isopoda: summer.

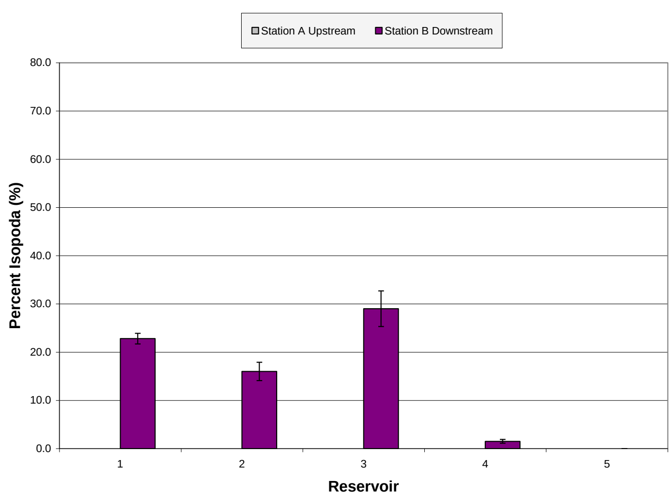

Figure 3.37. Mean (±SEM) relative abundance of Isopoda: fall.

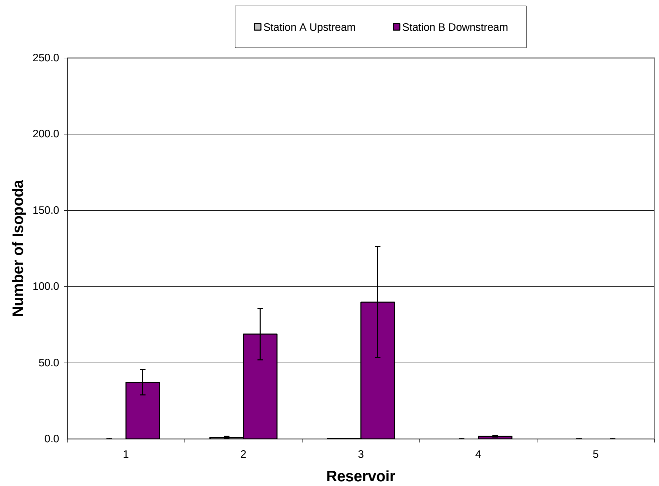

Figure 3.38. Mean (±SEM) abundance of Isopoda: spring.

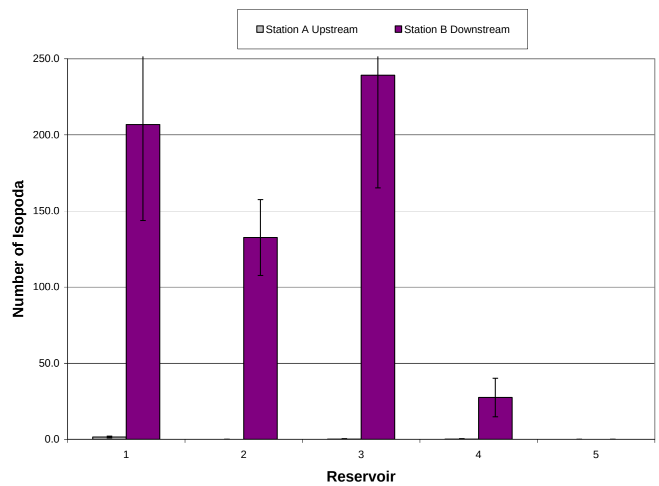

Figure 3.39. Mean (±SEM) abundance of Isopoda: summer.

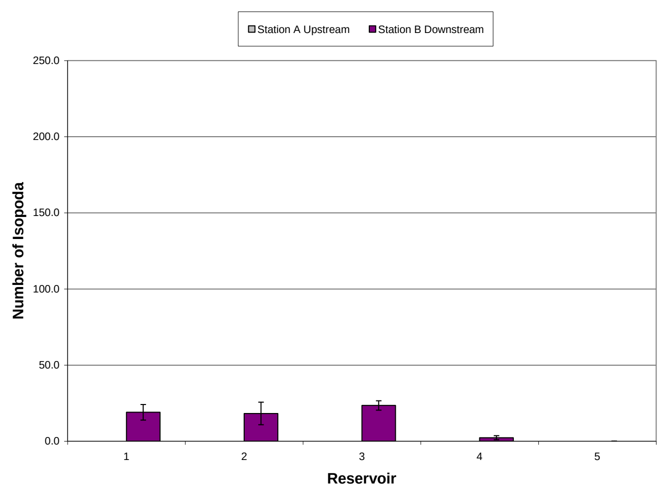

Figure 3.40. Mean (±SEM) abundance of Isopoda: fall.

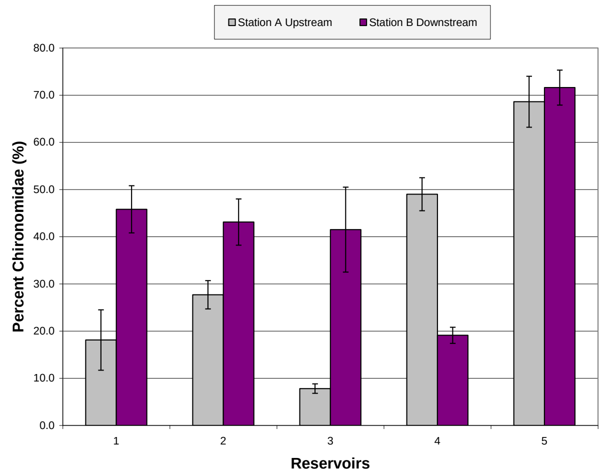

Figure 3.41. Mean (±SEM) relative abundance of Chironomidae: spring.

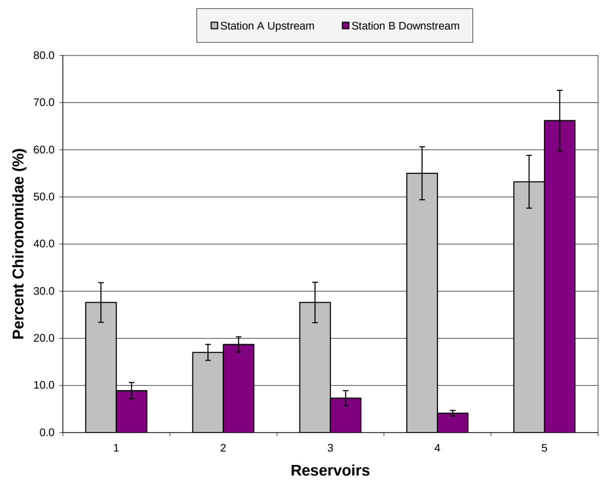

Figure 3.42. Mean (±SEM) relative abundance of Chironomidae: summer.

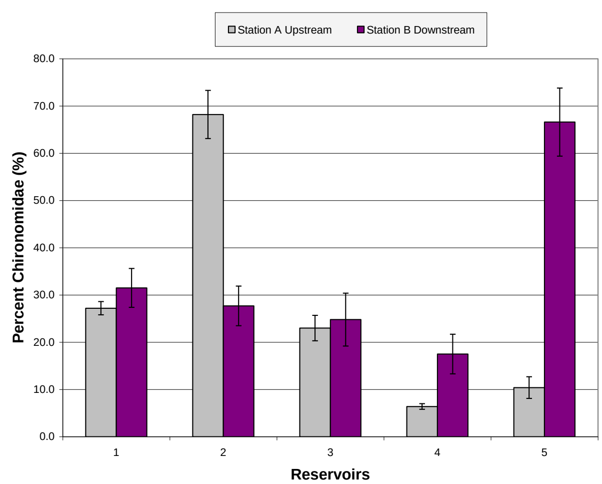

Figure 3.43. Mean (±SEM) relative abundance of Chironomidae: fall.

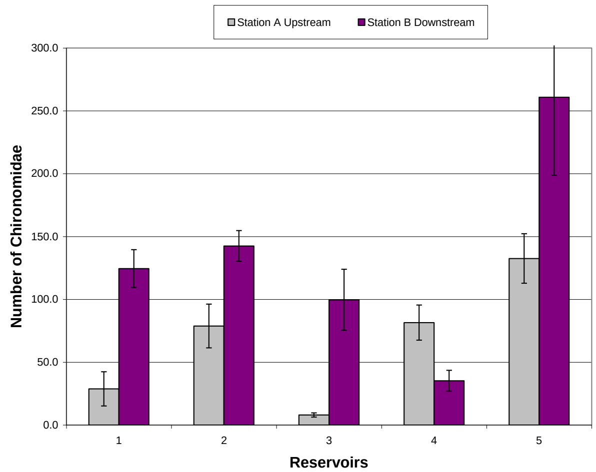

Figure 3.44. Mean (±SEM) abundance of Chironomidae: spring.

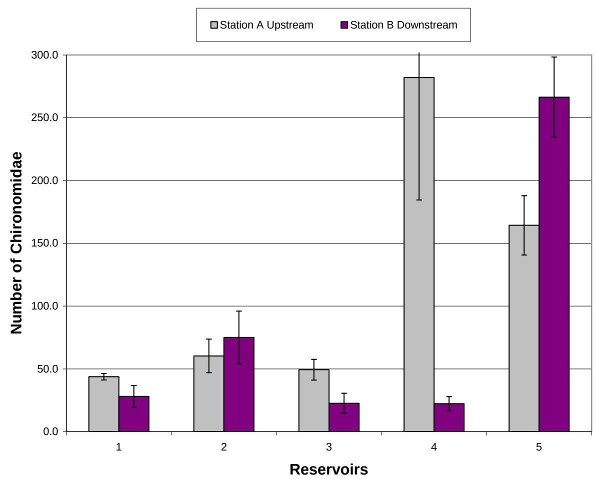

Figure 3.45. Mean (±SEM) abundance of Chironomidae: summer.

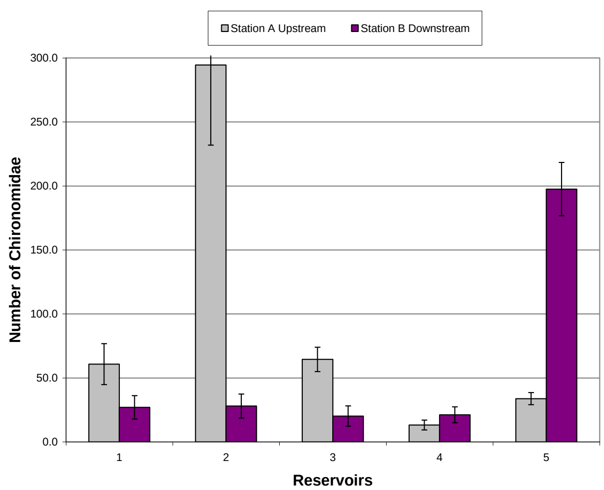

Figure 3.46. Mean (±SEM) abundance of Chironomidae: fall.

3.2.5 Benthic Macroinvertebrate Functional Feeding Groups

A summary of the mean percent composition of benthic macroinvertebrate functional feeding groups is presented in Table 3.5. Mean percent contribution of filter-feeders to benthic macroinvertebrate composition functional feeding groups ranged from a low of 2.5% ±1.2 at station 3B in spring to a high of 58.3% ±11.2 at station 4B in the fall (Figures 3.47-3.49). Mean abundance of filter-feeders ranged from a low of 4.3 ±2.1 at station 3B during spring to a high of 164.7 ±19.0 at station 5A during fall (Figure 3.50-3.52). Mean percent contribution of filter-feeders and mean abundance of filter-feeders varied between stations and sampling periods but on average was higher at station B during spring and fall. Most noticeably, filter-feeder populations downstream of surface release reservoir, Shade's Mills dam (4), were higher at station B in comparison to station A in all sampling periods.

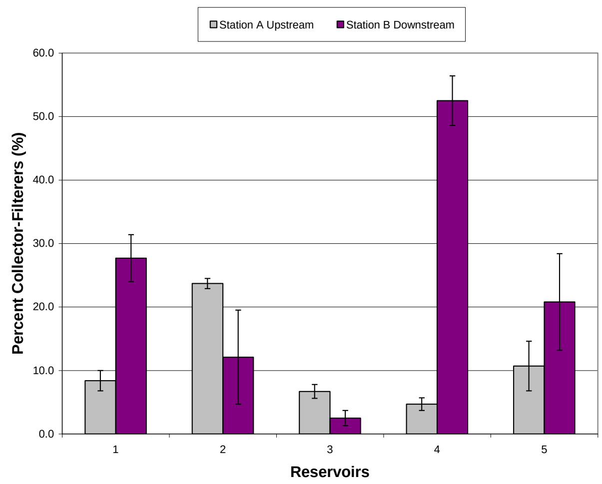

Figure 3.47. Mean (±SEM) relative abundance of filter-feeders: spring.

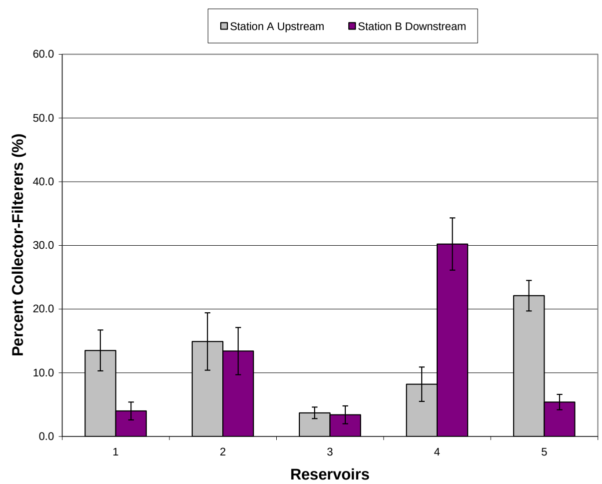

Figure 3.48. Mean (±SEM) relative abundance of filter-feeders: summer.

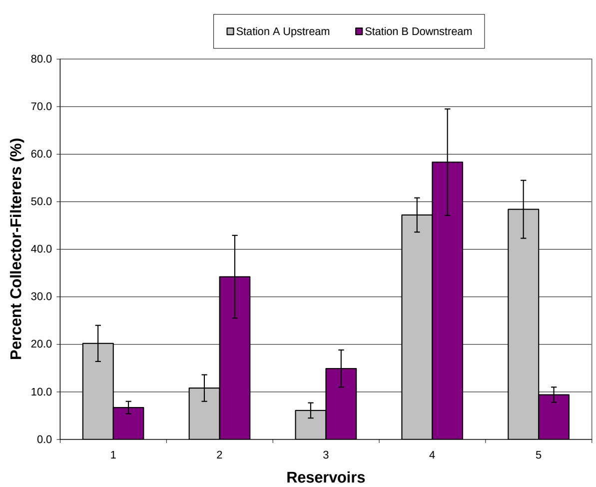

Figure 3.49. Mean (±SEM) relative abundance of filter-feeders: fall.

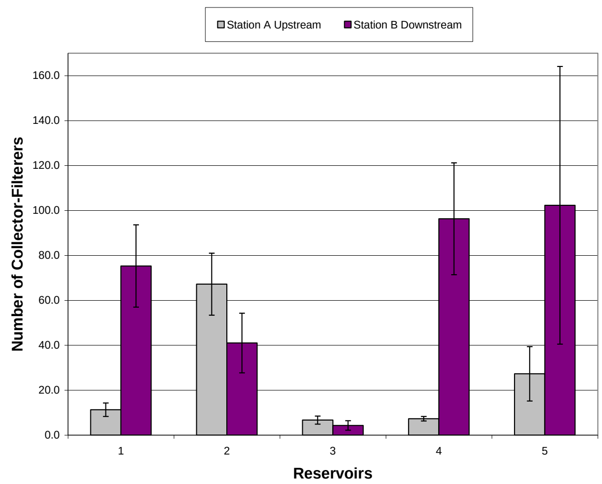

Figure 3.50. Mean (±SEM) abundance of filter-feeders: spring.

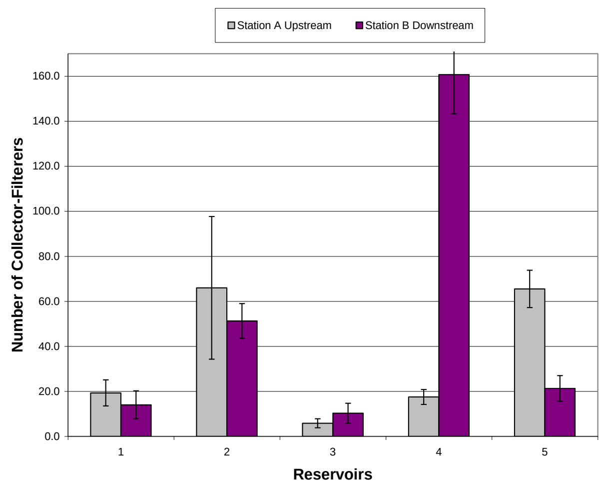

Figure 3.51. Mean (±SEM) abundance of filter-feeders: summer.

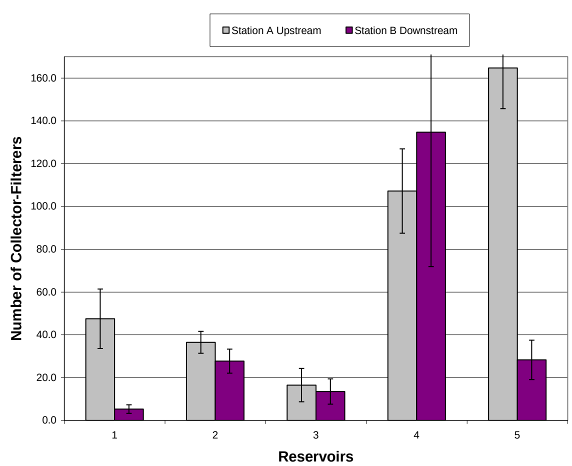

Figure 3.52. Mean (±SEM) abundance of filter-feeders: fall.

Table 3.5. Mean percent composition of benthic macroinvertebrate functional feeding groups.

Reservoir	Sampling Period	Station	Predators	Shredders	Scrapers	Detrivores	Filter-Feeders
1 Shand	1	A	4.0	0.2	30.2	57.2	8.4
		B	6.4	0.4	1.6	63.7	27.9
2 Conestogo	1	A	8.4	0.1	19.4	48.4	23.7
		B	9.2	0.4	7.1	71.2	12.1
3 Guelph	1	A	3.7	0.5	10.4	78.7	6.7
		B	3.3	0.0	5.8	88.4	2.5
4 Shade's Mills	1	A	1.4	0.6	28.0	62.2	4.7
		B	2.3	0.6	26.5	26.4	52.5
5 Laurel Creek	1	A	2.2	0.2	16.6	70.0	10.7
		B	4.4	0.6	0.9	73.3	20.8
1 Shand	2	A	6.6	1.5	28.5	49.9	13.5
		B	7.8	0.2	2.2	85.8	4.0
2 Conestogo	2	A	10.3	3.8	29.8	41.2	14.9
		B	12.2	0.1	14.6	59.7	13.4
3 Guelph	2	A	7.4	0.2	23.7	65.1	3.6
		B	14.5	0.0	2.8	79.3	3.4
4 Shade's Mills	2	A	6.1	0.3	15.6	69.2	8.8
		B	12.2	0.2	25.5	21.8	30.3
5 Laurel Creek	2	A	8.8	1.1	10.8	56.2	23.1
		B	8.8	0.0	0.8	85.0	5.4
1 Shand	3	A	10.4	0.4	22.5	46.2	20.5
		B	8.8	3.5	9.4	71.7	6.6
2 Conestogo	3	A	3.9	0.6	8.3	76.4	10.8
		B	2.8	0.9	9.9	52.1	34.3
3 Guelph	3	A	6.2	12.8	15.3	59.6	6.1
		B	13.8	0.4	3.0	67.9	14.9
4 Shade's Mills	3	A	2.1	2.1	23.7	25.0	47.1
		B	3.7	0.9	12.2	24.9	58.3
5 Laurel Creek	3	A	5.9	2.3	21.0	22.2	48.6
		B	4.1	0.1	0.2	85.5	9.5

Chapter 4: Discussion

4.1 General Responses of Benthic Macroinvertebrate Communities in Downstream Ecosystems of Reservoirs

There is abundant literature on the negative downstream effects of reservoirs on benthic macroinvertebrates in lotic stream ecosystems. Few researchers have examined these effects on benthic macroinvertebrates in the Grand River watershed. In this section, results from the present study are compared with studies that document the downstream effects of reservoirs on benthic macroinvertebrates.

In Table 4.1, abundance and diversity of benthic macroinvertebrates collected in the present study are compared with previous research over the past four decades. Several studies listed in Table 4.1 report a reduction in benthic macroinvertebrate diversity (Paterson and Fernando, 1970; Spence and Hynes, 1971; Ward, 1976; Scullion *et al.*, 1982; Sephton *et al.*, 1983; Munn and Brusven, 1991; Cortes *et al*., 2002; Mwaura *et al*., 2002; Brandimarte *et al*., 2005; Moreno and Callisto, 2006) while overall benthic macroinvertebrate abundance increases downstream of reservoirs (Spence and Hynes, 1971; Ward, 1976; Munn and Brusven, 1991; Al-Lami *et al*., 1998; Richardson *et al*., 2003). Sensitive species of Plecoptera and or Ephemeroptera are often eliminated or reduced (Hilsenhoff, 1971; Spence and Hynes, 1971; Lehmkuhl, 1972; Michaletz *et al*., 2005). However, more tolerant organisms, such as Isopoda and Chironomidae can become more abundant (Hilsenhoff, 1971; Munn and Brusven, 1991; Al-Lami *et al*., 1998; Mwaura *et al*., 2002; Richardson *et al*., 2003; Michaletz *et al*., 2005; Moreno and Callisto, 2006) and ultimately the trophic structure of the community can shift substantially downstream (Ward, 1976; Cortes *et al*., 2002; Vallania, 2007).

Table 4.1. Downstream effects of reservoirs on benthic macroinvertebrates.

Reference	Location	Observed Benthic Macroinvertebrate Changes
Present Study	Five reservoirs, Ontario	Reduced diversity and increased abundance. Post drawdown abundance decreased. EPT taxa reduced and HBI values increased. Ephemeroptera was reduced or absent. Isopoda and Chironomidae were increased. Detrivores feeders increased and scrapers decreased.
Paterson and Fernando, 1970	Laurel Creek Reservoir, Ontario	Reduced diversity of Chironomidae.
Hilsenhoff, 1971	Mill Creek, Wisconsin, USA	Many species eliminated and the fauna became dominated by *Simulium* sp., Chironomidae and *Gammarus* sp.
Spence and Hynes, 1971	Shand Dam, Ontario	Reduced diversity and increased abundance. Plecoptera were absent and Ephemeroptera were reduced.
Lehmkuhl, 1972	S. Saskatchewan River, Gardiner Dam, Canada	Marked reduction of macroinvertebrates downstream for over 100km. 15 species of Ephemeroptera were eliminated.
Ward, 1976	S. Platte River, Cheesman Lake, USA	Reduced diversity but increased standing crop for 32km.
Scullion *et al.*, 1982	River Elan, Craig Goch Reservoir UK	Reduced abundance and diversity.
Sephton *et al.*, 1983	Laurel Creek Reservoir, Ontario	Decreased standing stock and diversity of Chironomidae.
Munn and Brusven, 1991	Clearwater River, Idaho, USA	Found high abundance and low taxa richness. Dominated by Orthoclad Chironomid from 68-99%.
Al-Lami *et al.*, 1998	Radica Lake, Iraq	Total mean density and abundance increased. Benthic community was dominated by Oligochaeta.

Table 4.1 (Continued) Downstream effects of reservoirs on benthic macroinvertebrates.

Reference	Location	Observed Benthic Macroinvertebrate Changes
Ogbeibu and Oribhabor, 2001	Ikpoba River, Nigeria, Africa	Abundance and density of benthic macroinvertebrates were decreased.
Cereghino *et al*., 2002	River Oriege, France	Low abundance.
Cortes *et al*., 2002	Alto Lindoso and Touvedo Dam, Portugal	Decreased diversity of benthic macroinvertebrates.
Mwaura *et al*., 2002	Eight small reservoirs Kenya, Africa	Low diversity and abundance of benthic invertebrates. Dominated by Lumbriculidae and Chironomidae.
Lessard and Hayes, 2003	Ten small dams Michigan, USA	Community composition of benthic macroinvertebrates shifted.
Richardson *et al*., 2003	Peticodiac River New Brunswick	Abundance of resistant species: Chironomidae increased. Macroinvertebrates reduced downstream post drawdown.
Brandimarte *et al*., 2005	Mogi-Guacu River, Brazil	Reduced taxa richness.
Michaletz *et al*., 2005	Thirty impoundments Missouri, USA	Ephemeroptera and Odonata abundance decreased. Diptera abundance increased.
Furey *et al*., 2006	Sooke Lake Reservoir, British Columbia	Biomass of benthic macroinvertebrates decreased post reservoir drawdown.
Moreno and Callisto, 2006	Iberite Reservoir, Brazil	Low values of richness and diversity. High densities of tolerant organisms.
Vallania, 2007	Grande River, Argentina	Filter-feeders, scrapers and predators increased and detrivores and shredders decreased.

4.2 Downstream Effects of Reservoirs on Benthic Macroinvertebrates

General trends of the downstream impacts of reservoirs in the Grand River watershed and studies reported in the literature on benthic macroinvertebrates are discussed by comparing the following indices: abundance (number per sample), taxa richness, EPT taxa richness (number of individuals), Hilsenhoff's Biotic Index (HBI), percent dominance and relative abundance of Ephemeroptera, Isopoda and Chironomidae. Results of statistical analyses are reported and index values obtained in the present study are compared with literature to determine whether impoundments and their management strategies cause significant ($p<0.05$) changes to stream ecosystems.

4.2.1 Abundance and Richness

The downstream effects of reservoirs on benthic macroinvertebrates were evaluated by comparing abundances at stations A (upstream) and stations B (downstream) in two types of reservoirs (surface release and deep release reservoirs) during three sampling periods (spring, summer and fall). Results of repeated measures ANOVA indicated that there was a significant three-way interaction between stations, reservoir types, and sampling periods ($F(1,3) = 18.22$, $p=0.0028$). Further analysis of the data by sampling period showed that fall abundances significantly differed from spring and summer abundances ($F(2,6) = 46.65$, $p=0.0064$).

During spring and summer, the mean abundance was higher downstream of all reservoirs, but the opposite was observed in fall. This suggests that reservoir drawdown imposed downstream impacts on invertebrate abundance. A post hoc test indicated that abundance in stations A differed significantly from stations B in deep release reservoirs (1, 2 and 3) which suggests that during all sampling periods, deep release reservoirs impacted benthic macroinvertebrate abundance more than surface release reservoirs (4 and 5) in the Grand River watershed.

Spence and Hynes (1971) and Ward (1976) have suggested that, in addition to the lack of extreme current fluctuations and more stable conditions, increased abundance of benthic macroinvertebrates downstream of reservoirs is likely attributable to the increase in availability of food and microhabitats. Research on deep release reservoirs has shown that increased detritus availability from reservoir outflow (and enhanced algal growth) often prompted a downstream increase of benthic fauna abundance (Spence and Hynes, 1971; Ward, 1976; Al-Lami *et al.*, 1998).

Many studies report significant changes to benthic macroinvertebrate abundance below deep release reservoirs but fewer studies have been recorded on benthic macroinvertebrate abundance below surface release reservoirs. Lessard and Hayes (2003) suggested that increased abundance downstream of surface release reservoirs may result from temperature increases and enhanced algal growth. While further research is needed to examine the downstream abundances of benthic macroinvertebrates below surface release reservoirs, the increased temperatures recorded and the observed algae populations below surface release reservoirs, Shade's Mills Dam (4) and Laurel Creek Reservoir (5), in the Grand River watershed may account for the increased abundance of macroinvertebrates collected during spring and summer.

The downstream effects of reservoirs on benthic macroinvertebrates were also evaluated, in addition to abundance measurements, by taxa richness. A repeated measures ANOVA revealed that stations B differed significantly ($F(1, 3) = 56.44$, $p=0.0049$) from stations A for taxa richness in all sampling periods. The reduction in taxa richness observed indicate that habitat changes occurred downstream from the dams. The Grand River data show that both deep release and surface release reservoirs created low diversity downstream. There were no significant effects of reservoir type but there was an effect of sampling period season ($F(2,6) = 7.78$, $p=0.0216$). A post hoc test showed that spring taxa richness significantly differed from summer taxa richness.

Several authors report low taxa richness downstream of reservoirs attributing to downstream physical, chemical and biological alterations (Ward, 1976; Scullion *et al.*, 1982; Sephton *et al.*, 1983; Munn and Brusven, 1991). Physical changes include flow management and substrate modification (Scullion *et al.*, 1982; Sephton *et al.*, 1983; Munn and Brusven, 1991), while chemical changes include eutrophic conditions, low levels of dissolved oxygen, and temperature modifications (Ward, 1976; Munn and Brusven, 1991; Sephton *et al.*, 1983). In addition, biological changes include altered food supplies, trophic relationships and enhanced competitive interactions (Ward, 1976; Sephton *et al.*, 1983). All physical, chemical and biological changes may have reduced benthic fauna diversity downstream of deep release and surface release reservoirs in the Grand River watershed.

4.2.2 Biotic Indices

The Ephemeroptera, Plecoptera and Trichoptera (EPT) taxa richness index was used to evaluate the downstream effects of reservoirs on sensitive benthic macroinvertebrate taxa. A repeated measures ANOVA indicated that stations A and B significantly differed ($F(1,3) = 24.81$, $p=0.0156$) for both reservoir types during all sampling periods. This suggests that considerable changes in the benthic community occurred downstream. Both deep release and surface release reservoirs produced significant changes downstream for statistical examination showed no effect of reservoir type, however, there was a main effect for sampling period ($F(2,6) = 13.11$, $p=0.0065$). Post hoc showed that spring EPT taxa richness differed significantly from summer and fall EPT taxa richness.

The EPT taxa is a variation of the taxa richness index. Lenat (1988) determined that invertebrate taxa in the three EPT orders tend to be sensitive toward habitat disturbances and changes in water quality. Therefore, increased values of EPT taxa are expected in more natural and pristine conditions. Several authors have documented the elimination or reduction of EPT taxa below dams (Hilsenhoff, 1971; Spence and Hynes, 1971; Lehmkuhl, 1972; Ward, 1976; Scullion *et al.*, 1982; Munn and Brusven, 1991; Cortes *et al.*, 2002; Moreno and Callisto, 2006). Spence and Hynes (1971) documented that the elimination or reduction of EPT taxa below Shand Dam is evidence that reservoirs generate vast changes in the downstream community structure of benthic macroinvertebrate communities. Researchers have often linked this downstream decline of EPT taxa to changes in habitat diversity, fluctuating water levels, altered thermal regimes and altered food supplies (Hilsenhoff, 1971; Lehmkuhl, 1972; Ward, 1976; Munn and Brusven, 1991; Moreno and Callisto).

The significant change in the composition of benthic fauna and reduction of EPT taxa below reservoirs in the Grand River watershed suggest that habitat and trophic structure alterations occurred. This was most noticeable in deep release reservoirs where EPT taxa in reference conditions (Station A) were up to five times greater than downstream habitats (Station B). During fall, deep release reservoir, Guelph Dam (3), decreased downstream by 7 EPT taxa. In all sampling periods station B on average had 4 fewer EPT taxa than station A.

The Hilsenhoff's Biotic Index (HBI) is designed to detect organic enrichment (generally used in urban streams) and values of benthic macroinvertebrates are based on a scale from 1 to 10 with higher values corresponding to pollution and organic enrichment (Hilsenhoff, 1987; 1988). Spence and Hynes (1971) and Ward (1976) have suggested that benthic communities respond similarly downstream of reservoirs to mild organic pollution. Therefore, the HBI index was used to test downstream changes in benthic macroinvertebrate communities and to detect organic enrichment downstream of Grand River watershed reservoirs. Results of a repeated measures ANOVA indicated stations A and B significantly differed ($F(1,3) = 10.18$, $p=0.0497$).

The HBI values between stations moderately increased downstream and were greatest below deep release reservoirs, Shand Dam (1) and Guelph Dam (3), and surface release reservoir, Laurel Creek (5). Shantz *et al.*, (2004) found that concentrations of total phosphorous (TP) and total organic carbon (TOC) were high, in comparison to water quality standards, downstream of Laurel Creek Reservoir and Spence and Hynes (1971) also documented organic enrichment downstream of Shand Dam. The literature, in addition to higher HBI values, suggest that organic enrichment occurred downstream of Grand River reservoirs in the present study.

4.2.3 Relative Abundance Indices

Results of repeated measures ANOVA revealed that there was no significant difference in the relative abundance of the single most abundant taxon between stations, reservoir types and sampling periods. While there was no significance between the relative abundance of the single most abundant taxon, changes in the composition of benthic macroinvertebrates were evident (Table 3.3). The composition of benthic macroinvertebrates downstream of deep release reservoirs were impacted more than the composition of benthic macroinvertebrates downstream of surface release reservoirs. Reference conditions (Station A) were dominated by sensitive species of Ephemeroptera and or Trichoptera, while tolerant organisms in downstream environments (Station B) were abundant, including Chironomidae and Isopoda. This suggests that upstream habitats and water quality were more favorable to invertebrates.

Cortes *et al*. (2002) determined that the compound impacts of eutrophication and artificially low flow led to the colonization downstream of more tolerant species resistant to oxygen depletion and altered nutrient cycles. In addition, Cortes *et al*. (2002) determined that the downstream reduction of species richness, including the reduction of sensitive species of Ephemeroptera, Plecoptera and Trichoptera, in disturbed environments is often linked to the replacement of abundant species more tolerant to perturbations. In the present study, the observed changes in benthic macroinvertebrate composition downstream of deep release reservoirs in the Grand River watershed provide evidence that environmental degradation occurred.

A repeated measures ANOVA indicated that for the relative abundance of Ephemeroptera, stations A and B were significantly different ($F(1,3) = 16.26$, $p = 0.0274$). In all sampling periods, all reservoirs explicitly reduced (or eliminated) the abundance of Ephemeroptera downstream. Specifically, the relative abundance of Ephemeroptera decreased from 70.7% upstream to 0.1% downstream of deep release reservoir, Guelph Dam (3). Mean abundance of Ephemeroptera (number of individuals) at Guelph Dam (3) was reduced from 73.0 upstream to 0.3 downstream during spring, from 74.8 upstream to 1.0 downstream during summer and from 106.7 upstream to 1.5 downstream during fall. During all sampling periods, similar reductions and or eliminations of this order occurred downstream of the other deep release reservoirs, Shand Dam (1) and Conestogo Dam (2) (Table 3.3). This considerable reduction of Ephemeroptera in downstream reaches shows that reservoirs, and in particular deep release reservoirs, have measurable impacts on the diversity of downstream benthic macroinvertebrate communities.

Several authors have reported large reductions of Ephemeroptera downstream of reservoirs (Hilsenhoff, 1971; Lehmkuhl, 1972; Ward, 1976; Michaletz *et al.*, 2005). The observed reductions have been attributed to increased siltation, lower water temperatures and altered water quality. Lehmkuhl (1972) concluded that cooler water temperatures below deep release reservoirs were responsible for the elimination of Ephemeroptera downstream. The literature therefore suggests that reservoirs, and most notably deep release reservoirs, alter the downstream abundance and diversity of Ephemeroptera.

Results of repeated measures ANOVA showed that there was a significant difference between stations A and B ($F(1,3) = 30.74$, $p=0.0116$) for the relative abundance of Isopoda. Isopoda were abundant downstream of deep release reservoirs and during summer, Isopoda were in excess of 65% of the benthic community composition. Mean abundance of Isopoda (number of individuals) at Shand Dam (1) was increased from 0.0 upstream to 37.2 downstream during spring, from 1.5 upstream to 206.8 downstream during summer and from 0.0 upstream to 19.0 downstream during fall. Mean abundance of Isopoda at Conestogo Dam (2) was increased from 1.0 upstream to 68.8 downstream during spring, from 0.0 upstream to 132.5 downstream during summer and from 0.0 upstream to 18.2 downstream during fall. Mean abundance of Isopoda at Guelph Dam (3) was increased from 0.2 upstream to 89.8 downstream during spring, from 0.2 upstream to 239.2 downstream during summer and from 0.0 upstream to 23.5 downstream during fall. Isopoda, *Caecidotea intermedius*, though dominant downstream of deep release reservoirs, occurred only sporadically upstream of the dams and were relatively non-existent above and below surface release reservoirs.

The downstream abundance of the detrivore, *C. intermedius*, below deep release reservoirs has been attributed to the increase of microhabitats and food amongst algal growths and enriched detritus content (Spence and Hynes, 1971). *C. intermedius* is a cold stenothermal species (Bousfield, 1958) that can withstand the cooler temperatures caused by hypolimnetic release unlike other arthropods that favor warmer temperatures during spring and summer seasons. The downstream temperatures below deep reservoirs did not exceed 23^{o}C in spring and summer, while upstream temperatures exceeded 27^{o}C. In addition, Isopoda were abundant during fall with temperatures below 11^{o}C. This suggests that below deep release reservoirs the low summer and higher winter temperatures may have promoted the abundance of Isopoda in addition to increased availability of algal and organic matter content.

A repeated measures ANOVA indicated that there was a significant two-way interaction between reservoir types and sampling periods ($F(1,3) = 28.08$, $p=0.0009$) for the relative abundance of Chironomidae. Further analysis by sampling period showed no significant difference. Results of repeated measures ANOVA also indicated that there was a significant two-way interaction between sampling periods and reservoir stations ($F(2,6) = 5.69$, $p=0.0412$). Results show that while surface release reservoirs did not significantly differ in the relative abundance of Chironomidae, deep release reservoirs was significantly different by sampling period ($F(2,6) =20.98$, $p=0.0076$). A post hoc showed that in deep release reservoirs, spring and fall did not significantly differ but the relative abundances of Chironomidae were significantly different in summer. During spring and fall, the relative abundances of Chironomidae increased downstream while during summer Chironomidae decreased.

Several authors reported increases in Chironomidae abundance downstream of reservoirs (Paterson and Fernando, 1970; Spence and Hynes, 1971; Ward, 1976; Sephton *et al.*, 1983; Munn and Brusven, 1991; Al-Lami *et al.*, 1998; Richardson *et al.*, 2003; Moreno and Callisto, 2006). These studies suggest that chironomids are more tolerant and resistant to considerable environmental degradation (Moreno and Callisto, 2006). Hilsenhoff (1971) found that Chironomidae are often abundant in reaches downstream of reservoirs. Sephton *et al.*, (1983) and Munn and Brusven (1991) documented that the changes in chironomid populations downstream of reservoirs were associated with changes in the trophic structure of the substrate and eutrophific conditions. In relation to the present study, observed increases in Chironomidae abundance downstream of Grand River watershed reservoirs provides evidence that environmental degradation occurred. During fall after reservoir drawdown, Chironomidae were abundant in downstream reaches and specifically comprised more than 65% of the benthic macroinvertebrate community downstream of surface release reservoir, Laurel Creek (5). Mean abundance of Chironomidae (number of individuals) increased from 33.8 upstream to 197.5 downstream of Laurel Creek Reservoir (5).

4.2.4 Feeding Measures

Results of repeated measures ANOVA revealed that there were no significant differences in filter-feeder invertebrates between stations, reservoir types, and sampling periods. This was represented by the variable patterns of filter-feeding arthropods recorded above and below reservoirs during seasonal sampling. On average, filter-feeder abundances decreased downstream of Grand River watershed reservoirs, with the exception of surface release reservoir, Shade’s Mills Dam (4). Mean abundances of filter-feeders (number of individuals) at Shade’s Mills Dam (4) increased from 7.3 upstream to 96.3 downstream during spring, from 17.5 upstream to 160.7 during summer and from 107.2 upstream to 134.7 downstream during fall.

Ward (1984) suggested the downstream increase of plankton from an upstream reservoir outflow may account for the observed dense population of filter-feeding insects. In relation to the present study, this may have prompted the increased abundance of filter-feeders downstream of Shade's Mills Dam (4), however, it does not explain patterns in benthic macroinvertebrate functional feeding groups downstream of deep release reservoirs (1, 2 and 3) and surface release reservoir, Laurel Creek (5). This suggests that other functional feeding groups below reservoirs including predators, shredders, scrapers and detrivores must be examined (Figures 4.1-4.3).

Ward (1976) determined that the downstream effect of reservoirs may be detrimental to benthic macroinvertebrate functional feeding groups. Several authors have observed changes in benthic macroinvertebrate functional feeding groups, of which shredders and scrapers are most adversely impacted due to downstream changes in habitat diversity and water quality (Ward and Stanford, 1984; Camargo and Garcia de Jalón, 1990; Cortes *et al.*, 1998), while Vallania *et al*., (2007) found differences at the level of detrivores feeding invertebrates coinciding to downstream changes in organic matter content. In relation to the present study in the Grand River watershed, vast changes in benthic macroinvertebrate functional feeding groups were observed (Figures 3.47-3.52; Table 3.5). During all sampling periods downstream of Grand River watershed reservoirs scrapers were eliminated or reduced and detrivores feeders increased downstream. Predators increased during spring and summer and decreased during fall, while shredders increased during spring and decreased during summer and fall.

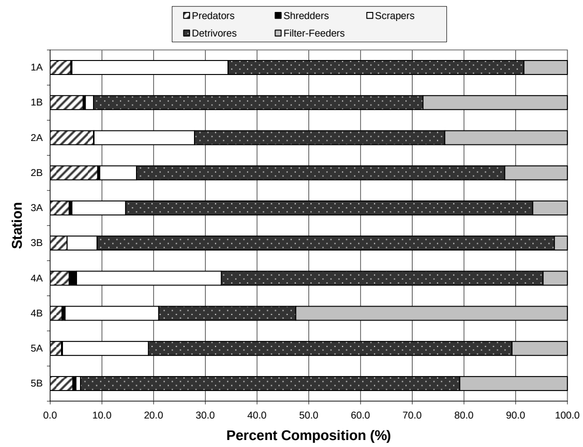

Figure 4.1. Benthic macroinvertebrate feeding measures: spring.

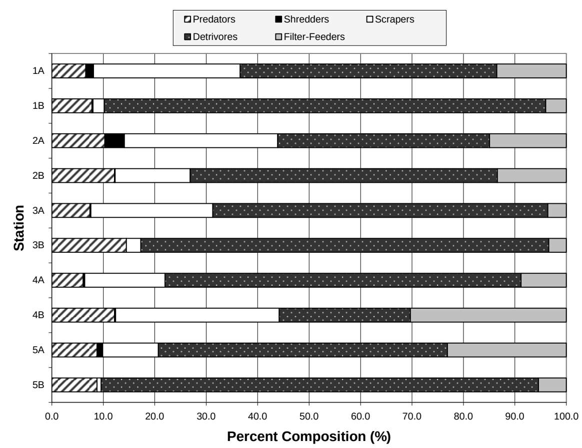

Figure 4.2. Benthic macroinvertebrate feeding measures: summer.

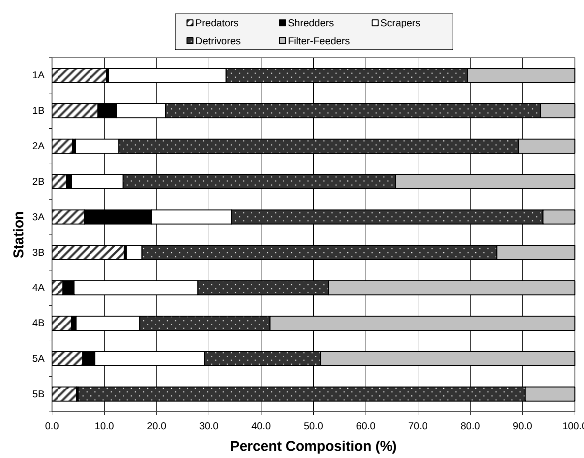

Figure 4.3. Benthic macroinvertebrate feeding measures: fall.

Spence and Hynes (1971) reveal that changes in benthic macroinvertebrate functional feeding groups downstream of reservoirs is a major indicator of the effects of deep release reservoirs on receiving stream ecosystems. Consequently, in relation to the present study, changes were more evident below deep release reservoirs than surface release reservoirs and sensitive benthic macroinvertebrates, including scrapers and shredders, were on average greatly reduced or eliminated. Following reservoir drawdown, all functional feeding groups were drastically reduced downstream, with the exception of detrivores, which dominated benthic macroinvertebrate populations downstream.

4.3 Factors Influencing Benthic Macroinvertebrate Communities Resulting From Impoundments

Results from the present study and a review of literature indicate that reservoirs have a negative effect on benthic macroinvertebrate communities in downstream lotic ecosystems. By isolating environmental impairments created by dams, which have influenced changes in the benthic macroinvertebrate community composition, inferences about causation can be made. Therefore, the following examines changes in flow, temperature, water quality, substrate and vegetation, resulting from the impoundment. In addition, the effects of reservoir management strategy, reservoir drawdown, for both deep release and surface release reservoirs is also examined.

4.3.1 Flow Management

Numerous studies have documented the negative impacts of flow management on stream ecosystems and benthic macroinvertebrate populations. As discussed in detail in Chapter 1, changes in flow downstream of reservoirs often result in a decrease in benthic faunal diversity but an increase in abundance. From this study in the Grand River watershed, the long periods of constant flow downstream of reservoirs created more stability in stream environments, which may have prompted the increase in abundance of fewer species. Meanwhile, the fluctuation in flow experienced in downstream reaches during reservoir drawdown may be responsible for changes in benthic macroinvertebrate abundance and diversity.

Ward (1976) determined that the variations in flow and associated parameters in unregulated streams may alternately favor different benthic macroinvertebrate species. Advantages may be derived from a more constant flow regime but only if a relatively natural seasonal flow pattern is maintained (Ward, 1976). For example, a constant flow regime may not provide the proper migration signals for some species of benthic invertebrates. Ward (1984) documented that the annual migration of some species of mayflies, which is initiated by rising water during spring runoff, will be restricted under a constant flow regime.

4.3.2 Water Quality

Changes in water quality from an upstream impoundment can alter the abundance and diversity of downstream benthic macroinvertebrate communities. Moreno and Callisto (2006) determined that the poor water quality and rapid eutrophication below an impoundment led to the degradation of the benthic macroinvertebrate community with low values of taxa richness and diversity and high abundances of tolerant organisms. Therefore, in relation to the present study, water quality changes observed below Grand River reservoirs may have been responsible for changes in benthic macroinvertebrate composition. While chemical analyses of the stream water and sediment were not conducted in the present study, water quality parameters: pH (Figures 4.4-4.6), dissolved oxygen (DO) (mgL^{-1}) (Figures 4.7-4.9) and conductivity (µS) (Figures 4.10-4.12) and total dissolved solids (TDS) (mgL^{-1}) (Figures 4.13-4.15) were analyzed and general inferences about their effect on benthic macroinvertebrate abundance and diversity can be made.

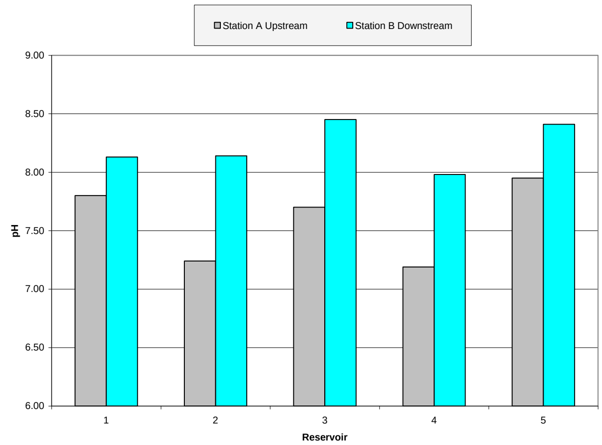

Figure 4.4. pH: spring.

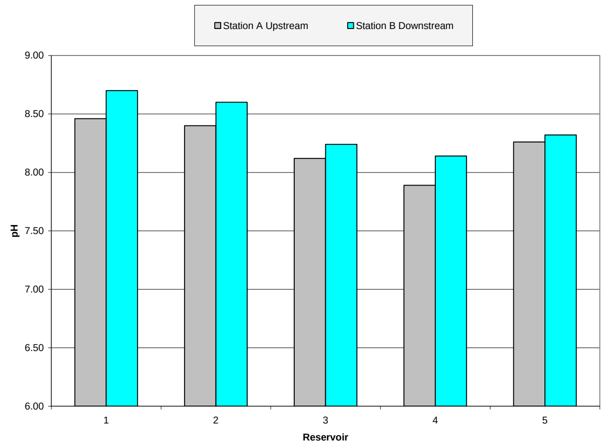

Figure 4.5. pH: summer.

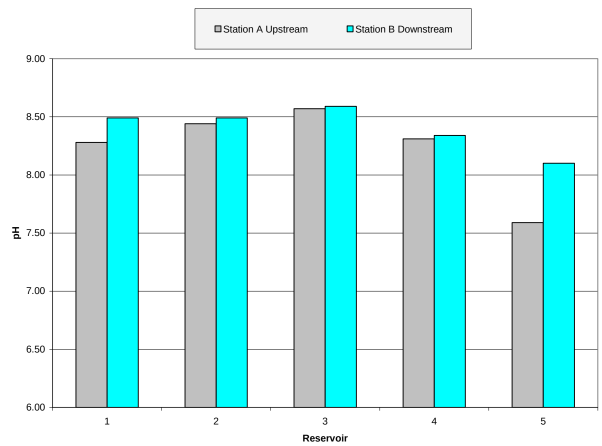

Figure 4.6. pH: fall.

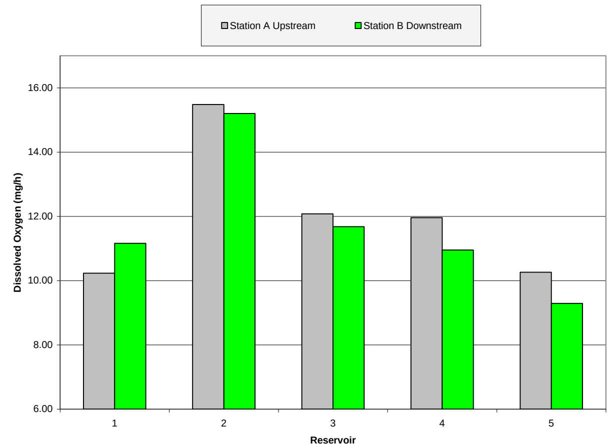

Figure 4.7. Dissolved oxygen (mgL^{-1}): spring.

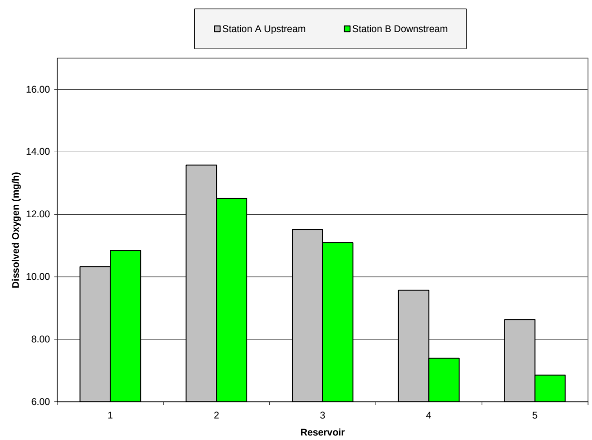

Figure 4.8. Dissolved oxygen (mgL^{-1}): summer.

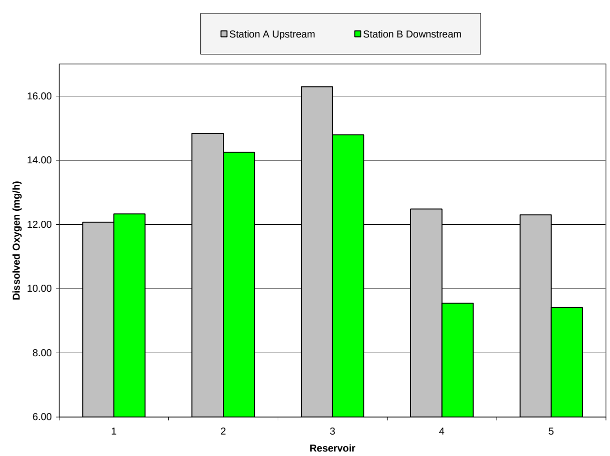

Figure 4.9. Dissolved oxygen (mgL^{-1}): fall.

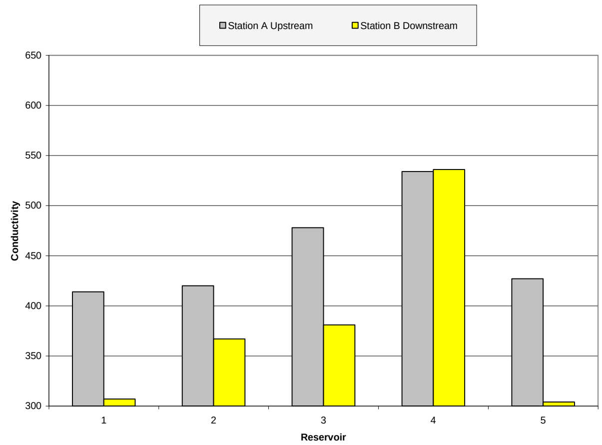

Figure 4.10. Conductivity (μS): spring.

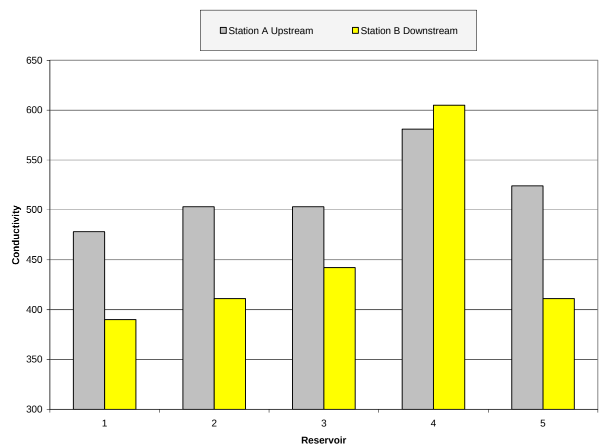

Figure 4.11. Conductivity (μS): summer.

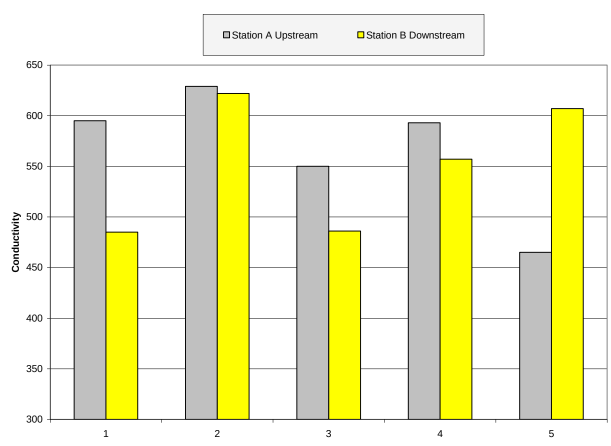

Figure 4.12. Conductivity (μS): fall.

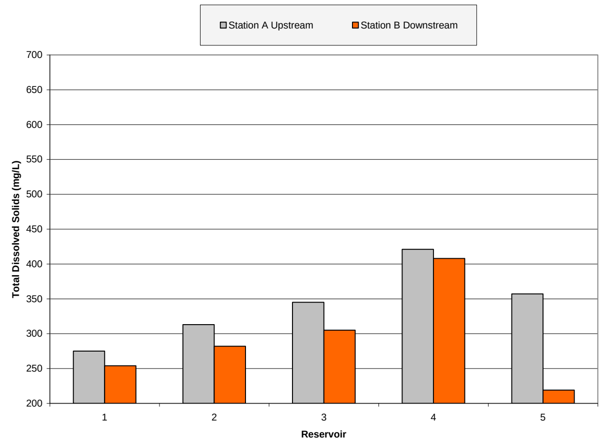

Figure 4.13. Total dissolved solids (mgL^{-1}): spring.

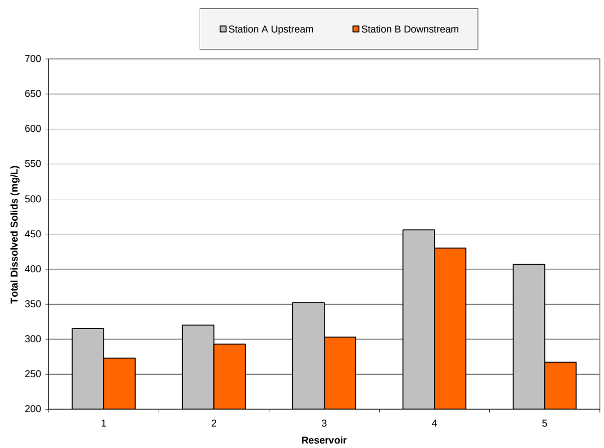

Figure 4.14. Total dissolved solids (mgL^{-1}): summer.

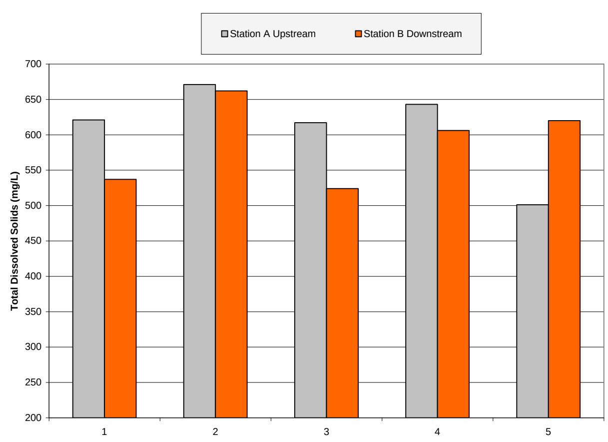

Figure 4.15. Total dissolved solids (mgL^{-1}): fall.

Changes in pH, DO, conductivity, and TDS were evident in downstream Grand River reaches during each of the three sampling periods (Table 3.2). Changes in pH were most noticeable during spring when differences downstream were considerably greater. During summer and fall, increased pH values recorded downstream of Grand River reservoirs were consistent but of lesser magnitude. Changes in DO were most notable during summer and fall and specifically were lower downstream of surface release reservoirs. DO levels in deep release reservoirs were on average slightly lower during all three sampling periods with the exception of Shand Dam (1) in which DO levels were increased. During spring and summer, conductivity measurements were considerably lower downstream, with the exception of Shade's Mills Dam (4) where conductivity moderately increased. Following reservoir drawdown, on average conductivity measurements during fall were higher in comparison to spring and summer and overall, conductivity measurements decreased downstream. TDS concentrations generally decreased in downstream reaches during all sampling periods. Post reservoir drawdown mean TDS concentrations were on average higher in both upstream and downstream reaches compared to spring and summer.

The observed changes in water quality parameters may have altered the abundance and diversity of benthic macroinvertebrate communities downstream of Grand River reservoirs. However, other physical and biological factors may be more causative to benthic macroinvertebrate changes (Spence and Hynes, 1971; Ward, 1976). Therefore, further research to quantify the effects of specific water quality parameters on benthic macroinvertebrate abundance and diversity is required.

4.3.3 Temperature

Temperature changes downstream of reservoirs are considered key factors contributing to changes in stream ecosystem integrity. Several studies have shown that cooler temperatures below deep release reservoirs and warmer temperatures below surface release reservoirs during spring and summer can reduce benthic macroinvertebrate diversity. In relation to the present study, temperature changes recorded below Grand River reservoirs may have caused significant reductions of macroinvertebrate diversity downstream.

Cooler temperatures in reaches below deep release reservoirs during spring and summer can be potentially harmful to benthic macroinvertebrate communities. Nebeker (1971) and Lehmkuhl (1972) reported that many life cycle phenomena, such as hatching, growth and emergence, depend on thermal cues. The thermal constancy and seasonal temperature pattern below deep release dams may not provide the thermal signals essential for completion of life cycles for certain species (Nebeker, 1971; Lehmkuhl, 1972). Ward (1976) suggests that only species able to complete their life cycles under relatively constant thermal conditions would be able to occupy the stream below deep release dams in temperate regions.

Modification of the temperature regime in downstream reaches due to deep release reservoirs may specifically contribute to the absence of several species of Ephemeroptera (Lehmkuhl, 1972; Ward and Stanford, 1979; Scullion *et* al., 1982). Lehmkuhl (1972) determined that cooler temperatures below a deep release reservoir were responsible for the elimination of 15 species of Ephemeroptera. Such reductions may also impact other benthic fauna. Lehmkuhl (1972) concludes that all deep release reservoirs in temperate climates will ultimately cause downstream faunal depletion in North America.

Temperature differences recorded in reaches below deep release reservoirs in the Grand River watershed during spring and summer displayed considerable cooling where temperatures were decreased more than 9^{O}C (Figures 4.16-4.17). As a result, cooler water temperatures recorded downstream may explain the reduction of taxa richness and specifically the reduction of Ephemeroptera species. The observed reduction in benthic macroinvertebrate diversity therefore provides evidence that temperature changes below deep release reservoirs may adversely alter the diversity of aquatic ecosystems.

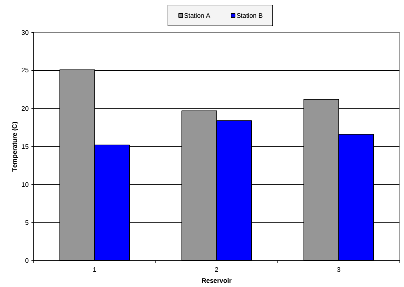

Figure 4.16. Temperatures (^{O}C) above and below deep release reservoirs: spring.

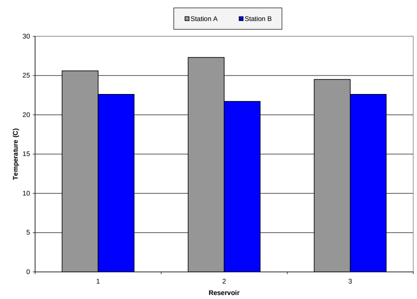

Figure 4.17. Temperatures (^{O}C) above and below deep release reservoirs: summer.

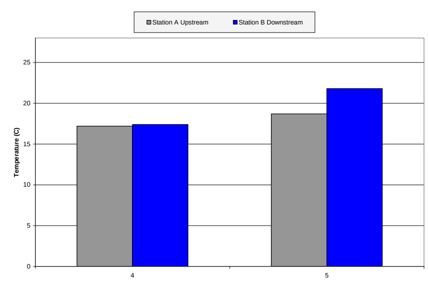

Figure 4.18. Temperatures (^{O}C) above and below surface release reservoirs: spring.

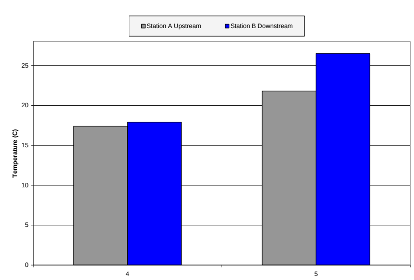

Figure 4.19. Temperatures (^{O}C) above and below surface release reservoirs: summer.

In contrast to deep release reservoirs, which generally produce cooler temperatures, few studies have examined the downstream temperature effects on benthic macroinvertebrates from surface release reservoirs, which generally produce warmer temperatures. In general, a great deal of literature has documented taxa richness and diversity to be linearly related to increasing stream temperature (Jacobsen *et al*., 1997). However, Lessard and Hayes (2003) observed that temperature increases below surface release dams coincided with a reduction in EPT taxa and these more tolerant organisms to higher temperatures were replaced by Chironomidae.

In reference to the present study, observed increases in temperatures downstream of surface release reservoirs in the Grand River watershed likely reduced the diversity of benthic macroinvertebrates downstream (Figures 4.18-4.19). This is most notable in Laurel Creek Reservoir (5) during spring and summer when downstream temperatures were increased by 8^{o}C above upstream sites. In downstream reaches below surface release dams, taxa richness and EPT taxa were reduced, while tolerant organisms, including Chironomidae, increased downstream. This suggests that temperature increases observed below surface release reservoirs may have contributed to the reduction of benthic macroinvertebrate diversity.

4.3.4 Stream Bed Modification

Modifications of the stream bed below reservoirs, changes in substrate particle size and organic matter content, have often been connected with lower habitat diversity and a reduction in benthic macroinvertebrate diversity. Modifications of the stream bed downstream of Grand River reservoirs may have directly or indirectly affected the abundance and diversity of benthic macroinvertebrates in the present study. However, additional controlled field and lab studies are required to define this relationship.

Heterogeneous particle size distribution of river bed sediment is important for providing varied microhabitats that support abundant and diverse macroinvertebrate fauna (Hynes, 1970; Ward, 1976; Scullion *et al*., 1982). Paterson and Fernando (1970) correlated the reduction in chironomid diversity below a surface release reservoir, Laurel Creek Reservoir, to changes in substrate heterogeneity. In the present study, only visual changes in substratum were observed. Overall, substratum heterogeneity was reduced and mean substrate particle size generally increased downstream in all reservoirs throughout the sampling duration. The coarsening as a result of the dam may have promoted changes in benthic macroinvertebrate abundance and diversity in downstream reaches in both deep release and surface release reservoirs.

Outflow from a deep release reservoir can increase the organic content downstream. Changes in organic matter downstream of impoundments have often been a key factor to changes in benthic macroinvertebrate communities (Cummings and Klug, 1979). Organic matter provides a vital food source that together with the substratum constitutes complex habitats for aquatic invertebrates (Cummings and Klug, 1979) and research has found that changes in organic matter can alter benthic macroinvertebrate functional feeding groups (Merritt and Cummins, 1996; Vallania *et al.*, 2007). The CPOM is reduced to FPOM since the transport of large size detritus is blocked and this generally decreases the abundance of shredders and increases the abundance filter-feeders downstream (Short and Ward, 1980; Ward, 1976). In the Grand River watershed, changes in organic content below reservoirs may have produced changes in benthic macroinvertebrate functional feeding groups. While shredders on average decreased downstream of deep release and surface release reservoirs, filter-feeders only increased downstream of one surface-release reservoir, Shade's Mills Dam (4) and therefore, relationships between reservoir outflow and benthic macroinvertebrate functional feeding groups were not very distinctive.

The hypolimnial outflow of organic matter downstream of deep release reservoirs, particularly zooplankton and phytoplankton, encourages detrivores (Spence and Hynes, 1971). Detritus feeders, *C. intermedius*, made up to 65% of the benthic composition downstream of deep release reservoirs.

4.3.5 Vegetation

Several authors have reported vegetation changes downstream of reservoirs. Decamps *et al.* (1979), Dudley *et al.* (1986) and Munn and Brusven (1991) documented that macroinvertebrate abundances are typically greater in areas with extensive plant growth due to increased habitat and food availability. While increased vegetation may favor some species, other macroinvertebrate fauna may negatively respond. For example, Ephemeroptera species often require rock susbtrata for colonization and therefore, any increase in algae on rock surfaces may prevent the establishment of certain forms of mayflies which utilize suckers or friction pads including species of Ephemerellidae and Heptageniidae. In reference to this study, enhanced algae populations were visually evident downstream of reservoirs and surface release reservoirs especially in the Grand River watershed. The increases in vegetation may have altered the abundance and diversity of benthic macroinvertebrate communities downstream of reservoirs.

4.3.6 Reservoir Drawdown

There is increasing concern regarding the disturbing effects of drawdown on stream ecosystem integrity and benthic macroinvertebrate populations. Sediment accumulation in reservoirs and subsequent nutrient pools can alter nutrient cycling and aquatic ecosystems in downstream environments (Shantz *et al*., 2004). During drawdown, the resuspension of sediments and particulate matter can alter downstream water quality (Shantz *et al*., 2004) and ultimately influence the abundance and diversity of benthic macroinvertebrate populations. In reference to this study, reservoirs in the Grand River watershed were drained in excess of 8 meters (Figure 4.20). However, Shantz *et al*. (2004) observed that lake levels drained 0.65 m below predrawdown conditions significantly increased suspended solids and TP concentrations downstream of Laurel Creek Reservoir.

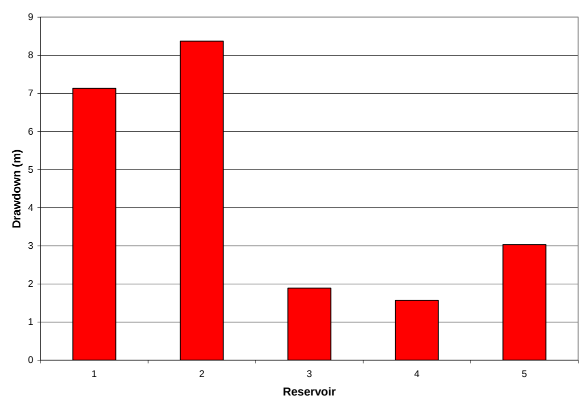

Figure 4.20. Reservoir drawdown (m): fall.

During fall, following drawdown, the abundance and diversity of benthic macroinvertebrate communities were altered. This is likely attributed to changes in bottom sediments and water quality changes. Downstream of surface release reservoir, Laurel Creek (5), which was drained completely to the stream channel (3.03 m), Chironomidae composed 70% of the benthic community and mean abundance of Chironomidae increased from 33.8 upstream to 197.5 downstream, and detrivores dominated benthic populations by 85%. Deep release reservoirs were drawn down over a longer period of time beginning in early summer. However, surface reservoir designs permitted complete drawdown in less than a month.

4.4 Implications for Watershed Planning and Reservoir Management

The impacts of river regulation on aquatic streams and watershed health have been increasingly recognized over the past few decades (Leopold, 1968; Petts, 1984). The use of reservoirs for flood control, irrigation and flow augmentation have been an integral part of watershed planning globally and in the Grand River watershed specifically (OMEE, 1993; Boyd *et al*., 2000). However, it is only in recent years that the Ontario government has suggested using watershed boundaries to integrate land use planning activities and water management objectives (OMOEE, 1993).

Watershed planning involves four basic stages: (1) issue identification and data gathering; (2) analysis and planning; (3) implementation; and (4) monitoring (Montgomery *et al*., 1995). A central issue in watershed planning is the implementation of ecosystem management, which is a set of conservation and protection strategies designed to reduce, limit or modify adverse effects of human activities on the aquatic environment and aquatic resources (OMEE, 1993; Montgomery *et al*., 1995). Ultimately, ecosystem management is founded on the principle of preserving ecosystem integrity while maintaining sustainable benefits to society (Montgomery *et al*., 1995). Therefore, the management of watersheds, including the regulation of rivers, is an integral part of watershed planning (OMEE, 1993) for the protection and preservation of stream ecosystems in receiving outflow.

Monitoring of downstream effects of reservoirs on aquatic ecosystems is essential to provide the information needed to update planning and management decisions needed to mitigate stream integrity (Montgomery *et al*., 1995). In the following sections, the implications of the present study detail the need for enhanced reservoir management strategies and ecosystem management practices for the conservation and preservation of aquatic stream ecosystems. In addition, the present study emphasizes the importance of biological monitoring and benthic macroinvertebrates in watershed planning and management.

4.4.1 Implications for Reservoir Planning and Management

Early reservoir design criteria and operation emphasized societal and economic uses of water and less attention was directed to long-term ecological consequences of flow management (Petts, 1984). The recognition for ecosystem preservation and enhancement developed only in recent years. Implementing ecosystem management approaches to watershed planning and decision making, including the supervision of reservoirs, requires new methods for linking science to planning (Montgomery *et al*., 1995). Such understanding is essential to make informed planning and management decisions to balance societal objectives against intrinsic landscape capabilities (Montgomery *et al*., 1995).

However, results of the present study and relevant literature suggest that the Grand River watershed planning and ecosystem management based decision making, as outlined by Ontario government, did not conserve nor preserve the integrity of stream ecosystems and benthic macroinvertebrate communities downstream of both deep release and surface release reservoirs. Watershed planning and reservoir management strategies should also consider the protection of downstream aquatic ecosystems and benthic macroinvertebrates. Watershed planning and reservoir management strategies must consider the influence of flow and temperature constancies, nutrient enriched waters, vegetation and substrate alterations, caused by impoundments, on downstream ecosystems and benthic macroinvertebrate communities.

In addition, the design and reservoir management strategies of deep release reservoirs adversely impacted downstream abundance and diversity of benthic macroinvertebrates more than the design and reservoir management strategies of surface release reservoirs in the Grand River watershed. The implications of this study and abundant literature support the need for further research on comparisons of deep release and surface release reservoirs management strategies on downstream aquatic ecosystems and benthic macroinvertebrate diversity and abundance.

4.4.2 Using Benthic Macroinvertebrates as Biological Indicators to Evaluate the Effects of River Regulation in Stream Ecosystems

Benthic macroinvertebrates are useful biological indicators and monitoring tools used to study the effects of reservoirs on downsteam ecosystems by measuring changes that could not be determined by chemical analyses alone. The Ontario government suggests using a variety of monitoring programs that use a range of physical, chemical and biological indicators, which are an integral component of watershed and subwatershed plans (OMEE, 1993). In the watershed analysis process, monitoring information is used to provide feedback on the status of aquatic resources and performances of policies, programs and legislation (Montgomery *et al*., 1995; Jones *et al.*, 2004).

In the past, monitoring, assessment and regulation of aquatic ecosystems has largely been based on physical and chemical measures of water quality. However, biological assessment is an important component of water quality and habitat monitoring programs can be more cost-effective than chemical testing (Barbour *et al*., 1999). In recent years, the Ontario Ministry of the Environment and Environment Canada have developed the Ontario Benthos Biomonitoring Network (OBBN), which is to be fully used to provide a biological complement to the Provincial Water Quality Monitoring Network in order to develop aquatic biocritera for the Province of Ontario (Jones *et al*., 2004).

Recent provincial and national initiatives like the OBBN underscore the importance of biological monitoring and the use of benthic macroinvertebrates. Results from the present study and literature indicate that biological monitoring using benthic macroinvertebrates are useful tools for understanding anthropogenic perturbations on stream ecosystems and ecological disturbances on individual populations. Therefore, benthic biological monitoring can be supplementary implemented at the GRCA to continue to monitor downstream effects of reservoirs on stream ecosystems and benthic macroinvertebrates in the Grand River watershed. In addition, benthic biological monitoring can be used to assess GRCA reservoir management strategies including deep release and surface release designs and reservoir drawdown.

Chapter 5: Conclusions and Recommendations

5.1 Conclusions

The main purpose of this study was to evaluate and compare the downstream effects of both deep release and surface release reservoir management strategies on benthic communities in the Grand River watershed. The outcomes of this research provide a better understanding of the environmental impacts of deep release and surface release reservoir management strategies on aquatic ecosystems. Based upon an analysis of results from the present study, the following conclusions are presented.

Effects of reservoirs on benthic macroinvertebrate communities

1. Invertebrate abundance decreased post reservoir drawdown, taxa richness decreased, EPT taxa reduced, HBI values increased, Isopoda and Chironomidae abundance increased and Ephemeroptera abundance reduced downstream (station B) from upstream (station A). It is likely that these streams were impacted by physical, chemical and biological changes induced by the impoundments from both deep release and surface release designs.

2. Benthic communities downstream of deep release reservoirs were adversely impacted compared to the benthic communities downstream of surface release reservoirs. While both reservoir management types experienced similar downstream variation in flow and reduction in habitat diversity and water quality, the altered thermal regime downstream of deep release reservoirs may have severely impacted the abundance and diversity of benthic macroinvertebrate populations. Temperature cooling downstream of deep release reservoirs may have considerably altered the diversity of benthic macroinvertebrate communities and specifically may have prompted the reduction and or elimination of several species of Ephemeroptera.

3. GRCA reservoir management strategies, including reservoir drawdown, are harmful to downstream benthic macroinvertebrate communities. Drawdown alters the abundance and diversity of benthic macroinvertebrate fauna and disturbs functional feeding trophic groups.

Implications for Planning and Management

4. Watershed planning and reservoir management strategies of both deep release and surface release reservoirs did not conserve or preserve the benthic macroinvertebrate communities downstream of reservoir outflow. Downstream aquatic stream ecosystems were disturbed and ecosystem integrity was not mitigated or maintained compared to upstream environments. Environmental degradation downstream of reservoirs must be reviewed and GRCA practices, such as reservoir drawdown, must be further examined. In addition, biological monitoring components of watershed planning and reservoir management decision making must be implemented for the conservation and preservation of downstream aquatic ecosystems and benthic macroinvertebrate communities.

5. Biological monitoring indicators and benthic macroinvertebrates measure the indirect effects of perturbations on biological changes and living organisms and provide important biological information which chemical indicators alone cannot. The use of biological indicators is an important tool for watershed planning to ensure that practices of ecosystem management are enforced in reservoir management and design.

Bibliography

Al-Lami, A. A., Jaweir, H. J., and Nashaat, M. R. (1998). Benthic invertebrate community of the River Euphrates upstream and downstream sectors of Al-Qadisia Dam, Iraq. *Regulated Rivers Research and Management*, 14, 383-390.

APHA (American Public Health Association). (2005). *Standard Methods for the Examination of Water and Wastewater* (19th ed.) New York: American Public Health Association.

Armitage, P. D. (1995). Faunal community change in response to flow manipulation. In D. M. Harper and A. J. D. Ferguson (Eds.), *The Ecological Basis for River Management* (pp. 59-80). London: John Wiley and Sons.

Barbour, M. T., Gerritsen, J., Snyder, B. D., and Stribbling, J. B. (1999). *Rapid Bioassessment Protocols for use in Streams and Wadeable Rivers: Periphyton, Benthic Macroinvertebrates and Fish* (2nd ed.). Washington: U.S. Environmental Protection Agency, Office of Water.

Barton, D. R., Kelton, N., and Eedy, R. I. (2000). The effects of carp (Cyprinus carpio L.) on sediment export from a small urban impoundment. *Journal of Aquatic Ecosystem Stress and Recovery*, 8(2), 155-159.

Barton, D. R. (1996). The use of Percent Model Affinity to assess the effects of agriculture on benthic invertebrate communities in headwater streams of southern Ontario, Canada. *Freshwater Biology*, 36, 397-410.

Bednarek, A. (2001). Undamming Rivers: A review of the ecological impacts of dam removal. *Environmental Management,* 27(6), 803-814.

Boulton, A. J., Findlay, S., Marmonier, P., Stanley, E. H., and Valett, H. M. (1998). The functional significance of the hyporheic zone in streams and rivers. *Annual Review of Ecology and Systematics*, 29, 59-81.

Bousfield, E. L. (1958). Fresh-water amphipod crustaceans of glaciated north America. *Can. Fld. Nat*., 72, 55-113.

Boyd, D., Smith, A. F., and Veale, B. (2000). *Flood Management on the Grand River Basin*. Cambridge Ontario: GRCA (Grand River Conservation Authority).

Brabec, E., Schulte, S., and Richards, P. L. (2002). Impervious surfaces and water quality: a review of current literature and its implications for watershed planning. *Journal of Planning Literature*, 16(4), 499-514.

Brandimarte, A. L., Anaya, M., and Shimizu, G. Y. (2005). Downstream impact of Mogi-Guacu River damming on the benthic invertebrates (Soa Paulo, Brazil). *Acta Limnologica Brasiliensia*, 17(1), 27-36.

Brussard, P. F., Reed, J. M., Tracy, C. R. (1998). Ecosystem management: what is it really? *Landscape and Urban Planning*, 40, 9-20.

Cairns, J. Jr. (1974). Indicator species vs. the concept of community structure as an index of pollution. *Water Research Bulletin*, 10, 338-347.

Camargo, J. A., and Jalon, G. D. (1990). The downstream impacts of the Burgomillodo reservoir, Spain. *Regulated Rivers: Research and Management*, 15, 395-403.

Cereghino, R., Cugny, P., and Lavandier, P. (2002). Influence of intermittent hydropeaking on the longitudinal zonation patterns of benthic invertebrates in a mountain stream. *International Review of Hydrobiology*, 87, 47-60.

Chapman, D., and Kimstach, V. (1996). Selection of water quality variables. In D. Chapman (Ed.), Water Quality Assessments: A Guide to the Use of Biota, Sediments and Water in Environmental Monitoring (2nd ed.) (pp. 58-126). London: UNESCO/WHO/UNEP, E and FN Spon.

Collier, M., Webb, R. H., and Schmidt, J. C. (1996). *Dams and Rivers: Primer on the Downstream Effects of Dams*. US Geological Survey.

Cortes, R. M. V., Ferreira, M. T., Oliveira, S. V., and Oliveira, D. (2002). Macroinvertebrate community structure in a regulated river segment with different flow conditions. *River Research and Applications*, 18, 367-382.

Cortes, R. M. V., Ferreira, M. T., Oliveira, S. V., Godinho, F. (1998). Contrasting impact of small dams on macroinvertebrates of two Iberian rivers. *Hydrobiologia*, 389, 51-61.

Cowell, B. C., Hull, H. C. Jr., and Fuller, A. Recolonization of small-scale disturbances by benthic invertebrates in Florida freshwater ecosystems. *Insect. Behav. Ecol.*, 70, 1-14.

Cummins, K. W., and Klug, K. W. (1979). Feeding ecology of stream invertebrates. *Annual Review of Ecology and Systematics*, 10, 14-172.

Decamps, H., Capblancq, J., Cassanova, H., Touring, J. M. (1979). Hydrobiology of some regulated rivers in the south-west of France. In J. V. Ward, and J. A. Stanford. (Eds.), *The Ecology of Regulated Streams* (pp. 273-288). New York: Plenum Press.

De Pauw, N., and Vanhooren, G. (1983). Method for biological quality assessment of watercourses in Belgium. *Hydrobiologia*, 100, 153-168.

Dudley, J. L., Cooper, S. D., and Hemphill, N. (1986). Effects of macroalgae on a stream invertebrates community. *Journal of the North American Benthological Society*, 5, 93-106.

Fisher, W. S. (1998). Development and validation of ecological indicators: an ORD approach. *Environmental Monitoring and Assessment*, 51(1-2), 23-28.

Friedrich, G., Chapman, D., and Beim, A. (1996). The use of biological material. In D. Chapman (Ed.), Water Quality Assessments: A Guide to the use of Biota, Sediments and Water in Environmental Monitoring (2nd ed.) (pp. 175-242). London: UNESCO/WHO/UNEP, E and FN Spon.

Furey, P. C., Nordin, R. N., and Mazumder, A. (2006). Littoral benthic macroinvertebrates under contrasting drawdown in a reservoir and a natural lake. *Journal of the North American Benthological Society*, 25(1), 19-31.

Gore, J. A., and Petts, G. E. (1989). *Alternatives in Regulated River Management*. Boca Raton: CRC Press Inc.

Gore, J. A. (1980). Ordinational analysis of benthic communities upstream and downstream of a prairie storage reservoir. *Hydrobiologia*, 96, 33-44.

GRCA (Grand River Conservation Authority). (2004). *The Grand*. Cambridge, Ontario: Grand River Conservation Authority.

Green, R. H. (1979). *Sampling Design and Statistical Methods for Environmental Statistical Methods for Environmental Biologists*. New York: Wiley.

Harvey, E. (2005). Personal Communications. Waterloo: Statistical Consulting Service, University of Waterloo.

Haslam, S. M. (1982). A proposed method for monitoring river pollution using macrophytes. *Environmental Technology Letters*, 3(1), 19-34.

Hellawell, J. M. (1986). *Biological Indicators of Freshwater Pollution and Environmental Management*. London: Elsevier.

Hilsenhoff, W. L. (1988). Rapid field assessment of organic pollution with a family-level biotic index. *Journal of the North American Benthological Society*, 7(1), 65-68.

Hilsenhoff, W. L. (1987). An improved biotic index of organic stream pollution. *The Great Lakes Entomologist*, 21(1), 31-39.

Hilsenhoff, W. L. (1971). Changes in the downstream insect and amphipod fauna caused by an impoundment with a hypolimnion drain. *Annals of the Entomological Society of America*, 64(3), 743-746.

Hofmann, N. (2001). Urban consumption of agricultural land. *Rural and Small Town Canada Analysis Bulletin*, 3(2), 1-13.

Hynes, H. B. N. (1970). The Ecology of Running Waters. Liverpool: Liverpool University Press.

Illies, J., and Botosaneany, L. (1963). Problems et methods de la classification et de la zonation ecologique des eaux courantes considerees surtout du point de rue faunistique. *Mitteilungen Der Internationalen Vereinigung fur Theoretische Und Angewandte Limnologie*, 12, 1-57.

Irvine, J. R., and Henriques, P. R. (1984). A preliminary investigation on effects of fluctuating flows on invertebrates of the Hawea River, a large regulated river in New Zealand. *New Zealand Journal of Marine and Freshwater Research*, 18, 283-290.

Jacobsen, D., Schultz, R., Encalada, A. (1997). Structure and diversity of stream invertebrate assemblages: the influence of temperature with altitude and latitude. *Freshwater Biology*, 38, 247-261.

Jones, C., Somers, K. M., Craig, B., and Reynoldson, T. B. (2004). *Ontario Benthos Biomonitoring Network Protocol Manual*. Ontario: Ontario Ministry of the Environment and Environment Canada.

Junk, J. W., Bayley, P. B., and Sparks, R. E. (1989). The flood pulse concept in river-floodplain systems. In D. P. Dodge (Ed.), *Proceedings of the International Large River Symposium, Canadian Journal of Fisheries and Aquatic Science Special Publication*, 106, 110-127.

Karr, J. R. and Chu, E. W. (1997). Biological Monitoring and Assessment: Using Multimetric Indexes Effectively. Seattle: University of Washington. Environmental Protection Agency.

Karr, J. R. (1991). Biological integrity: A long-neglected aspect of water resource management. *Ecological Applications*, 1(1), 66-84.

Kaster, J. L., and Jacobi, G. Z. (1978). Benthic macroinvertebrates of a fluctuating reservoir. *Freshwater Biology*, 8, 283-290.

Kerans, B. L., and Karr, J. R. (1994). A Benthic Index of Biotic Integrity (B-IBI) for rivers of the Tennessee Valley. *Ecological Applications*, 4(4), 768-785.

Kilgour, B. W., and Barton, D. R. (1999). Association between stream fish and benthos across environmental gradients in southern Ontario, Canada. *Freshwater Biology*, 41, 553-566.

Kroger, R. L. (1973). Biological effects of fluctuating water levels in the Snake River, Grand Teton National Park, Wyoming. *The American Midland Naturalist*, 89, 478-481.

Lehmkuhl, D. M. (1972). Change in thermal regime as a cause of reduction of benthic fauna downstream of a reservoir. *Journal of Fisheries Research Board of Canada*, 29, 1329-1332.

Lenat, D. R. (1988). Water quality assessment of streams using a qualitative collection method for benthic macroinvertebrates. *Journal of the North American Benthological Society*, 7(3), 222-233.

Loeb, S. L., and Spacie, A. (1994). *Biological Monitoring of Aquatic Systems*. Boca Raton: Lewis Publishers.

Leopold, L. B. (1968). *Hydrology for Urban Land Planning: A Guidebook on the Hydrologic Effects of Urban Land Use* (Geological Survey Circular 554 ed.). Washington: United States Geological Survey.

Leopold, L. B., Gordon, W. M., and Miller, J. P. (1964). *Fluvial Processes in Geomorphology*. San Francisco: W. H. Freeman.

Lessard, J. L., and Hayes, D. B. (2003). Effects of elevated water temperature on fish and macroinvertebrate communities below small dams. *River Research and Applications*, 19, 721-732.

Lorenz, C. M., Van Dijk, G. M., Van Hattum, A. G. M, and Cofino, W. P. (1997). Concepts in river ecology: Implications for indicator development. *Regulated Rivers Research and Management*, 13(6), 501-516.

Mackie, G. L. (2000). *Common Algae, Macrophytes, Benthic Invertebrates and Zooplankton in the Speed River Watershed*. Guelph, Ontario: Department of Zoology, University of Guelph.

Markert, B. A., Breure, A. M., and Zechmeister, H. G. (2003). *Bioindicators and Biomonitors: Principles, Concepts and Applications*. London: Elsevier.

Maul, J. D., Farris, J. L., Milam, C. D., Cooper, C. M., Testa III, S., and Feldman, D. L. (2004). The influence of stream habitat and water quality on macroinvertebrate communities in degraded streams of northwest Mississippi. *Hydrobiologia*, 518, 79-94.

McCartney, M. P., Sullivan, C., and Acreman, C. (2000). Ecosystem Impacts of Large Dams. World Commission on Dams. Retrieved Fed 15, 2006 from http://www.dams.org/docs/kbase/contrib/env244.pdf

Merritt, R. W., and Cummins, K. W. (1996). *An Introduction to the Aquatic Insects of North America*. Dubuque, Iowa: Kendall/Hunt Pub. Co.

Metcalfe-Smith, J. L. (1994). Biological water-quality assessment of rivers: use of macroinvertebrate communities. In P. Callow, and G. E. Petts (Eds.), *The Rivers Handbook: Hydrological and Ecological Principles (Vol II)* (pp. 76-97). Oxford: Blackwell Scientific Publications.

Michaletz, P. H., Doisy, K. E., and Rabeni, C. F. (2005). Influence of productivity, vegetation, and fish on macroinvertebrate abundance and size in Midwestern U.S.A. impoundments. *Hydrobiologia*, 543(1), 147-158.

Milner, A. M. (1994). System recovery. In P. Calow, and G. E. Petts (Eds.), *The Rivers Handbook: Hydrological and Ecological Principles (Vol. II)* (pp. 76-97). Oxford: Blackwell Scientific Publications.

Minshall, G. W. (1984). Aquatic insect-substratum relationships. In V. H. Resh and D. M. Rosenberg (Eds.), *The Ecology of Aquatic Insects* (pp. 359-400). New York: Praeger.

Montgomery, D. R., Grant, G. E., and Sullivan, K. (1995). Watershed analysis as a framework for implementing ecosystem management. *Water Resources Bulletin*, 31(3), 369-386.

Moreno, P., and Callisto, M. (2006). Benthic macroinvertebrates in the watershed of an urban reservoir in southeastern Brazil. *Hydrobiologia*, 560, 311-321.

Munn, M. D., and Brusven, M. A. (1991). Benthic macroinvertebrate communities in nonregulated and regulated waters of the Clearwater River, Idaho, U.S.A. *Regulated Rivers: Research and Management*, 6, 1-11.

Mwaura, F., Mavuti, K., and Wamicha, W. N. (2002). Biodiversity characteristics of small high-altitude tropical man-made reservoirs in the Eastern Rift Valley, Kenya. *Lakes and Reservoir: Research and Management*, 7(1), 1-12.

Nebeker, A. V. (1971). Effect of high winter water temperatures on adult emergence of aquatic insects. *Water Research*, 5, 777-783.

Newbold, J. D., Elwood, J. W., O'Neill, R. V., and Van Winkle, W. (1981). Measuring nutrient spiralling in streams. *Canadian Journal of Fisheries Aquatic Sciences*, 38(7), 860-863.

Ogbeibu, A. E., and Oribhabor, B. J. (2001). Ecological impact of river impoundment using benthic macro-invertebrates as indicators. *Water Research*, 36, 2427-2436.

Olive, J.H., Jackson, J. L., Bass, J., Holland, L., Savisky, T. (1988). Benthic Macroinvertebrates as indexes of water quality in the Upper Cuyahoga River. *Ohio Journal of Science*, 88(3), 91-98.

OMAFRA (Ontario Ministry of Agriculture, Food and Rural Affairs). (2000). *Soil Types of Ontario (Computer File)*. Guelph, Ontario: OMAFRA.

OMOEE (Ontario Ministry of the Environment and Energy) (1993). *Guidelines for the Protection and Management of Aquatic Sediment Quality in Ontario*. Toronto: Queens' Printer for Ontario.

Parr, W. (1994). Water quality monitoring. In P. Callow, and G. E. Petts (Eds.), *The Rivers Handbook: Hydrological and Ecological Principles (Vol. II)* (pp. 124-143). Oxford: Blackwell Scientific Publications.

Paul, M. J., and Meyer, J. L. (2001). Streams in the urban landscape. Annual Review of Ecology and Systematics, 32, 333-365.

Paterson, C. G., and Fernando, C. H. (1970). Benthic colonization of a new reservoir with particular reference to the Chironomidae. *Journal of Fisheries Research Board of Canada*, 27, 213-232.

Petts, G. E. (1994). Rivers: Dynamic Components of Catchment Ecosystems. In P. Calow and G. E. Petts (Eds.), *The Rivers Handbook: Hydrological and Ecological Principles (Vol. II)* (pp. 3-22). Oxford: Blackwell Scientific Publications.

Petts, G. E. (1984). *Impounded Rivers: Perspectives for Ecological Management*. Chichester England: John and Wiley & Sons.

Pizzuto, J. (2002). Effects of dam removal on river form and process. *BioScience*, 52(8), 683-691.

Poff, N. L., and Hart, D. D. (2002). How dams vary and why it matters for the emerging science of dam removal. *BioScience*, 52(8), 659-668.

Pratt, J. M., and Coler, R. A. (1976). A procedure for the routine evaluation of urban runoff in small rivers. *Water Research*, 10, 1019-1025.

Reynoldson, T. B. (1984). The utility of benthic invertebrates in water quality monitoring. *Water Quality Bulletin*, 10, 21-28.

Richardson, S. M., Hanson, J. M., and Locke, A. (2002). Effects of impoundment and water-level fluctuations on macrophyte and macroinvertebrate communities of a dammed tidal river. *Aquatic Ecology*, 36, 493-510.

Rosenberg, D. M., and Resh, V. H. (1993). *Freshwater Biomonitoring and Benthic Macroinvertebrates*. New York: Chapman and Hall.

Scullion, J., Parish, C. A., Morgan, N., and Edwards, R. W. (1982). Comparison of benthic macroinvertebrate fauna and substratum composition in riffles and pools in the impounded River Elan and the unregulated River Wye, mid-Wales. *Freshwater Biology*, 12(6), 579-595.

Sephton, T. W., Hicks, B. A., Fernando, C. H., and Paterson, C. G. (1983). Changes in the chironomid (diptera: chironomidae) fauna of Laurel Creek Reservoir, Waterloo, Ontario. *Journal of Freshwater Ecology*, 2(1), 89-102.

Shantz, M., Dowsett, E., Canham, E., Tavenier, G., Stone, M., and Price, J. (2004). The effect of drawdown on suspended solids and phosphorous export from Columbia Lake, Waterloo, Canada. *Hydrological Processes*, 18, 865-878.

Short, R. A., and Ward, J. V. (1980). Leaf litter processing in a regulated rocky mountain stream. *Journal of Fisheries Aquatic Science*, 37, 123-127.

Spence, J. A., and Hynes, H. B. N. (1971). Difference in benthos upstream and downstream of an impoundment. *Journal of Fisheries Research Board of Canada*, 28(1), 35-43.

Stanford, J. A. and Ward, J. V. (2001). Revisiting the serial discontinuity concept. *Regulated Rivers Research and Management*, 17, 303-310.

Stanford, J. A., and Ward, J. V. (1993). An ecosystem perspective of alluvial rivers: connectivity and the hyporheic corridor. *Journal of the North American Benthological Society*, 12(1), 48-60.

Stanford, J. A., and Ward, J. V. (1979). Stream Regulation in North America. In J. V. Ward, and J. A. Stanford (Eds.), *The Ecology of Regulated Streams*. New York: Plenum Press.

Statzner, B., and Higler, B. (1986). Stream hydraulics as a major determinant of benthic invertebrate zonation patterns. *Freshwater Biology*, 16(1), 127-139.

Symons, J. M. (1969). *Water Quality Behaviour in Reservoirs*. Cincinatti, Ohio: U. S. Department of Health, Education and Welfare.

Thorp, J. H., and Covich, A. P. (2001). *Ecology and Classification of North American Freshwater Invertebrates*. San Diego: Academic Press.

Thorp, J. H., and Delong, M. D. (1994). The riverine productivity model: a heuristic view of carbon-sources and organic-processing in large river ecosystems. *Oikos*, 70(2), 305-308.

University of Waterloo. (2005). *University of Waterloo Weather Station Data Archives*. Retrieved March 5, 2006 from http://weather.uwaterloo.ca/data.htm.

Vallania, A., and Corigliano, M. D. C. (2007). The effect of regulation caused by a dam on the distribution of the functional feeding groups of the benthos in the sub basin of the Grande River (San Luis, Argentina). *Environmental Monitoring and Assessment*, 124(1-3), 201-209.

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

Voshell, J. R. Jr., and Simmons, G. M. Jr. (1984). Colonization and succession of benthic macroinvertebrates in a new reservoir. *Hydrobiologia*, 112, 27-39.

Ward, J. V., and Stanford, J. A. (1987). The ecology of regulated streams: past accomplishments and direction for future research. In J. F. Craig, and J. B. Kemper (Eds.), *International*

Symposium on Regulated Streams (3rd ed.): Regulated Streams Advances in Ecology (pp. 391-409). New York: Plenum Press.

Ward, J. V., and Stanford, J. A. (1984). The regulated stream as a testing ground for ecological theory. In A. Lillehammer, and S. J. Saltveit (Eds.), Regulated Rivers (pp. 139-165). Norway: Oslo University Press.

Ward, J. V. (1984). (1984). Ecological perspectives in the management of aquatic insect habitat. In V. H. Resh, and D. M. Rosenberg (Eds.), The Ecology of Aquatic Insects (pp. 558-577). New York: Praeger Publishers.

Ward, J. V., and Stanford, J. A. (1983). The serial discontinuity concept of lotic ecosystems. In T. D. Fontaine, and S. M. Bartell (Eds.), *Dynamics of Lotic Ecosystems* (pp. 29-42). Ann Arbor: Ann Arbor Science Publishers.

Ward, J. V. (1976). Comparative limnology of differentially regulated sections of a Colorado mountain river. *Archiv fur Hydrobiologie*, 78(3), 319-342.

Ward, J. V. (1975). Downstream fate of zooplankton from a hypolimnial release mountain reservoir. *Verh. Int. Verein. Theor. Angew. Limnol*, 19, 1798-1804.

Whitehurst, I. T. (1991). The Gammarus: Asellus ratio as an index of organic pollution. *Water Research*, 25(3), 333-339.

Appendix A: Photographs of Study Sites

Figure A.1. Station 1A (Shand Dam, upstream).

Figure A.2. Station 1B (Shand Dam, downstream).

Figure A.3. Station 2A (Conestogo Dam, upstream).

Figure A.4. Station 2B (Conestogo Dam, downstream).

Figure A.5. Station 3A (Guelph Dam, upstream).

Figure A.6. Station 3B (Guelph Dam, downstream).

Figure A.7. Station 4A (Shade's Mills Dam, upstream).

Figure A.8. Station 4B (Shade's Mills Dam, downstream).

Figure A.9. Station 5A (Laurel Creek Reservoir, upstream).

Figure A.10. Station 5B (Laurel Creek Reservoir, downstream).

Appendix B: Complete Taxonomic Lists

Sample	Taxon	Count
1A1SPG	Tubificidae	5
	Sphaerium spp.	5
	Ephemerella spp.	18
	Stenonema spp.	16
	Baetis spp.	10
	Caenis spp.	14
	Paragnetina spp.	1
	Stenelmis spp.	27
	Optiocervus spp.	3
	Psephenus spp.	5
	Petrophila spp.	1
	Simulium spp.	1
	Hemerodromia spp.	6
	Hydropsyche spp.	11
	Chimarra spp.	5
	Diamesinae	10
	Pentaneurini	13
	Orthocladiinae	48
	Chironomini	1
	Tanytarsini	16
	Total	**216**
1A2SPG	*Sphaerium spp.*	2
	Ephemerella spp.	21
	Stenonema spp.	14
	Baetis spp.	5
	Stenelmis spp.	29
	Psephenus spp.	7
	Petrophila spp.	1
	Simulium spp.	2
	Hemerodromia spp.	7
	Tipula spp.	1
	Hydropsyche spp.	5
	Cheumatopsyche spp.	1
	Chimarra spp.	2
	Diamesinae	1
	Pentaneurini	6
	Orthocladiinae	29
	Chironomini	2
	Tanytarsini	11
	Total	**146**
1A3SPG	Tubificidae	1
	Amnicola limosa	1
	Ephemerella spp.	17
	Stenonema spp.	24
	Baetis spp.	8
	Caenis spp.	7
	Stenelmis spp.	12
	Hydropsyche spp.	14
	Agraylea spp.	5
	Diamesinae	1
	Pentaneurini	1
	Orthocladiinae	10
	Tanytarsini	2
	Total	**103**
1A4SPG	Tubificidae	4
	Valvata tricarinata	2
	Amnicola limosa	1
	Ephemerella spp.	16
	Stenonema spp.	11
	Baetis spp.	14
	Stenelmis spp.	25
	Psephenus spp.	4
	Petrophila spp.	1
	Cheumatopsyche spp.	2
	Agraylea spp.	3
	Diamesinae	1
	Total	**84**

Sample	Taxon	Count
1A5SPG	*Ephemerella spp.*	29
	Stenonema spp.	19
	Baetis spp.	17
	Caenis spp.	2
	Paragnetina spp.	1
	Stenelmis spp.	15
	Psephenus spp.	13
	Tipula spp.	1
	Hydropsyche spp.	10
	Chimarra spp.	4
	Agraylea spp.	6
	Limnephilus spp.	2
	Psilotreta spp.	1
	Pentaneurini	1
	Orthocladiinae	3
	Pseudochironomus spp.	1
	Chironomini	1
	Tanytarsini	2
	Total	**128**
1A6SPG	Tubificidae	1
	Physella integra	1
	Valvata tricarinata	5
	Amnicola limosa	1
	Ephemerella spp.	18
	Caenis spp.	5
	Stenelmis spp.	18
	Optiocervus spp.	2
	Psephenus spp.	15
	Paraponynx spp.	1
	Petrophila spp.	2
	Simulium spp.	1
	Hemerodromia spp.	2
	Stratiomys spp.	1
	Cheumatopsyche spp.	2
	Chimarra spp.	1
	Hydroptila spp.	1
	Neophylax spp.	8
	Diamesinae	1
	Pentaneurini	1
	Orthocladiinae	11
	Total	**98**

Sample	*Taxon*	Count
1B1SPG	*Nematoda*	2
	Dugesia spp.	3
	Naididae	2
	Tubificidae	4
	Hydrachnida	2
	Caecidotea intermedius	10
	Stenelmis spp.	2
	Optiocervus spp.	4
	Simulium spp.	3
	Hemerodromia spp.	2
	Antocha spp.	2
	Hydropsyche spp.	9
	Cheumatopsyche spp.	90
	Diamesinae	2
	Pentaneurini	16
	Orthocladiinae	23
	Chironomini	66
	Tanytarsini	1
	Total	**243**
1B2SPG	*Dugesia spp.*	4
	Naididae	5
	Caecidotea intermedius	24
	Paraponynx spp.	1
	Simulium spp.	40
	Prosimulium spp.	1
	Hemerodromia spp.	1
	Hydropsyche spp.	5
	Cheumatopsyche spp.	50
	Glossosoma spp.	7
	Diamesinae	2
	Pentaneurini	8
	Orthocladiinae	82
	Chironomini	90
	Tanytarsini	4
	Total	**324**

Sample	Taxon	Count
1B3SPG	*Dugesia spp.*	13
	Tubificidae	2
	Caecidotea intermedius	56
	Crangonyx spp.	5
	Simulium spp.	35
	Hydropsyche spp.	2
	Cheumatopsyche spp.	2
	Glossosoma spp.	3
	Pentaneurini	8
	Orthocladiinae	76
	Chironomini	26
	Total	**228**
1B4SPG	*Dugesia spp.*	12
	Caecidotea intermedius	40
	Optiocervus spp.	3
	Simulium spp.	60
	Prosimulium spp.	1
	Hydropsyche spp.	5
	Cheumatopsyche spp.	56
	Glossosoma spp.	7
	Pentaneurini	2
	Orthocladiinae	66
	Chironomini	28
	Total	**280**
1B5SPG	Naididae	47
	Tubificidae	18
	Dina spp.	7
	Caecidotea intermedius	29
	Crangonyx spp.	4
	Baetis spp.	7
	Simulium spp.	3
	Brachycentrus spp.	4
	Orthocladiinae	62
	Chironomini	87
	Tanytarsini	4
	Total	**272**
1B6SPG	*Dugesia spp.*	20
	Dina spp.	3
	Caecidotea intermedius	64
	Simulium spp.	57
	Hydropsyche spp.	10
	Cheumatopsyche spp.	23
	Polycentropus spp.	4
	Orthocladiinae	60
	Chironomini	34
	Total	**275**
2A1SPG	*Dugesia spp.*	2
	Naididae	124
	Sphaerium spp.	2
	Caecidotea intermedius	1
	Baetis spp.	39
	Caenis spp.	5
	Aeshna spp.	1
	Nigronia spp.	1
	Stenelmis spp.	5
	Optiocervus spp.	6
	Psephenus spp.	16
	Simulium spp.	8
	Hydropsyche spp.	88
	Glossosoma spp.	1
	Chimarra spp.	11
	Pycnopsyche spp.	36
	Pentaneurini	6
	Orthocladiinae	50
	Chironomini	13
	Tanytarsini	9
	Total	**424**

Sample	Taxon	Count
2A2SPG	*Dugesia spp.*	10
	Naididae	3
	Tubificidae	2
	Sphaerium spp.	7
	Ephemerella spp.	1
	Baetis spp.	39
	Caenis spp.	14
	Argia spp.	1
	Nigronia spp.	1
	Stenelmis spp.	19
	Optiocervus spp.	14
	Psephenus spp.	6
	Bezzia spp.	1
	Hydropsyche spp.	27
	Glossosoma spp.	1
	Chimarra spp.	1
	Pycnopsyche spp.	34
	Pentaneurini	5
	Orthocladiinae	7
	Chironomini	15
	Tanytarsini	5
	Total	**213**
2A3SPG	*Dugesia spp.*	4
	Naididae	1
	Hydrachnida	3
	Caecidotea intermedius	5
	Baetis spp.	16
	Caenis spp.	5
	Nigronia spp.	1
	Optiocervus spp.	2
	Psephenus spp.	4
	Hydropsyche spp.	27
	Glossosoma spp.	1
	Pentaneurini	6
	Orthocladiinae	3
	Chironomini	19
	Tanytarsini	1
	Total	**98**

Sample	Taxon	Count
2A4SPG	*Dugesia spp.*	13
	Naididae	5
	Sphaerium spp.	2
	Hydrachnida	4
	Baetis spp.	37
	Caenis spp.	16
	Stenelmis spp.	19
	Optiocervus spp.	14
	Psephenus spp.	19
	Simulium spp.	1
	Hydropsyche spp.	41
	Glossosoma spp.	1
	Chimarra spp.	9
	Pycnopsyche spp.	22
	Pentaneurini	5
	Orthocladiinae	35
	Chironomini	67
	Tanytarsini	24
	Total	**334**
2A5SPG	*Dugesia spp.*	14
	Naididae	3
	Tubificidae	2
	Hydrachnida	3
	Ephemerella spp.	1
	Baetis spp.	30
	Caenis spp.	13
	Aeshna spp.	2
	Argia spp.	1
	Stenelmis spp.	16
	Optiocervus spp.	19
	Psephenus spp.	19
	Hydropsyche spp.	80
	Glossosoma spp.	1
	Chimarra spp.	2
	Pycnopsyche spp.	25
	Brachycentrus spp.	1
	Pentaneurini	8
	Orthocladiinae	15
	Chironomini	44
	Tanytarsini	17
	Total	**316**

Sample	Taxon	Count
2A6SPG	*Dugesia spp.*	20
	Naididae	1
	Hydrachnida	2
	Baetis spp.	28
	Caenis spp.	11
	Sialis spp	1
	Stenelmis spp.	17
	Optiocervus spp.	11
	Psephenus spp.	9
	Simulium spp.	20
	Hydropsyche spp.	71
	Glossosoma spp.	1
	Chimarra spp.	6
	Pentaneurini	9
	Orthocladiinae	30
	Chironomini	65
	Tanytarsini	15
	Total	**317**
2B1SPG	*Nematoda*	13
	Dugesia spp.	6
	Naididae	10
	Sphaerium spp.	3
	Hydrachnida	10
	Caecidotea intermedius	132
	Serratella spp.	8
	Baetis spp.	2
	Stenelmis spp.	9
	Optiocervus spp.	3
	Petrophila spp.	3
	Simulium spp.	9
	Glossosoma spp.	2
	Chimarra spp.	7
	Pentaneurini	3
	Orthocladiinae	61
	Chironomini	46
	Tanytarsini	13
	Total	**340**

Sample	Taxon	Count
2B2SPG	*Dugesia spp.*	26
	Naididae	4
	Tubificidae	2
	Caecidotea intermedius	47
	Serratella spp.	23
	Baetis spp.	4
	Stenelmis spp.	22
	Optiocervus spp.	16
	Psephenus spp.	2
	Simulium spp.	2
	Hydropsyche spp.	16
	Chimarra spp.	3
	Orthocladiinae	90
	Chironomini	38
	Tanytarsini	11
	Total	**306**
2B3SPG	*Dugesia spp.*	11
	Naididae	14
	Tubificidae	10
	Sphaerium spp.	1
	Hydrachnida	20
	Caecidotea intermedius	51
	Serratella spp.	10
	Stenelmis spp.	22
	Optiocervus spp.	5
	Psephenus spp.	1
	Hydropsyche spp.	32
	Brachycentrus spp.	2
	Pentaneurini	2
	Orthocladiinae	97
	Chironomini	44
	Tanytarsini	3
	Total	**325**

Sample	Taxon	Count
2B4SPG	*Nematoda*	2
	Dugesia spp.	10
	Naididae	5
	Tubificidae	2
	Hydrachnida	29
	Caecidotea intermedius	108
	Serratella spp.	5
	Baetis spp.	1
	Stenelmis spp.	3
	Optiocervus spp.	14
	Hydropsyche spp.	32
	Brachycentrus spp.	3
	Pentaneurini	5
	Orthocladiinae	72
	Chironomini	33
	Tanytarsini	9
	Total	**333**
2B5SPG	*Dugesia spp.*	4
	Naididae	4
	Tubificidae	3
	Sphaerium spp.	1
	Hydrachnida	17
	Caecidotea intermedius	26
	Serratella spp.	15
	Stenelmis spp.	8
	Optiocervus spp.	17
	Psephenus spp.	1
	Bezzia spp.	1
	Hydropsyche spp.	14
	Chimarra spp.	20
	Brachycentrus spp.	2
	Pentaneurini	2
	Orthocladiinae	166
	Chironomini	24
	Tanytarsini	8
	Total	**333**

Sample	Taxon	Count
2B6SPG	*Nematoda*	14
	Dugesia spp.	7
	Naididae	1
	Sphaerium spp.	3
	Hydrachnida	21
	Caecidotea intermedius	49
	Serratella spp.	9
	Baetis spp.	3
	Stenelmis spp.	10
	Optiocervus spp.	4
	Petrophila spp.	3
	Simulium spp.	4
	Hydropsyche spp.	99
	Diamesinae	1
	Pentaneurini	7
	Orthocladiinae	78
	Chironomini	35
	Tanytarsini	7
	Total	**355**
3A1SPG	*Nematoda*	3
	Dugesia spp.	1
	Naididae	2
	Caecidotea intermedius	1
	Serratella spp.	5
	Stenonema spp.	3
	Baetis spp.	53
	Caenis spp.	21
	Paragnetina spp.	1
	Nigronia spp.	1
	Stenelmis spp.	3
	Hydropsyche spp.	12
	Pentaneurini	1
	Orthocladiinae	5
	Chironomini	1
	Total	**113**

Sample	Taxon	Count
3A2SPG	*Nematoda*	2
	Naididae	1
	Stenonema spp.	3
	Baetis spp.	61
	Paraleptophlebia spp.	1
	Caenis spp.	16
	Paragnetina spp.	1
	Nigronia spp.	1
	Stenelmis spp.	4
	Optiocervus spp.	2
	Psephenus spp.	1
	Hydropsyche spp.	5
	Pentaneurini	1
	Orthocladiinae	2
	Chironomini	1
	Tanytarsini	1
	Total	**103**
3A3SPG	Naididae	1
	Erpobdella spp.	2
	Serratella spp.	5
	Baetis spp.	41
	Caenis spp.	25
	Stenelmis spp.	9
	Optiocervus spp.	3
	Hydropsyche spp.	6
	Glossosoma spp.	1
	Orthocladiinae	1
	Chironomini	1
	Tanytarsini	4
	Total	**99**

Sample	Taxon	Count
3A4SPG	*Ephemerella spp.*	2
	Serratella spp.	5
	Baetis spp.	87
	Paraleptophlebia spp.	2
	Caenis spp.	4
	Nigronia spp.	1
	Stenelmis spp.	8
	Optiocervus spp.	7
	Stratiomys spp.	1
	Hydropsyche spp.	12
	Glossosoma spp.	1
	Pentaneurini	4
	Orthocladiinae	8
	Chironomini	3
	Tanytarsini	1
	Total	**146**
3A5SPG	*Nematoda*	1
	Dugesia spp.	1
	Naididae	1
	Erpobdella spp.	3
	Crangonyx spp.	1
	Serratella spp.	8
	Stenonema spp.	2
	Baetis spp.	43
	Paraleptophlebia spp.	1
	Caenis spp.	23
	Leuctra spp.	1
	Nigronia spp.	1
	Hydropsyche spp.	3
	Orthocladiinae	8
	Total	**97**

Sample	Taxon	Count
3A6SPG	Naididae	1
	Hydrachnida	2
	Stenonema spp.	5
	Baetis spp.	22
	Caenis spp.	3
	Stenelmis spp.	3
	Optiocervus spp.	6
	Psephenus spp.	2
	Hydropsyche spp.	2
	Neophylax spp.	2
	Ryacophila spp.	1
	Brachycentrus spp.	1
	Pentaneurini	2
	Orthocladiinae	2
	Chironomini	1
	Tanytarsini	1
	Total	**56**
3B1SPG	Naididae	14
	Sphaerium spp.	1
	Hydrachnida	6
	Caecidotea intermedius	16
	Baetis spp.	1
	Stenelmis spp.	1
	Optiocervus spp.	3
	Simulium spp.	3
	Prosimulium spp.	1
	Hydropsyche spp.	7
	Cheumatopsyche spp.	2
	Glossosoma spp.	6
	Agraylea spp.	1
	Diamesinae	2
	Pentaneurini	3
	Orthocladiinae	96
	Tanytarsini	40
	Total	**203**

Sample	Taxon	Count
3B2SPG	*Dugesia spp.*	2
	Naididae	10
	Dina spp.	1
	Caecidotea intermedius	204
	Baetis spp.	1
	Stenelmis spp.	6
	Hydropsyche spp.	2
	Glossosoma spp.	9
	Hydroptila spp.	2
	Limnephilus spp.	1
	Pentaneurini	2
	Orthocladiinae	138
	Tanytarsini	24
	Total	**402**
3B3SPG	*Dugesia spp.*	2
	Naididae	54
	Hydrachnida	4
	Caecidotea intermedius	17
	Hydroptila spp.	8
	Diamesinae	1
	Orthocladiinae	131
	Chironomini	3
	Tanytarsini	14
	Total	**234**
3B4SPG	*Dugesia spp.*	2
	Naididae	12
	Tubificidae	4
	Hydrachnida	5
	Caecidotea intermedius	59
	Crangonyx spp.	1
	Simulium spp.	2
	Hydroptila spp.	10
	Diamesinae	1
	Orthocladiinae	54
	Chironomini	5
	Tanytarsini	1
	Total	**156**

Sample	Taxon	Count
3B5SPG	*Dugesia spp.*	2
	Tubificidae	28
	Physella integra	1
	Sphaerium spp.	6
	Caecidotea intermedius	41
	Stenelmis spp.	4
	Optiocervus spp.	2
	Psephenus spp.	1
	Hydroptila spp.	5
	Orthocladiinae	11
	Chironomini	6
	Total	**107**
3B6SPG	*Dugesia spp.*	12
	Tubificidae	36
	Dina spp.	1
	Caecidotea intermedius	202
	Crangonyx spp.	3
	Optiocervus spp.	4
	Psephenus spp.	3
	Simulium spp.	2
	Hydroptila spp.	3
	Polycentropus spp.	1
	Pentaneurini	3
	Orthocladiinae	53
	Chironomini	4
	Tanytarsini	6
	Total	**333**

Sample	Taxon	Count
4A1SPG	*Nematoda*	3
	Sphaerium spp.	8
	Hydrachnida	1
	Crangonyx spp.	6
	Baetis spp.	20
	Stenelmis spp.	28
	Optiocervus spp.	28
	Psephenus spp.	1
	Simulium spp.	1
	Hemerodromia spp.	1
	Tipula spp.	1
	Glossosoma spp.	3
	Diamesinae	4
	Natarsia spp.	1
	Pentaneurini	1
	Orthocladiinae	25
	Chironomini	1
	Tanytarsini	2
	Total	**135**
4A2SPG	*Nematoda*	2
	Naididae	1
	Crangonyx spp.	1
	Baetis spp.	8
	Caenis spp.	14
	Leuctra spp.	1
	Nigronia spp.	1
	Stenelmis spp.	19
	Optiocervus spp.	15
	Simulium spp.	2
	Hydropsyche spp.	1
	Glossosoma spp.	8
	Hydroptila spp.	1
	Diamesinae	8
	Pentaneurini	1
	Orthocladiinae	76
	Tanytarsini	5
	Total	**164**

Sample	Taxon	Count
4A3SPG	Naididae	1
	Sphaerium spp.	3
	Baetis spp.	13
	Caenis spp.	13
	Paragnetina spp.	1
	Stenelmis spp.	16
	Optiocervus spp.	15
	Simulium spp.	1
	Hemerodromia spp.	1
	Hydropsyche spp.	4
	Chimarra spp.	1
	Hydroptila spp.	1
	Diamesinae	4
	Natarsia spp.	1
	Pentaneurini	1
	Orthocladiinae	83
	Tanytarsini	11
	Total	**170**
4A4SPG	*Nematoda*	1
	Hydrachnida	2
	Baetis spp.	20
	Paragnetina spp.	3
	Stenelmis spp.	33
	Optiocervus spp.	13
	Simulium spp.	2
	Prosimulium spp.	1
	Hydropsyche spp.	5
	Glossosoma spp.	1
	Chimarra spp.	1
	Diamesinae	3
	Orthocladiinae	81
	Tanytarsini	1
	Total	**167**

Sample	Taxon	Count
4A5SPG	*Dugesia spp.*	1
	Naididae	3
	Hydrachnida	6
	Baetis spp.	12
	Paragnetina spp.	6
	Stenelmis spp.	14
	Optiocervus spp.	8
	Prosimulium spp.	3
	Hydropsyche spp.	3
	Glossosoma spp.	1
	Hydroptila spp.	4
	Diamesinae	5
	Orthocladiinae	113
	Chironomini	3
	Tanytarsini	8
	Total	**190**
4A6SPG	Naididae	1
	Tubificidae	3
	Hydrachnida	6
	Caenis spp.	14
	Paragnetina spp.	1
	Leuctra spp.	1
	Nigronia spp.	1
	Stenelmis spp.	32
	Optiocervus spp.	15
	Prosimulium spp.	1
	Stratiomys spp.	1
	Hydropsyche spp.	7
	Glossosoma spp.	4
	Brachycentrus spp.	2
	Diamesinae	4
	Pentaneurini	1
	Orthocladiinae	41
	Tanytarsini	5
	Total	**140**

Sample	Taxon	Count
4B1SPG	*Dugesia spp.*	4
	Hydrachnida	2
	Caecidotea intermedius	2
	Ephemerella spp.	2
	Stenonema spp.	2
	Stenelmis spp.	19
	Optiocervus spp.	8
	Simulium spp.	63
	Cheumatopsyche spp.	133
	Chimarra spp.	4
	Orthocladiinae	60
	Chironomini	7
	Total	**306**
4B2SPG	*Dugesia spp.*	4
	Sphaerium spp.	4
	Caecidotea intermedius	1
	Crangonyx spp.	1
	Paraleptophlebia spp.	6
	Tricorythodes spp.	1
	Stenelmis spp.	18
	Optiocervus spp.	7
	Simulium spp.	5
	Cheumatopsyche spp.	68
	Chimarra spp.	2
	Diamesinae	6
	Pentaneurini	1
	Orthocladiinae	14
	Total	**138**
4B3SPG	*Dugesia spp.*	4
	Naididae	1
	Tubificidae	1
	Sphaerium spp.	5
	Hydrachnida	4
	Stenonema spp.	11
	Stenelmis spp.	29
	Optiocervus spp.	21
	Simulium spp.	13
	Tipula spp.	1
	Cheumatopsyche spp.	59
	Chimarra spp.	4
	Diamesinae	8
	Orthocladiinae	41
	Total	**202**

Sample	Taxon	Count
4B4SPG	*Dugesia spp.*	3
	Tubificidae	1
	Caecidotea intermedius	2
	Ephemerella spp.	11
	Paraleptophlebia spp.	1
	Stenelmis spp.	22
	Optiocervus spp.	16
	Simulium spp.	7
	Hemerodromia spp.	1
	Cheumatopsyche spp.	126
	Diamesinae	14
	Pentaneurini	3
	Orthocladiinae	18
	Chironomini	2
	Tanytarsini	1
	Total	**228**
4B5SPG	Tubificidae	1
	Sphaerium spp.	4
	Hydrachnida	1
	Caecidotea intermedius	3
	Ephemerella spp.	6
	Stenelmis spp.	6
	Optiocervus spp.	18
	Psephenus spp.	1
	Simulium spp.	4
	Cheumatopsyche spp.	17
	Chimarra spp.	10
	Diamesinae	1
	Orthocladiinae	9
	Tanytarsini	2
	Total	**83**

Sample	**Taxon**	**Count**
4B6SPG	*Caecidotea intermedius*	3
	Crangonyx spp.	2
	Ephemerella spp.	5
	Stenonema spp.	2
	Stenelmis spp.	5
	Optiocervus spp.	4
	Psephenus spp.	1
	Simulium spp.	6
	Tipula spp.	3
	Cheumatopsyche spp.	41
	Chimarra spp.	3
	Diamesinae	5
	Orthocladiinae	13
	Orthocladiinae	1
	Tanytarsini	2
	Total	**96**
5A1SPG	*Paraleptophlebia spp.*	2
	Stenelmis spp.	2
	Optiocervus spp.	2
	Simulium spp.	4
	Hemerodromia spp.	2
	Hydropsyche spp.	5
	Cheumatopsyche spp.	1
	Glossosoma spp.	12
	Orthocladiinae	62
	Chironomini	8
	Tanytarsini	81
	Total	**181**
5A2SPG	Sphaerium spp.	1
	Baetis spp.	1
	Stenelmis spp.	11
	Hemerodromia spp.	3
	Glossosoma spp.	7
	Pentaneurini	1
	Orthocladiinae	8
	Chironomini	20
	Tanytarsini	35
	Total	**87**

Sample	**Taxon**	**Count**
5A3SPG	Naididae	1
	Sphaerium spp.	10
	Stenonema spp.	1
	Baetis spp.	3
	Stenelmis spp.	9
	Simulium spp.	3
	Hemerodromia spp.	2
	Tipula spp.	3
	Hydropsyche spp.	3
	Cheumatopsyche spp.	2
	Glossosoma spp.	4
	Orthocladiinae	44
	Chironomini	101
	Tanytarsini	51
	Total	**237**
5A4SPG	Naididae	2
	Sphaerium spp.	5
	Stenelmis spp.	10
	Optiocervus spp.	5
	Petrophila spp.	4
	Hemerodromia spp.	4
	Atherix spp.	1
	Hydropsyche spp.	1
	Glossosoma spp.	6
	Pentaneurini	1
	Orthocladiinae	2
	Chironomini	34
	Tanytarsini	64
	Total	**139**
5A5SPG	Sphaerium spp.	3
	Stenonema spp.	1
	Baetis spp.	2
	Stenelmis spp.	5
	Optiocervus spp.	3
	Simulium spp.	38
	Hemerodromia spp.	2
	Hydropsyche spp.	30
	Glossosoma spp.	44
	Diamesinae	7
	Orthocladiinae	104
	Chironomini	6
	Tanytarsini	51
	Total	**296**

Sample	Taxon	Count
5A6SPG	Sphaerium spp.	4
	Stenonema spp.	2
	Baetis spp.	2
	Stenelmis spp.	3
	Optiocervus spp.	3
	Simulium spp.	3
	Hemerodromia spp.	1
	Atherix spp.	1
	Hydropsyche spp.	50
	Glossosoma spp.	86
	Chimarra spp.	1
	Pentaneurini	2
	Orthocladiinae	74
	Chironomini	3
	Tanytarsini	36
	Total	**271**
5B1SPG	*Dugesia spp.*	3
	Naididae	4
	Tubificidae	1
	Physella integra	1
	Valvata tricarinata	1
	Hydrachnida	1
	Optiocervus spp.	1
	Simulium spp.	22
	Cheumatopsyche spp.	7
	Pentaneurini	1
	Orthocladiinae	33
	Chironomini	41
	Tanytarsini	9
	Total	**125**
5B2SPG	*Nematoda*	1
	Dugesia spp.	13
	Naididae	2
	Hydrachnida	14
	Crangonyx spp.	1
	Simulium spp.	5
	Cheumatopsyche spp.	10
	Orthocladiinae	259
	Chironomini	48
	Tanytarsini	14
	Total	**367**

Sample	Taxon	Count
5B3SPG	*Nematoda*	1
	Dugesia spp.	68
	Naididae	2
	Dina spp.	1
	Sphaerium spp.	1
	Hydrachnida	2
	Stenelmis spp.	3
	Optiocervus spp.	2
	Simulium spp.	17
	Hemerodromia spp.	2
	Cheumatopsyche spp.	386
	Pentaneurini	3
	Orthocladiinae	108
	Chironomini	72
	Tanytarsini	42
	Total	**710**
5B4SPG	*Dugesia spp.*	12
	Naididae	4
	Sphaerium spp.	1
	Stenelmis spp.	1
	Simulium spp.	2
	Cheumatopsyche spp.	27
	Orthocladiinae	131
	Chironomini	35
	Tanytarsini	29
	Total	**242**
5B5SPG	*Dugesia spp.*	6
	Naididae	13
	Tubificidae	1
	Sphaerium spp.	2
	Hydrachnida	5
	Crangonyx spp.	9
	Baetis spp.	1
	Stenelmis spp.	5
	Simulium spp.	3
	Cheumatopsyche spp.	101
	Orthocladiinae	364
	Chironomini	135
	Tanytarsini	31
	Total	**676**

Sample	Taxon	Count
5B6SPG	*Dugesia spp.*	1
	Naididae	3
	Hydrachnida	3
	Stenelmis spp.	1
	Cheumatopsyche spp.	31
	Orthocladiinae	80
	Chironomini	122
	Tanytarsini	8
	Total	**249**

Sample	Taxon	Count
1A1SUM	Naididae	2
	Sphaerium spp.	1
	Hydrachnida	2
	Caecidotea intermedius	4
	Stenonema spp.	5
	Baetis spp.	11
	Paraleptophlebia spp.	15
	Caenis spp.	15
	Paragnetina spp.	1
	Leuctra spp.	1
	Stenelmis spp.	50
	Psephenus spp.	8
	Hemerodromia spp.	1
	Stratiomys spp.	1
	Hydropsyche spp.	12
	Glossosoma spp.	2
	Chimarra spp.	5
	Neophylax spp.	33
	Pentaneurini	22
	Orthocladiinae	7
	Pseudochironomus spp.	1
	Chironomini	10
	Tanytarsini	32
	Total	**241**
1A2SUM	*Dina spp.*	1
	Sphaerium spp.	19
	Caecidotea intermedius	1
	Serratella spp.	11
	Stenonema spp.	13
	Baetis spp.	33
	Leuctra spp.	1
	Argia spp.	2
	Stenelmis spp.	13
	Optiocervus spp.	10
	Psephenus spp.	28
	Hydropsyche spp.	13
	Chimarra spp.	9
	Neophylax spp.	12
	Brachycentrus spp.	3
	Pentaneurini	4
	Orthocladiinae	8
	Chironomini	17
	Tanytarsini	9
	Total	**207**

Sample	Taxon	Count
1A3SUM	Naididae	19
	Physella integra	1
	Stenonema spp.	2
	Baetis spp.	3
	Caenis spp.	1
	Paragnetina spp.	1
	Argia spp.	1
	Stenelmis spp.	14
	Optiocervus spp.	5
	Psephenus spp.	4
	Hydropsyche spp.	3
	Helicopsyche borealis	4
	Brachycentrus spp.	1
	Pentaneurini	7
	Orthocladiinae	4
	Chironomini	21
	Tanytarsini	13
	Total	**104**
1A4SUM	Naididae	6
	Sphaerium spp.	10
	Serratella spp.	3
	Stenonema spp.	21
	Baetis spp.	9
	Caenis spp.	8
	Leuctra spp.	1
	Argia spp.	3
	Stenelmis spp.	20
	Optiocervus spp.	1
	Psephenus spp.	4
	Stratiomys spp.	1
	Hydropsyche spp.	18
	Chimarra spp.	9
	Brachycentrus spp.	3
	Pentaneurini	8
	Orthocladiinae	1
	Chironomini	20
	Tanytarsini	16
	Total	**162**

Sample	Taxon	Count
1A5SUM	Naididae	3
	Dina spp.	1
	Caecidotea intermedius	1
	Serratella spp.	6
	Stenonema spp.	11
	Baetis spp.	16
	Leuctra spp.	1
	Argia spp.	2
	Stenelmis spp.	23
	Psephenus spp.	9
	Hydropsyche spp.	16
	Neophylax spp.	12
	Psilotreta spp.	1
	Pentaneurini	5
	Orthocladiinae	5
	Chironomini	20
	Tanytarsini	20
	Total	**152**
1A6SUM	Naididae	5
	Lumbricidae	1
	Sphaerium spp.	3
	Hydrachnida	1
	Serratella spp.	1
	Stenonema spp.	11
	Baetis spp.	13
	Caenis spp.	9
	Paragnetina spp.	1
	Leuctra spp.	1
	Argia spp.	3
	Stenelmis spp.	7
	Optiocervus spp.	1
	Psephenus spp.	1
	Hydropsyche spp.	10
	Chimarra spp.	9
	Neophylax spp.	28
	Brachycentrus spp.	2
	Pentaneurini	1
	Orthocladiinae	2
	Chironomini	13
	Tanytarsini	1
	Total	**124**

Sample	Taxon	Count
1B1SUM	*Dugesia spp.*	29
	Tubificidae	4
	Physella integra	4
	Hydrachnida	4
	Caecidotea intermedius	185
	Crangonyx spp.	8
	Caenis spp.	1
	Stenelmis spp.	1
	Hydropsyche spp.	10
	Cheumatopsyche spp.	20
	Hydroptila spp.	7
	Pentaneurini	4
	Orthocladiinae	23
	Chironomini	24
	Tanytarsini	5
	Total	**329**
1B2SUM	*Dugesia spp.*	25
	Tubificidae	3
	Physella integra	5
	Hydrachnida	3
	Caecidotea intermedius	461
	Crangonyx spp.	5
	Serratella spp.	15
	Stenonema spp.	9
	Stenelmis spp.	9
	Psephenus spp.	1
	Simulium spp.	1
	Hydropsyche spp.	8
	Cheumatopsyche spp.	27
	Hydroptila spp.	12
	Pentaneurini	2
	Orthocladiinae	6
	Chironomini	31
		623

Sample	Taxon	Count
1B3SUM	*Dugesia spp.*	11
	Tubificidae	26
	Physella integra	2
	Hydrachnida	4
	Caecidotea intermedius	311
	Crangonyx spp.	3
	Caenis spp.	3
	Cheumatopsyche spp.	3
	Brachycentrus spp.	1
	Pentaneurini	1
	Orthocladiinae	3
	Chironomini	20
	Tanytarsini	12
	Total	**400**
1B4SUM	*Dugesia spp.*	10
	Tubificidae	5
	Caecidotea intermedius	60
	Crangonyx spp.	5
	Helicopsyche borealis	2
	Brachycentrus spp.	1
	Orthocladiinae	2
	Chironomini	1
	Tanytarsini	4
	Total	**90**
1B5SUM	*Dugesia spp.*	11
	Tubificidae	15
	Hydrachnida	1
	Caecidotea intermedius	159
	Crangonyx spp.	12
	Hydropsyche spp.	5
	Cheumatopsyche spp.	6
	Orthocladiinae	8
	Chironomini	2
	Tanytarsini	1
	Total	**220**

Sample	Taxon	Count
1B6SUM	*Dugesia spp.*	11
	Tubificidae	8
	Hydrachnida	1
	Caecidotea intermedius	65
	Crangonyx spp.	14
	Hydropsyche spp.	2
	Cheumatopsyche spp.	2
	Glossosoma spp.	1
	Hydroptila spp.	2
	Chironomini	5
	Tanytarsini	5
	Total	**116**
2A1SUM	*Dugesia spp.*	19
	Tubificidae	1
	Sphaerium spp.	10
	Hydrachnida	7
	Serratella spp.	1
	Stenonema spp.	10
	Baetis spp.	45
	Caenis spp.	20
	Aeshna spp.	1
	Argia spp.	1
	Stenelmis spp.	35
	Optiocervus spp.	52
	Dubiraphia spp.	2
	Psephenus spp.	35
	Petrophila spp.	2
	Hydropsyche spp.	72
	Cheumatopsyche spp.	5
	Chimarra spp.	11
	Hydroptila spp.	1
	Brachycentrus spp.	6
	Pentaneurini	19
	Orthocladiinae	6
	Chironomini	22
	Tanytarsini	17
	Total	**400**

Sample	Taxon	Count
2A2SUM	*Dugesia spp.*	28
	Tubificidae	5
	Sphaerium spp.	26
	Hydrachnida	1
	Crangonyx spp.	1
	Serratella spp.	3
	Stenonema spp.	29
	Baetis spp.	71
	Caenis spp.	16
	Stenelmis spp.	88
	Optiocervus spp.	58
	Dubiraphia spp.	4
	Psephenus spp.	45
	Hemerodromia spp.	1
	Hydropsyche spp.	148
	Cheumatopsyche spp.	11
	Chimarra spp.	23
	Agraylea spp.	2
	Brachycentrus spp.	11
	Pentaneurini	27
	Orthocladiinae	10
	Chironomini	85
	Tanytarsini	13
	Total	**706**
2A3SUM	Sphaerium spp.	4
	Hydrachnida	16
	Stenonema spp.	3
	Baetis spp.	39
	Caenis spp.	95
	Stenelmis spp.	8
	Optiocervus spp.	3
	Dubiraphia spp.	5
	Psephenus spp.	15
	Hemerodromia spp.	1
	Hydropsyche spp.	23
	Neophylax spp.	3
	Helicopsyche borealis	4
	Brachycentrus spp.	32
	Pentaneurini	3
	Orthocladiinae	3
	Chironomini	39
	Tanytarsini	8
	Total	**304**

Sample	Taxon	Count
2A4SUM	*Dugesia spp.*	16
	Sphaerium spp.	4
	Hydrachnida	5
	Serratella spp.	1
	Stenonema spp.	3
	Baetis spp.	25
	Caenis spp.	32
	Optiocervus spp.	34
	Dubiraphia spp.	13
	Psephenus spp.	53
	Hydropsyche spp.	27
	Cheumatopsyche spp.	8
	Chimarra spp.	13
	Helicopsyche borealis	6
	Brachycentrus spp.	6
	Pentaneurini	7
	Orthocladiinae	1
	Chironomini	14
	Tanytarsini	7
	Total	**275**
2A5SUM	*Dugesia spp.*	5
	Sphaerium spp.	1
	Hydrachnida	1
	Crangonyx spp.	1
	Serratella spp.	1
	Baetis spp.	7
	Caenis spp.	41
	Stenelmis spp.	28
	Optiocervus spp.	19
	Dubiraphia spp.	2
	Psephenus spp.	26
	Hydropsyche spp.	5
	Cheumatopsyche spp.	1
	Helicopsyche borealis	1
	Brachycentrus spp.	4
	Pentaneurini	10
	Orthocladiinae	4
	Chironomini	22
	Tanytarsini	8
	Total	**187**

Sample	Taxon	Count
2A6SUM	*Dugesia spp.*	12
	Naididae	1
	Tubificidae	1
	Sphaerium spp.	2
	Hydrachnida	40
	Stenonema spp.	3
	Baetis spp.	19
	Caenis spp.	113
	Sialis spp	1
	Stenelmis spp.	38
	Optiocervus spp.	9
	Dubiraphia spp.	2
	Psephenus spp.	59
	Hydropsyche spp.	12
	Cheumatopsyche spp.	1
	Chimarra spp.	1
	Helicopsyche borealis	3
	Brachycentrus spp.	20
	Pentaneurini	13
	Orthocladiinae	5
	Chironomini	27
	Tanytarsini	16
	Total	**398**
2B1SUM	*Dugesia spp.*	64
	Tubificidae	22
	Hydrachnida	2
	Caecidotea intermedius	232
	Crangonyx spp.	11
	Serratella spp.	1
	Stenonema spp.	1
	Baetis spp.	3
	Caenis spp.	1
	Stenelmis spp.	1
	Optiocervus spp.	1
	Psephenus spp.	5
	Hemerodromia spp.	3
	Cheumatopsyche spp.	19
	Diamesinae	3
	Pentaneurini	17
	Orthocladiinae	8
	Chironomini	69
	Tanytarsini	6
	Total	**469**

Sample	Taxon	Count
2B2SUM	*Dugesia spp.*	41
	Tubificidae	27
	Valvata tricarinata	30
	Sphaerium spp.	30
	Caecidotea intermedius	118
	Crangonyx spp.	3
	Stenonema spp.	1
	Baetis spp.	7
	Caenis spp.	1
	Nigronia spp.	1
	Stenelmis spp.	78
	Psephenus spp.	3
	Cheumatopsyche spp.	9
	Chimarra spp.	7
	Brachycentrus spp.	1
	Pentaneurini	7
	Orthocladiinae	3
	Chironomini	15
	Total	**382**
2B3SUM	*Dugesia spp.*	32
	Tubificidae	94
	Valvata tricarinata	23
	Sphaerium spp.	23
	Caecidotea intermedius	145
	Orconectes spp.	2
	Serratella spp.	3
	Stenonema spp.	2
	Baetis spp.	6
	Stenelmis spp.	53
	Simulium spp.	33
	Cheumatopsyche spp.	2
	Chimarra spp.	4
	Pentaneurini	4
	Chironomini	35
	Total	**461**

Sample	Taxon	Count
2B4SUM	*Nematoda*	2
	Dugesia spp.	2
	Tubificidae	15
	Valvata tricarinata	8
	Sphaerium spp.	8
	Caecidotea intermedius	157
	Stenonema spp.	2
	Baetis spp.	10
	Caenis spp.	5
	Stenelmis spp.	28
	Optiocervus spp.	16
	Simulium spp.	3
	Tipula spp.	1
	Antocha spp.	1
	Cheumatopsyche spp.	44
	Chimarra spp.	16
	Helicopsyche borealis	3
	Pentaneurini	11
	Orthocladiinae	1
	Chironomini	36
	Total	**369**
2B5SUM	*Dugesia spp.*	21
	Tubificidae	26
	Valvata tricarinata	4
	Sphaerium spp.	4
	Hydrachnida	8
	Caecidotea intermedius	81
	Stenonema spp.	5
	Baetis spp.	6
	Stenelmis spp.	41
	Psephenus spp.	8
	Cheumatopsyche spp.	30
	Chimarra spp.	11
	Pentaneurini	8
	Orthocladiinae	22
	Chironomini	124
	Tanytarsini	10
	Total	**409**

Sample	Taxon	Count
2B6SUM	*Dugesia spp.*	57
	Tubificidae	8
	Caecidotea intermedius	62
	Serratella spp.	1
	Stenonema spp.	1
	Baetis spp.	4
	Caenis spp.	4
	Stenelmis spp.	44
	Cheumatopsyche spp.	49
	Chimarra spp.	16
	Diamesinae	3
	Pentaneurini	5
	Orthocladiinae	8
	Chironomini	51
	Tanytarsini	4
	Total	**317**
3A1SUM	*Nematoda*	8
	Dugesia spp.	2
	Sphaerium spp.	1
	Hydrachnida	6
	Caecidotea intermedius	1
	Stenonema spp.	23
	Baetis spp.	42
	Nigronia spp.	1
	Stenelmis spp.	18
	Optiocervus spp.	11
	Psephenus spp.	5
	Antocha spp.	1
	Hydropsyche spp.	10
	Glossosoma spp.	1
	Hydroptila spp.	1
	Agraylea spp.	4
	Pentaneurini	7
	Orthocladiinae	9
	Chironomini	8
	Tanytarsini	14
	Total	**173**

Sample	Taxon	Count
3A2SUM	*Nematoda*	2
	Sphaerium spp.	1
	Hydrachnida	3
	Stenonema spp.	23
	Baetis spp.	5
	Ephemera spp.	1
	Nigronia spp.	2
	Stenelmis spp.	20
	Optiocervus spp.	7
	Psephenus spp.	2
	Hemerodromia spp.	1
	Hydropsyche spp.	5
	Glossosoma spp.	2
	Pentaneurini	4
	Orthocladiinae	7
	Chironomini	6
	Tanytarsini	18
	Total	**109**
3A3SUM	*Nematoda*	1
	Dugesia spp.	1
	Naididae	1
	Hydrachnida	1
	Crangonyx spp.	1
	Stenonema spp.	10
	Baetis spp.	61
	Stenelmis spp.	1
	Optiocervus spp.	1
	Glossosoma spp.	2
	Pycnopsyche spp.	1
	Helicopsyche borealis	1
	Pentaneurini	6
	Orthocladiinae	2
	Chironomini	22
	Tanytarsini	31
	Total	**143**

Sample	Taxon	Count
3A4SUM	*Dugesia spp.*	5
	Naididae	2
	Sphaerium spp.	1
	Hydrachnida	2
	Serratella spp.	9
	Stenonema spp.	13
	Baetis spp.	45
	Caenis spp.	34
	Ephemera spp.	3
	Argia spp.	1
	Nigronia spp.	3
	Stenelmis spp.	3
	Optiocervus spp.	1
	Psephenus spp.	3
	Hydropsyche spp.	5
	Pentaneurini	2
	Orthocladiinae	3
	Chironomini	24
	Tanytarsini	33
	Total	**192**
3A5SUM	*Nematoda*	2
	Dugesia spp.	5
	Naididae	2
	Hydrachnida	8
	Serratella spp.	5
	Stenonema spp.	3
	Baetis spp.	25
	Caenis spp.	16
	Nigronia spp.	1
	Stenelmis spp.	77
	Psephenus spp.	1
	Hydropsyche spp.	5
	Ryacophila spp.	1
	Helicopsyche borealis	2
	Brachycentrus spp.	1
	Orthocladiinae	4
	Chironomini	10
	Tanytarsini	9
	Total	**177**

Sample	Taxon	Count
3A6SUM	*Dugesia spp.*	1
	Hydrachnida	7
	Stenonema spp.	8
	Baetis spp.	115
	Caenis spp.	8
	Aeshna spp.	1
	Stenelmis spp.	63
	Optiocervus spp.	16
	Psephenus spp.	5
	Simulium spp.	2
	Prosimulium spp.	1
	Hemerodromia spp.	1
	Hydropsyche spp.	9
	Agraylea spp.	5
	Helicopsyche borealis	8
	Pentaneurini	7
	Orthocladiinae	7
	Chironomini	32
	Tanytarsini	31
	Total	**327**
3B1SUM	*Dugesia spp.*	33
	Tubificidae	5
	Hydrachnida	3
	Caecidotea intermedius	285
	Crangonyx spp.	1
	Serratella spp.	1
	Stenelmis spp.	22
	Simulium spp.	1
	Hemerodromia spp.	1
	Hydropsyche spp.	3
	Cheumatopsyche spp.	3
	Chimarra spp.	1
	Pentaneurini	6
	Orthocladiinae	9
	Chironomini	19
		393

Sample	Taxon	Count
3B2SUM	*Dugesia spp.*	70
	Tubificidae	21
	Sphaerium spp.	11
	Hydrachnida	3
	Caecidotea intermedius	142
	Crangonyx spp.	4
	Baetis spp.	1
	Stenelmis spp.	8
	Cheumatopsyche spp.	1
	Diamesinae	1
	Pentaneurini	7
	Orthocladiinae	5
	Chironomini	42
	Total	**316**
3B3SUM	*Dugesia spp.*	28
	Tubificidae	67
	Dina spp.	1
	Sphaerium spp.	26
	Hydrachnida	1
	Caecidotea intermedius	153
	Crangonyx spp.	1
	Stenelmis spp.	9
	Simulium spp.	4
	Hemerodromia spp.	4
	Cheumatopsyche spp.	1
	Agraylea spp.	1
	Pentaneurini	3
	Orthocladiinae	1
	Chironomini	16
	Total	**316**
3B4SUM	*Dugesia spp.*	56
	Tubificidae	6
	Caecidotea intermedius	138
	Crangonyx spp.	2
	Stenelmis spp.	6
	Hydropsyche spp.	2
	Cheumatopsyche spp.	2
	Agraylea spp.	1
	Pentaneurini	5
	Orthocladiinae	4
	Chironomini	9
	Tanytarsini	1
	Total	**232**

Sample	Taxon	Count
3B5SUM	*Dugesia spp.*	9
	Tubificidae	7
	Physella integra	1
	Sphaerium spp.	2
	Hydrachnida	1
	Caecidotea intermedius	127
	Crangonyx spp.	1
	Baetis spp.	1
	Stenelmis spp.	3
	Simulium spp.	1
	Hemerodromia spp.	1
	Cheumatopsyche spp.	1
	Orthocladiinae	3
	Chironomini	2
	Total	**160**
3B6SUM	*Dugesia spp.*	37
	Tubificidae	50
	Sphaerium spp.	1
	Caecidotea intermedius	590
	Crangonyx spp.	1
	Baetis spp.	1
	Simulium spp.	1
	Hemerodromia spp.	1
	Cheumatopsyche spp.	1
	Agraylea spp.	1
	Pentaneurini	1
	Chironomini	1
	Total	**686**
4A1SUM	*Nematoda*	1
	Naididae	1
	Hydrachnida	1
	Caecidotea intermedius	1
	Caenis spp.	36
	Paragnetina spp.	1
	Stenelmis spp.	5
	Optiocervus spp.	9
	Hemerodromia spp.	3
	Hydropsyche spp.	15
	Cheumatopsyche spp.	3
	Glossosoma spp.	6
	Pentaneurini	1
	Orthocladiinae	7
	Chironomini	1
	Tanytarsini	166
	Total	**257**

Sample	Taxon	Count
4A2SUM	*Nematoda*	2
	Naididae	1
	Hydrachnida	15
	Serratella spp.	29
	Stenonema spp.	10
	Baetis spp.	17
	Isonychia spp.	13
	Stenelmis spp.	17
	Optiocervus spp.	29
	Psephenus spp.	2
	Hemerodromia spp.	5
	Hydropsyche spp.	90
	Cheumatopsyche spp.	15
	Glossosoma spp.	19
	Hydroptila spp.	8
	Ryacophila spp.	1
	Helicopsyche borealis	14
	Polycentropus spp.	6
	Brachycentrus spp.	7
	Pentaneurini	4
	Orthocladiinae	75
	Chironomini	39
	Tanytarsini	267
	Total	**685**
4A3SUM	*Nematoda*	10
	Naididae	10
	Hydrachnida	25
	Baetis spp.	28
	Caenis spp.	5
	Isonychia spp.	3
	Taeniopteryx spp.	1
	Stenelmis spp.	31
	Optiocervus spp.	37
	Psephenus spp.	1
	Hemerodromia spp.	1
	Bezzia spp.	1
	Hydropsyche spp.	8
	Glossosoma spp.	3
	Helicopsyche borealis	1
	Polycentropus spp.	2
	Diamesinae	1
	Pentaneurini	1
	Orthocladiinae	27
	Chironomini	5
	Tanytarsini	81
	Total	**282**

Sample	Taxon	Count
4A4SUM	Naididae	1
	Valvata tricarinata	2
	Elimia acuta	4
	Sphaerium spp.	5
	Hydrachnida	18
	Crangonyx spp.	2
	Serratella spp.	35
	Baetis spp.	25
	Caenis spp.	59
	Stenelmis spp.	9
	Optiocervus spp.	13
	Hydropsyche spp.	18
	Cheumatopsyche spp.	1
	Glossosoma spp.	5
	Hydroptila spp.	2
	Polycentropus spp.	4
	Pentaneurini	25
	Orthocladiinae	13
	Chironomini	50
	Tanytarsini	623
	Total	**914**
4A5SUM	*Dugesia spp.*	1
	Naididae	12
	Valvata tricarinata	1
	Elimia acuta	2
	Hydrachnida	5
	Crangonyx spp.	1
	Baetis spp.	24
	Nigronia spp.	2
	Stenelmis spp.	5
	Optiocervus spp.	14
	Hemerodromia spp.	9
	Hydropsyche spp.	30
	Glossosoma spp.	6
	Hydroptila spp.	5
	Pycnopsyche spp.	2
	Polycentropus spp.	2
	Diamesinae	1
	Pentaneurini	2
	Orthocladiinae	21
	Chironomini	1
	Tanytarsini	34
	Total	**180**

Sample	Taxon	Count
4A6SUM	Naididae	20
	Valvata tricarinata	3
	Elimia acuta	1
	Sphaerium spp.	3
	Hydrachnida	16
	Crangonyx spp.	5
	Serratella spp.	47
	Caenis spp.	20
	Stenelmis spp.	50
	Optiocervus spp.	27
	Hydropsyche spp.	5
	Cheumatopsyche spp.	1
	Glossosoma spp.	2
	Helicopsyche borealis	4
	Polycentropus spp.	2
	Brachycentrus spp.	2
	Pentaneurini	8
	Orthocladiinae	8
	Chironomini	18
	Tanytarsini	213
	Total	**455**
4B1SUM	*Nematoda*	15
	Dugesia spp.	33
	Tubificidae	33
	Lumbricidae	1
	Elimia acuta	2
	Sphaerium spp.	13
	Hydrachnida	2
	Caecidotea intermedius	9
	Serratella spp.	26
	Stenonema spp.	17
	Baetis spp.	7
	Caenis spp.	4
	Argia spp.	2
	Stenelmis spp.	56
	Optiocervus spp.	93
	Simulium spp.	7
	Antocha spp.	2
	Cheumatopsyche spp.	98
	Chimarra spp.	85
	Dolophilodes spp.	11
	Pentaneurini	7
	Orthocladiinae	30
	Chironomini	2
	Tanytarsini	2
	Total	**557**

Sample	Taxon	Count
4B2SUM	*Dugesia spp.*	66
	Naididae	34
	Sphaerium spp.	18
	Caecidotea intermedius	5
	Serratella spp.	6
	Stenonema spp.	21
	Caenis spp.	9
	Stenelmis spp.	84
	Optiocervus spp.	62
	Psephenus spp.	9
	Hemerodromia spp.	5
	Cheumatopsyche spp.	104
	Chimarra spp.	87
	Dolophilodes spp.	4
	Pentaneurini	8
	Orthocladiinae	5
	Total	**527**
4B3SUM	*Dugesia spp.*	42
	Naididae	48
	Sphaerium spp.	34
	Hydrachnida	14
	Crangonyx spp.	14
	Serratella spp.	12
	Stenonema spp.	6
	Baetis spp.	9
	Stenelmis spp.	105
	Optiocervus spp.	102
	Simulium spp.	4
	Cheumatopsyche spp.	61
	Chimarra spp.	40
	Brachycentrus spp.	5
	Pentaneurini	4
	Orthocladiinae	32
	Total	**532**

Sample	Taxon	Count
4B4SUM	*Dugesia spp.*	30
	Dina spp.	34
	Sphaerium spp.	26
	Hydrachnida	12
	Caecidotea intermedius	83
	Serratella spp.	6
	Stenonema spp.	8
	Stenelmis spp.	158
	Optiocervus spp.	70
	Simulium spp.	4
	Cheumatopsyche spp.	48
	Chimarra spp.	28
	Dolophilodes spp.	8
	Pentaneurini	8
	Orthocladiinae	8
	Chironomini	4
	Total	**535**
4B5SUM	*Nematoda*	75
	Dugesia spp.	57
	Naididae	43
	Sphaerium spp.	113
	Caecidotea intermedius	38
	Caenis spp.	33
	Stenelmis spp.	132
	Cheumatopsyche spp.	19
	Pentaneurini	18
	Total	**528**
4B6SUM	*Nematoda*	50
	Dugesia spp.	35
	Naididae	22
	Lumbricidae	5
	Sphaerium spp.	34
	Hydrachnida	5
	Caecidotea intermedius	30
	Crangonyx spp.	5
	Serratella spp.	21
	Stenonema spp.	54
	Stenelmis spp.	89
	Optiocervus spp.	24
	Psephenus spp.	5
	Cheumatopsyche spp.	59
	Chimarra spp.	49
	Dolophilodes spp.	10
	Pentaneurini	5
	Total	**502**

Sample	Taxon	Count
5A1SUM	Naididae	1
	Tubificidae	4
	Sphaerium spp.	4
	Hydrachnida	1
	Baetis spp.	6
	Nigronia spp.	5
	Stenelmis spp.	3
	Optiocervus spp.	6
	Hemerodromia spp.	8
	Antocha spp.	1
	Hydropsyche spp.	83
	Glossosoma spp.	9
	Helicopsyche borealis	9
	Polycentropus spp.	9
	Brachycentrus spp.	6
	Pentaneurini	12
	Orthocladiinae	30
	Chironomini	3
	Tanytarsini	219
	Total	**419**
5A2SUM	*Nematoda*	1
	Tubificidae	1
	Sphaerium spp.	19
	Hydrachnida	3
	Stenonema spp.	2
	Baetis spp.	27
	Nigronia spp.	3
	Stenelmis spp.	21
	Optiocervus spp.	33
	Hemerodromia spp.	19
	Hydropsyche spp.	36
	Glossosoma spp.	1
	Ryacophila spp.	1
	Helicopsyche borealis	4
	Polycentropus spp.	2
	Brachycentrus spp.	3
	Natarsia spp.	3
	Orthocladiinae	11
	Chironomini	8
	Tanytarsini	105
	Total	**303**

Sample	Taxon	Count
5A3SUM	Tubificidae	2
	Sphaerium spp.	8
	Baetis spp.	6
	Nigronia spp.	1
	Stenelmis spp.	5
	Antocha spp.	1
	Hydropsyche spp.	65
	Glossosoma spp.	17
	Helicopsyche borealis	8
	Brachycentrus spp.	4
	Pentaneurini	4
	Orthocladiinae	6
	Chironomini	2
	Tanytarsini	160
	Total	**289**
5A4SUM	Naididae	1
	Sphaerium spp.	13
	Hydrachnida	2
	Stenonema spp.	4
	Baetis spp.	13
	Nigronia spp.	9
	Stenelmis spp.	10
	Simulium spp.	2
	Hemerodromia spp.	3
	Stratiomys spp.	2
	Hydropsyche spp.	55
	Glossosoma spp.	16
	Helicopsyche borealis	3
	Polycentropus spp.	8
	Brachycentrus spp.	5
	Orthocladiinae	1
	Chironomini	1
	Tanytarsini	130
	Total	**278**

Sample	Taxon	Count
5A5SUM	Tubificidae	3
	Sphaerium spp.	17
	Stenonema spp.	2
	Baetis spp.	5
	Nigronia spp.	40
	Stenelmis spp.	22
	Simulium spp.	3
	Hemerodromia spp.	6
	Hydropsyche spp.	10
	Helicopsyche borealis	2
	Diamesinae	2
	Natarsia spp.	4
	Orthocladiinae	6
	Chironomini	20
	Tanytarsini	158
	Total	**300**
5A6SUM	Sphaerium spp.	8
	Stenonema spp.	1
	Baetis spp.	2
	Nigronia spp.	19
	Stenelmis spp.	13
	Hemerodromia spp.	5
	Stratiomys spp.	1
	Hydropsyche spp.	70
	Glossosoma spp.	10
	Brachycentrus spp.	1
	Pentaneurini	7
	Orthocladiinae	11
	Tanytarsini	85
	Total	**233**
5B1SUM	*Dugesia spp.*	14
	Tubificidae	54
	Dina spp.	3
	Sphaerium spp.	3
	Stenelmis spp.	6
	Hemerodromia spp.	3
	Cheumatopsyche spp.	37
	Orthocladiinae	48
	Chironomini	71
	Tanytarsini	164
	Total	**403**

Sample	Taxon	Count
5B2SUM	*Nematoda*	2
	Dugesia spp.	22
	Tubificidae	92
	Hydrachnida	8
	Caenis spp.	1
	Stenelmis spp.	1
	Hemerodromia spp.	1
	Cheumatopsyche spp.	15
	Glossosoma spp.	1
	Hydroptila spp.	1
	Orthocladiinae	11
	Chironomini	203
	Tanytarsini	121
	Total	**479**
5B3SUM	*Dugesia spp.*	85
	Tubificidae	82
	Hemerodromia spp.	2
	Cheumatopsyche spp.	21
	Orthocladiinae	15
	Pseudochironomus spp.	2
	Chironomini	117
	Tanytarsini	63
	Total	**387**
5B4SUM	Tubificidae	154
	Dina spp.	36
	Sphaerium spp.	26
	Orconectes spp.	5
	Cheumatopsyche spp.	10
	Pentaneurini	5
	Orthocladiinae	5
	Chironomini	77
	Tanytarsini	56
	Total	**374**

Sample	Taxon	Count
5B5SUM	*Nematoda*	7
	Dugesia spp.	2
	Tubificidae	14
	Dina spp.	1
	Sphaerium spp.	1
	Hydrachnida	1
	Caenis spp.	4
	Stenelmis spp.	4
	Cheumatopsyche spp.	6
	Pentaneurini	3
	Orthocladiinae	13
	Pseudochironomus spp.	3
	Chironomini	163
	Tanytarsini	135
	Total	**357**
5B6SUM	*Dugesia spp.*	8
	Tubificidae	40
	Sphaerium spp.	2
	Hydrachnida	17
	Caenis spp.	3
	Stenelmis spp.	5
	Cheumatopsyche spp.	7
	Helicopsyche borealis	1
	Brachycentrus spp.	1
	Orthocladiinae	9
	Pseudochironomus spp.	3
	Chironomini	99
	Tanytarsini	212
	Total	**407**

Sample	Taxon	Count
1A1FALL	*Dugesia spp.*	5
	Sphaerium spp.	16
	Hydrachnida	5
	Serratella spp.	30
	Stenonema spp.	1
	Baetis spp.	3
	Paraleptophlebia spp.	6
	Paragnetina spp.	6
	Stenelmis spp.	13
	Psephenus spp.	2
	Hydropsyche spp.	33
	Glossosoma spp.	91
	Chimarra spp.	13
	Neophylax spp.	3
	Pentaneurini	10
	Orthocladiinae	18
	Tanytarsini	5
	Total	**260**
1A2FALL	*Dugesia spp.*	2
	Sphaerium spp.	4
	Hydrachnida	5
	Serratella spp.	16
	Stenonema spp.	4
	Baetis spp.	3
	Paraleptophlebia spp.	4
	Caenis spp.	7
	Paragnetina spp.	3
	Leuctra spp.	5
	Stenelmis spp.	26
	Optiocervus spp.	9
	Psephenus spp.	3
	Hydropsyche spp.	33
	Cheumatopsyche spp.	9
	Chimarra spp.	2
	Pentaneurini	16
	Orthocladiinae	18
	Chironomini	18
	Tanytarsini	11
	Total	**198**

Sample	Taxon	Count
1A3FALL	*Dugesia spp.*	41
	Sphaerium spp.	7
	Hydrachnida	2
	Serratella spp.	34
	Stenonema spp.	2
	Stenelmis spp.	2
	Optiocervus spp.	5
	Psephenus spp.	2
	Simulium spp.	2
	Hydropsyche spp.	55
	Cheumatopsyche spp.	3
	Chimarra spp.	26
	Pentaneurini	7
	Orthocladiinae	28
	Chironomini	12
	Tanytarsini	9
	Total	**237**
1A4FALL	Sphaerium spp.	5
	Hydrachnida	2
	Crangonyx spp.	2
	Serratella spp.	16
	Stenonema spp.	4
	Baetis spp.	4
	Paragnetina spp.	2
	Simulium spp.	2
	Hydropsyche spp.	39
	Cheumatopsyche spp.	19
	Orthocladiinae	123
	Chironomini	2
	Tanytarsini	12
	Total	**232**

Sample	Taxon	Count
1A5FALL	Naididae	26
	Physella integra	3
	Sphaerium spp.	17
	Serratella spp.	50
	Caenis spp.	13
	Stenelmis spp.	7
	Psephenus spp.	13
	Glossosoma spp.	13
	Limnephilus spp.	10
	Helicopsyche borealis	26
	Pentaneurini	7
	Orthocladiinae	13
	Chironomini	17
	Tanytarsini	3
	Total	**218**
1A6FALL	*Dugesia spp.*	3
	Hydrachnida	16
	Serratella spp.	63
	Stenonema spp.	4
	Baetis spp.	8
	Stenelmis spp.	8
	Psephenus spp.	32
	Neophylax spp.	32
	Pentaneurini	8
	Orthocladiinae	20
	Chironomini	8
	Total	**202**
1B1FALL	*Dugesia spp.*	2
	Naididae	8
	Physella integra	3
	Caecidotea intermedius	8
	Stenelmis spp.	2
	Simulium spp.	1
	Cheumatopsyche spp.	1
	Orthocladiinae	13
	Chironomini	1
	Tanytarsini	5
	Total	**44**

Sample	Taxon	Count
1B2FALL	*Dugesia spp.*	5
	Naididae	3
	Tubificidae	3
	Physella integra	4
	Caecidotea intermedius	13
	Crangonyx spp.	3
	Stenelmis spp.	1
	Hydropsyche spp.	1
	Cheumatopsyche spp.	1
	Helicopsyche borealis	5
	Brachycentrus spp.	1
	Diamesinae	3
	Orthocladiinae	2
	Chironomini	5
	Total	**50**
1B3FALL	*Dugesia spp.*	6
	Tubificidae	14
	Physella integra	5
	Caecidotea intermedius	35
	Crangonyx spp.	4
	Caenis spp.	3
	Stenelmis spp.	3
	Simulium spp.	1
	Hydropsyche spp.	4
	Cheumatopsyche spp.	7
	Brachycentrus spp.	1
	Diamesinae	1
	Pentaneurini	5
	Orthocladiinae	26
	Chironomini	12
	Tanytarsini	20
	Total	**147**

Sample	Taxon	Count
1B4FALL	*Dugesia spp.*	8
	Tubificidae	4
	Physella integra	7
	Hydrachnida	1
	Caecidotea intermedius	26
	Crangonyx spp.	3
	Caenis spp.	2
	Stenelmis spp.	1
	Simulium spp.	1
	Hemerodromia spp.	1
	Antocha spp.	1
	Hydropsyche spp.	2
	Cheumatopsyche spp.	8
	Helicopsyche borealis	6
	Diamesinae	14
	Pentaneurini	5
	Orthocladiinae	12
	Chironomini	2
	Tanytarsini	10
	Total	**114**
1B5FALL	Naididae	1
	Physella integra	1
	Hydrachnida	1
	Caecidotea intermedius	4
	Simulium spp.	1
	Hemerodromia spp.	2
	Antocha spp.	3
	Cheumatopsyche spp.	1
	Diamesinae	2
	Orthocladiinae	1
	Tanytarsini	2
	Total	**19**

Sample	Taxon	Count
1B6FALL	*Dugesia spp.*	1
	Tubificidae	51
	Physella integra	1
	Sphaerium spp.	2
	Caecidotea intermedius	28
	Crangonyx spp.	3
	Stenelmis spp.	1
	Simulium spp.	1
	Hemerodromia spp.	1
	Brachycentrus spp.	2
	Diamesinae	1
	Orthocladiinae	5
	Chironomini	6
	Tanytarsini	8
	Total	**111**
2A1FALL	*Dugesia spp.*	2
	Naididae	2
	Physella integra	1
	Sphaerium spp.	2
	Hydrachnida	5
	Crangonyx spp.	1
	Stenonema spp.	2
	Baetis spp.	16
	Caenis spp.	30
	Leuctra spp.	3
	Stenelmis spp.	4
	Optiocervus spp.	1
	Dubiraphia spp.	1
	Psephenus spp.	4
	Prosimulium spp.	7
	Hydropsyche spp.	13
	Glossosoma spp.	4
	Chimarra spp.	1
	Helicopsyche borealis	1
	Pentaneurini	14
	Orthocladiinae	375
	Chironomini	46
	Tanytarsini	46
	Total	**581**

Sample	Taxon	Count
2A2FALL	*Dugesia spp.*	4
	Physella integra	1
	Sphaerium spp.	1
	Hydrachnida	2
	Orconectes spp.	1
	Stenonema spp.	2
	Baetis spp.	8
	Caenis spp.	24
	Leuctra spp.	2
	Stenelmis spp.	15
	Optiocervus spp.	4
	Psephenus spp.	5
	Simulium spp.	3
	Hydropsyche spp.	22
	Glossosoma spp.	8
	Chimarra spp.	6
	Pentaneurini	7
	Orthocladiinae	61
	Chironomini	21
	Tanytarsini	30
	Total	**227**
2A3FALL	Lumbricidae	1
	Sphaerium spp.	3
	Hydrachnida	3
	Baetis spp.	4
	Caenis spp.	21
	Leuctra spp.	2
	Taeniopteryx spp.	1
	Stenelmis spp.	2
	Optiocervus spp.	1
	Simulium spp.	1
	Prosimulium spp.	8
	Hydropsyche spp.	30
	Glossosoma spp.	7
	Orthocladiinae	252
	Chironomini	38
	Tanytarsini	22
	Total	**396**

Sample	Taxon	Count
2A4FALL	*Nematoda*	1
	Dugesia spp.	3
	Sphaerium spp.	4
	Stenonema spp.	2
	Baetis spp.	15
	Caenis spp.	11
	Leuctra spp.	2
	Optiocervus spp.	2
	Psephenus spp.	1
	Simulium spp.	1
	Prosimulium spp.	3
	Hydropsyche spp.	28
	Glossosoma spp.	15
	Chimarra spp.	3
	Pentaneurini	3
	Orthocladiinae	232
	Chironomini	26
	Tanytarsini	59
	Total	**411**
2A5FALL	*Dugesia spp.*	4
	Tubificidae	1
	Physella integra	2
	Serratella spp.	14
	Stenonema spp.	2
	Baetis spp.	6
	Caenis spp.	18
	Leuctra spp.	2
	Stenelmis spp.	9
	Optiocervus spp.	4
	Psephenus spp.	6
	Hydropsyche spp.	42
	Cheumatopsyche spp.	1
	Glossosoma spp.	17
	Chimarra spp.	14
	Helicopsyche borealis	7
	Pentaneurini	10
	Orthocladiinae	54
	Chironomini	28
	Tanytarsini	18
	Total	**259**

Sample	Taxon	Count
2A6FALL	*Dugesia spp.*	1
	Tubificidae	2
	Physella integra	2
	Sphaerium spp.	5
	Hydrachnida	2
	Crangonyx spp.	1
	Baetis spp.	14
	Caenis spp.	65
	Stenelmis spp.	10
	Optiocervus spp.	5
	Dubiraphia spp.	1
	Psephenus spp.	3
	Hydropsyche spp.	10
	Glossosoma spp.	1
	Chimarra spp.	11
	Helicopsyche borealis	10
	Pentaneurini	34
	Orthocladiinae	208
	Chironomini	81
	Tanytarsini	102
	Total	**568**
2B1FALL	*Dugesia spp.*	1
	Tubificidae	1
	Dina spp.	1
	Physella integra	1
	Sphaerium spp.	1
	Caecidotea intermedius	6
	Serratella spp.	2
	Stenelmis spp.	1
	Simulium spp.	4
	Cheumatopsyche spp.	15
	Chimarra spp.	21
	Orthocladiinae	2
	Chironomini	6
	Total	**62**
2B2FALL	Tubificidae	13
	Caecidotea intermedius	9
	Baetis spp.	2
	Stenelmis spp.	2
	Simulium spp.	5
	Cheumatopsyche spp.	5
	Chimarra spp.	23
	Orthocladiinae	3
	Chironomini	3
	Tanytarsini	1
	Total	**66**

Sample	Taxon	Count
2B3FALL	*Nematoda*	1
	Dugesia spp.	5
	Tubificidae	1
	Sphaerium spp.	8
	Hydrachnida	2
	Caecidotea intermedius	54
	Ephemerella spp.	1
	Serratella spp.	2
	Baetis spp.	3
	Stenelmis spp.	50
	Psephenus spp.	2
	Simulium spp.	1
	Hydropsyche spp.	3
	Cheumatopsyche spp.	17
	Glossosoma spp.	15
	Chimarra spp.	8
	Neophylax spp.	2
	Pentaneurini	9
	Orthocladiinae	19
	Chironomini	29
	Tanytarsini	13
	Total	**245**
2B4FALL	*Dugesia spp.*	2
	Tubificidae	1
	Physella integra	1
	Sphaerium spp.	2
	Caecidotea intermedius	20
	Stenonema spp.	1
	Baetis spp.	2
	Stenelmis spp.	6
	Simulium spp.	6
	Tipula spp.	1
	Hydropsyche spp.	2
	Cheumatopsyche spp.	8
	Glossosoma spp.	1
	Chimarra spp.	23
	Pentaneurini	2
	Orthocladiinae	12
	Chironomini	6
	Tanytarsini	3
	Total	**99**

Sample	Taxon	Count
2B5FALL	*Dugesia spp.*	2
	Tubificidae	2
	Physella integra	1
	Caecidotea intermedius	10
	Stenonema spp.	2
	Tricorythodes spp.	2
	Leuctra spp.	3
	Stenelmis spp.	3
	Psephenus spp.	2
	Simulium spp.	3
	Hydropsyche spp.	1
	Cheumatopsyche spp.	4
	Chimarra spp.	4
	Orthocladiinae	27
	Chironomini	4
	Tanytarsini	2
	Total	**72**
2B6FALL	*Nematoda*	1
	Tubificidae	9
	Sphaerium spp.	2
	Caecidotea intermedius	10
	Stenelmis spp.	1
	Simulium spp.	3
	Cheumatopsyche spp.	2
	Glossosoma spp.	2
	Chimarra spp.	3
	Orthocladiinae	6
	Chironomini	21
	Total	**60**

Sample	Taxon	Count
3A1FALL	*Dugesia spp.*	2
	Hydrachnida	5
	Serratella spp.	13
	Baetis spp.	54
	Paraleptophlebia spp.	12
	Caenis spp.	12
	Isonychia spp.	3
	Leuctra spp.	49
	Stenelmis spp.	25
	Hemerodromia spp.	2
	Glossosoma spp.	2
	Chimarra spp.	2
	Pentaneurini	7
	Orthocladiinae	15
	Chironomini	8
	Tanytarsini	35
	Total	**246**
3A2FALL	*Dugesia spp.*	7
	Naididae	3
	Sphaerium spp.	7
	Hydrachnida	4
	Crangonyx spp.	1
	Serratella spp.	50
	Baetis spp.	113
	Caenis spp.	32
	Isonychia spp.	2
	Leuctra spp.	23
	Taeniopteryx spp.	2
	Aeshna spp.	2
	Stenelmis spp.	27
	Optiocervus spp.	9
	Dubiraphia spp.	5
	Psephenus spp.	1
	Hemerodromia spp.	2
	Tipula spp.	1
	Hydropsyche spp.	3
	Chimarra spp.	2
	Brachycentrus spp.	5
	Pentaneurini	8
	Orthocladiinae	15
	Pseudochironomus spp.	3
	Chironomini	14
	Tanytarsini	40
	Total	**381**

Sample	Taxon	Count
3A3FALL	*Dugesia spp.*	2
	Hydrachnida	2
	Serratella spp.	17
	Stenonema spp.	2
	Baetis spp.	65
	Paraleptophlebia spp.	10
	Caenis spp.	12
	Leuctra spp.	44
	Taeniopteryx spp.	3
	Stenelmis spp.	31
	Optiocervus spp.	9
	Hemerodromia spp.	3
	Hydropsyche spp.	5
	Glossosoma spp.	2
	Brachycentrus spp.	2
	Pentaneurini	5
	Orthocladiinae	16
	Chironomini	3
	Tanytarsini	7
	Total	**240**
3A4FALL	*Dugesia spp.*	2
	Naididae	20
	Tubificidae	3
	Sphaerium spp.	5
	Hydrachnida	3
	Crangonyx spp.	3
	Serratella spp.	7
	Stenonema spp.	3
	Baetis spp.	50
	Caenis spp.	25
	Leuctra spp.	65
	Taeniopteryx spp.	7
	Argia spp.	3
	Stenelmis spp.	30
	Hydropsyche spp.	7
	Glossosoma spp.	2
	Pentaneurini	10
	Orthocladiinae	30
	Chironomini	17
	Tanytarsini	42
	Total	**334**

Sample	Taxon	Count
3A5FALL	Tubificidae	1
	Sphaerium spp.	2
	Hydrachnida	1
	Orconectes spp.	2
	Serratella spp.	23
	Stenonema spp.	2
	Baetis spp.	27
	Paraleptophlebia spp.	2
	Caenis spp.	19
	Leuctra spp.	10
	Aeshna spp.	3
	Argia spp.	2
	Stenelmis spp.	23
	Optiocervus spp.	15
	Dubiraphia spp.	4
	Psephenus spp.	4
	Hemerodromia spp.	2
	Hydropsyche spp.	6
	Glossosoma spp.	8
	Pentaneurini	2
	Orthocladiinae	36
	Chironomini	6
	Tanytarsini	10
	Total	**210**
3A6FALL	*Dugesia spp.*	10
	Hydrachnida	2
	Serratella spp.	29
	Baetis spp.	16
	Paraleptophlebia spp.	24
	Caenis spp.	16
	Paragnetina spp.	2
	Leuctra spp.	2
	Aeshna spp.	2
	Nigronia spp.	2
	Stenelmis spp.	21
	Optiocervus spp.	8
	Dubiraphia spp.	2
	Psephenus spp.	2
	Tipula spp.	3
	Hydropsyche spp.	44
	Glossosoma spp.	6
	Chimarra spp.	11
	Pentaneurini	6
	Orthocladiinae	21
	Chironomini	21
	Tanytarsini	10
	Total	**260**

Sample	Taxon	Count
3B1FALL	*Dugesia spp.*	6
	Sphaerium spp.	1
	Caecidotea intermedius	19
	Baetis spp.	2
	Stenelmis spp.	1
	Cheumatopsyche spp.	3
	Helicopsyche borealis	1
	Brachycentrus spp.	1
	Orthocladiinae	1
	Chironomini	3
	Total	**38**
3B2FALL	*Dugesia spp.*	2
	Sphaerium spp.	32
	Hydrachnida	1
	Caecidotea intermedius	33
	Baetis spp.	2
	Stenelmis spp.	1
	Simulium spp.	5
	Cheumatopsyche spp.	3
	Agraylea spp.	1
	Pentaneurini	7
	Orthocladiinae	9
	Chironomini	37
	Tanytarsini	2
	Total	**135**
3B3FALL	*Dugesia spp.*	4
	Tubificidae	4
	Sphaerium spp.	2
	Caecidotea intermedius	21
	Crangonyx spp.	6
	Stenelmis spp.	2
	Simulium spp.	2
	Pentaneurini	6
	Orthocladiinae	17
	Chironomini	4
	Tanytarsini	3
	Total	**71**

Sample	Taxon	Count
3B4FALL	*Dugesia spp.*	5
	Tubificidae	2
	Elimia acuta	2
	Sphaerium spp.	4
	Caecidotea intermedius	12
	Baetis spp.	1
	Nigronia spp.	1
	Stenelmis spp.	1
	Diamesinae	1
	Pentaneurini	1
	Orthocladiinae	3
	Chironomini	9
	Total	**42**
3B5FALL	*Dugesia spp.*	11
	Tubificidae	24
	Dina spp.	1
	Sphaerium spp.	20
	Caecidotea intermedius	29
	Crangonyx spp.	1
	Chironomini	9
	Tanytarsini	1
	Total	**96**
3B6FALL	*Dugesia spp.*	10
	Tubificidae	5
	Sphaerium spp.	7
	Caecidotea intermedius	27
	Crangonyx spp.	4
	Baetis spp.	4
	Simulium spp.	2
	Agraylea spp.	1
	Pentaneurini	1
	Orthocladiinae	5
	Chironomini	1
	Tanytarsini	1
	Total	**68**

Sample	Taxon	Count
4A1FALL	Naididae	19
	Tubificidae	9
	Elimia acuta	5
	Sphaerium spp.	5
	Stenonema spp.	30
	Baetis spp.	23
	Caenis spp.	2
	Leuctra spp.	2
	Taeniopteryx spp.	6
	Stenelmis spp.	5
	Simulium spp.	2
	Hemerodromia spp.	2
	Hydropsyche spp.	60
	Cheumatopsyche spp.	2
	Hydroptila spp.	4
	Helicopsyche borealis	2
	Chironomini	6
	Tanytarsini	4
	Total	**188**
4A2FALL	Naididae	4
	Tubificidae	2
	Valvata tricarinata	2
	Elimia acuta	2
	Sphaerium spp.	3
	Hydrachnida	2
	Stenonema spp.	33
	Baetis spp.	25
	Caenis spp.	4
	Stenelmis spp.	2
	Optiocervus spp.	6
	Simulium spp.	4
	Hydropsyche spp.	35
	Cheumatopsyche spp.	6
	Hydroptila spp.	4
	Helicopsyche borealis	19
	Brachycentrus spp.	10
	Orthocladiinae	2
	Chironomini	2
	Tanytarsini	27
	Total	**194**

Sample	Taxon	Count
4A3FALL	*Elimia acuta*	2
	Stenonema spp.	15
	Baetis spp.	13
	Caenis spp.	7
	Leuctra spp.	3
	Stenelmis spp.	3
	Prosimulium spp.	2
	Hemerodromia spp.	3
	Hydropsyche spp.	129
	Cheumatopsyche spp.	55
	Hydroptila spp.	7
	Pentaneurini	5
	Orthocladiinae	2
	Chironomini	2
	Total	**248**
4A4FALL	*Elimia acuta*	2
	Stenonema spp.	30
	Baetis spp.	26
	Caenis spp.	3
	Leuctra spp.	3
	Optiocervus spp.	4
	Simulium spp.	2
	Hydropsyche spp.	104
	Cheumatopsyche spp.	12
	Hydroptila spp.	30
	Helicopsyche borealis	7
	Orthocladiinae	4
	Chironomini	2
	Tanytarsini	2
	Total	**231**

Sample	Taxon	Count
4A5FALL	*Nematoda*	10
	Naididae	20
	Hydrachnida	6
	Crangonyx spp.	2
	Stenonema spp.	5
	Baetis spp.	31
	Caenis spp.	2
	Stenelmis spp.	9
	Optiocervus spp.	9
	Simulium spp.	2
	Hydropsyche spp.	57
	Cheumatopsyche spp.	39
	Hydroptila spp.	7
	Brachycentrus spp.	2
	Diamesinae	2
	Natarsia spp.	2
	Pentaneurini	2
	Orthocladiinae	5
	Chironomini	4
	Tanytarsini	2
	Total	**218**
4A6FALL	Naididae	22
	Elimia acuta	2
	Hydrachnida	2
	Stenonema spp.	30
	Baetis spp.	18
	Caenis spp.	7
	Stenelmis spp.	7
	Optiocervus spp.	20
	Simulium spp.	2
	Hemerodromia spp.	3
	Hydropsyche spp.	114
	Cheumatopsyche spp.	8
	Hydroptila spp.	5
	Helicopsyche borealis	2
	Pentaneurini	2
	Orthocladiinae	3
	Tanytarsini	1
	Total	**248**

Sample	Taxon	Count
4B1FALL	*Dugesia spp.*	1
	Sphaerium spp.	1
	Caecidotea intermedius	9
	Crangonyx spp.	1
	Serratella spp.	6
	Stenonema spp.	1
	Taeniopteryx spp.	3
	Simulium spp.	1
	Prosimulium spp.	403
	Cheumatopsyche spp.	4
	Brachycentrus spp.	1
	Orthocladiinae	14
	Tanytarsini	2
	Total	**447**
4B2FALL	Sphaerium spp.	2
	Caecidotea intermedius	1
	Crangonyx spp.	1
	Serratella spp.	3
	Stenonema spp.	3
	Leuctra spp.	1
	Stenelmis spp.	3
	Prosimulium spp.	22
	Cheumatopsyche spp.	7
	Pentaneurini	4
	Orthocladiinae	30
	Chironomini	1
	Tanytarsini	5
	Total	**83**
4B3FALL	*Nematoda*	3
	Sphaerium spp.	13
	Hydrachnida	1
	Caecidotea intermedius	1
	Crangonyx spp.	1
	Serratella spp.	4
	Leuctra spp.	1
	Stenelmis spp.	4
	Simulium spp.	1
	Prosimulium spp.	136
	Cheumatopsyche spp.	5
	Diamesinae	1
	Pentaneurini	1
	Orthocladiinae	27
	Chironomini	3
	Tanytarsini	5
	Total	**207**

Sample	Taxon	Count
4B4FALL	*Nematoda*	3
	Dugesia spp.	1
	Sphaerium spp.	3
	Hydrachnida	2
	Caecidotea intermedius	1
	Crangonyx spp.	1
	Serratella spp.	3
	Stenelmis spp.	8
	Prosimulium spp.	5
	Cheumatopsyche spp.	4
	Chimarra spp.	2
	Pentaneurini	1
	Orthocladiinae	2
	Total	**36**
4B5FALL	*Nematoda*	3
	Dugesia spp.	2
	Sphaerium spp.	5
	Caecidotea intermedius	1
	Crangonyx spp.	1
	Serratella spp.	1
	Leuctra spp.	1
	Stenelmis spp.	18
	Prosimulium spp.	1
	Cheumatopsyche spp.	3
	Chimarra spp.	3
	Orthocladiinae	6
	Chironomini	2
	Total	**47**
4B6FALL	*Nematoda*	1
	Hydrachnida	1
	Caecidotea intermedius	1
	Crangonyx spp.	1
	Serratella spp.	5
	Taeniopteryx spp.	2
	Stenelmis spp.	3
	Optiocervus spp.	4
	Prosimulium spp.	152
	Hemerodromia spp.	1
	Cheumatopsyche spp.	34
	Chimarra spp.	2
	Orthocladiinae	19
	Chironomini	3
	Tanytarsini	1
	Total	**230**

Sample	Taxon	Count
5A1FALL	Naididae	1
	Sphaerium spp.	6
	Hydrachnida	6
	Stenonema spp.	18
	Baetis spp.	12
	Stenelmis spp.	30
	Optiocervus spp.	12
	Hemerodromia spp.	14
	Stratiomys spp.	1
	Hydropsyche spp.	103
	Glossosoma spp.	8
	Chimarra spp.	17
	Limnephilus spp.	5
	Polycentropus spp.	5
	Brachycentrus spp.	11
	Orthocladiinae	5
	Chironomini	1
	Tanytarsini	38
	Total	**293**
5A2FALL	Sphaerium spp.	17
	Hydrachnida	6
	Stenonema spp.	29
	Baetis spp.	9
	Aeshna spp.	4
	Stenelmis spp.	44
	Optiocervus spp.	6
	Simulium spp.	3
	Hemerodromia spp.	20
	Hydropsyche spp.	93
	Glossosoma spp.	20
	Limnephilus spp.	3
	Brachycentrus spp.	3
	Orthocladiinae	3
	Tanytarsini	44
	Total	**304**

Sample	Taxon	Count
5A3FALL	Sphaerium spp.	30
	Hydrachnida	2
	Stenonema spp.	28
	Baetis spp.	18
	Nigronia spp.	2
	Stenelmis spp.	35
	Optiocervus spp.	8
	Hemerodromia spp.	5
	Hydropsyche spp.	146
	Glossosoma spp.	8
	Brachycentrus spp.	25
	Tanytarsini	25
	Total	**332**
5A4FALL	Sphaerium spp.	42
	Hydrachnida	8
	Stenonema spp.	51
	Baetis spp.	25
	Stenelmis spp.	80
	Optiocervus spp.	21
	Simulium spp.	5
	Hemerodromia spp.	5
	Stratiomys spp.	5
	Hydropsyche spp.	93
	Glossosoma spp.	13
	Chimarra spp.	4
	Limnephilus spp.	4
	Helicopsyche borealis	4
	Pentaneurini	4
	Orthocladiinae	8
	Tanytarsini	21
	Total	**393**

Sample	Taxon	Count
5A5FALL	Naididae	9
	Sphaerium spp.	29
	Hydrachnida	2
	Stenonema spp.	22
	Baetis spp.	13
	Nigronia spp.	1
	Stenelmis spp.	23
	Optiocervus spp.	9
	Simulium spp.	2
	Hemerodromia spp.	2
	Hydropsyche spp.	123
	Glossosoma spp.	7
	Chimarra spp.	28
	Limnephilus spp.	3
	Brachycentrus spp.	4
	Natarsia spp.	2
	Pentaneurini	2
	Orthocladiinae	6
	Chironomini	4
	Tanytarsini	23
	Total	**314**
5A6FALL	Sphaerium spp.	72
	Hydrachnida	13
	Stenonema spp.	9
	Nigronia spp.	5
	Stenelmis spp.	18
	Optiocervus spp.	45
	Psephenus spp.	5
	Hemerodromia spp.	13
	Hydropsyche spp.	162
	Cheumatopsyche spp.	5
	Glossosoma spp.	9
	Helicopsyche borealis	13
	Pentaneurini	4
	Orthocladiinae	4
	Tanytarsini	9
	Total	**386**

Sample	Taxon	Count
5B1	*Dugesia spp.*	1
	Naididae	90
	Sphaerium spp.	1
	Hydrachnida	7
	Crangonyx spp.	13
	Caenis spp.	2
	Hemerodromia spp.	1
	Cheumatopsyche spp.	15
	Pentaneurini	1
	Orthocladiinae	1
	Chironomini	102
	Tanytarsini	72
	Total	**306**
5B2	*Dugesia spp.*	11
	Naididae	96
	Physella integra	1
	Hydrachnida	4
	Crangonyx spp.	7
	Caenis spp.	7
	Simulium spp.	4
	Hemerodromia spp.	8
	Cheumatopsyche spp.	19
	Pentaneurini	25
	Orthocladiinae	7
	Chironomini	155
	Tanytarsini	85
	Total	**429**
5B3	*Nematoda*	1
	Naididae	27
	Physella integra	1
	Hydrachnida	1
	Crangonyx spp.	6
	Caenis spp.	8
	Simulium spp.	7
	Hemerodromia spp.	3
	Bezzia spp.	1
	Cheumatopsyche spp.	25
	Chironomini	47
	Tanytarsini	186
	Total	**313**

Sample	Taxon	Count
3B4	*Dugesia spp.*	5
	Naididae	23
	Hydrachnida	1
	Crangonyx spp.	2
	Caenis spp.	1
	Simulium spp.	48
	Hemerodromia spp.	1
	Cheumatopsyche spp.	23
	Chimarra spp.	1
	Orthocladiinae	10
	Chironomini	64
	Tanytarsini	127
	Total	**306**
3B5	*Dugesia spp.*	2
	Naididae	47
	Physella integra	1
	Hydrachnida	1
	Crangonyx spp.	3
	Caenis spp.	1
	Simulium spp.	1
	Hemerodromia spp.	1
	Tipula spp.	1
	Cheumatopsyche spp.	14
	Orthocladiinae	8
	Chironomini	111
	Tanytarsini	59
	Total	**250**
3B6	*Dugesia spp.*	9
	Naididae	30
	Crangonyx spp.	8
	Caenis spp.	1
	Simulium spp.	8
	Cheumatopsyche spp.	4
	Orthocladiinae	2
	Chironomini	28
	Tanytarsini	95
	Total	**185**

Appendix C: Reservoir Storage Levels

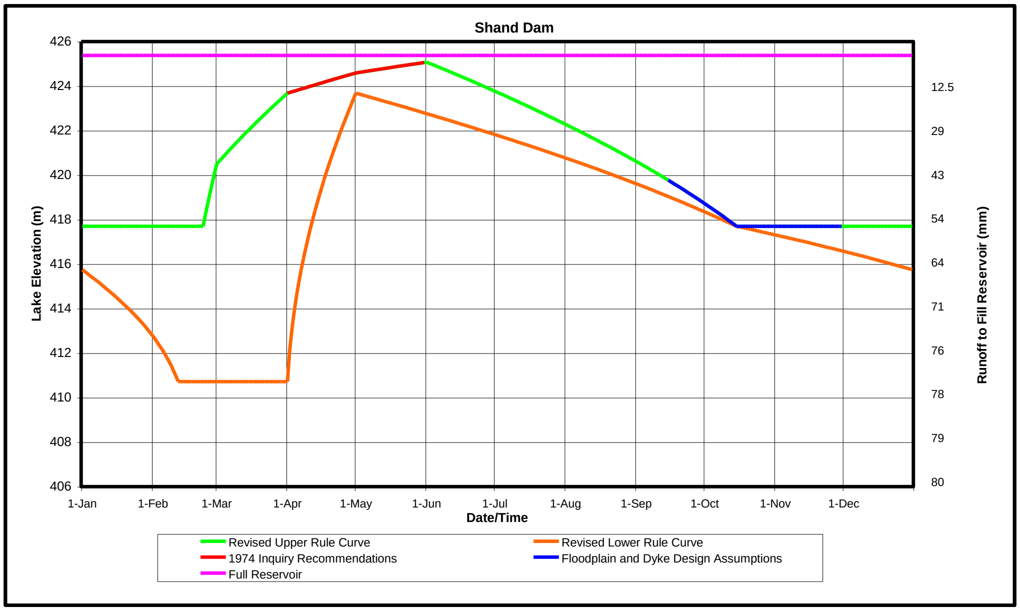

Figure A. 11. Shand Dam Annual Hydraulic Curves.

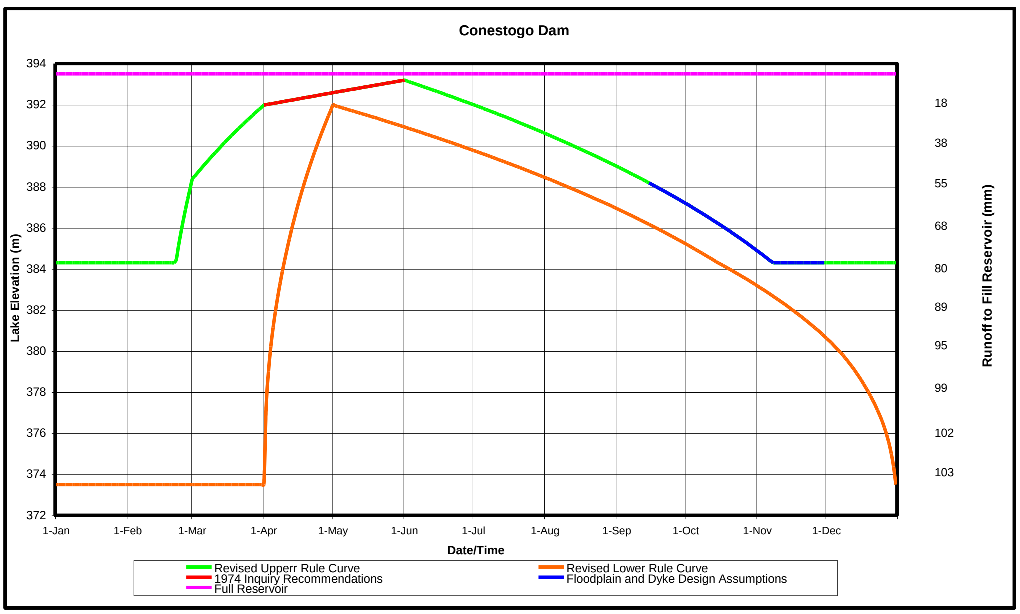

Figure A. 12. Conestogo Dam Annual Hydraulic Curves.

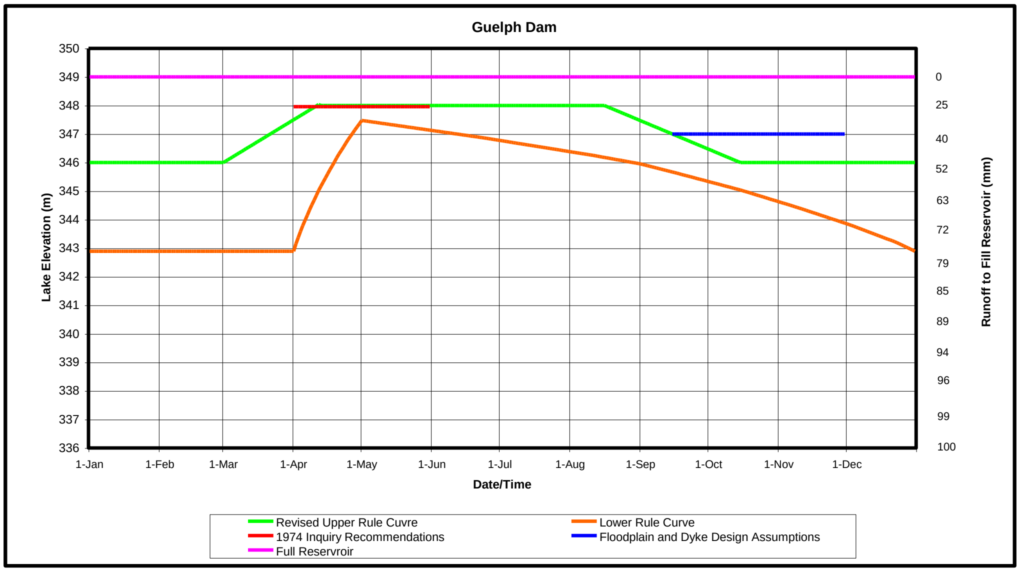

Figure A. 13. Guelph Dam Annual Hydraulic Curves.

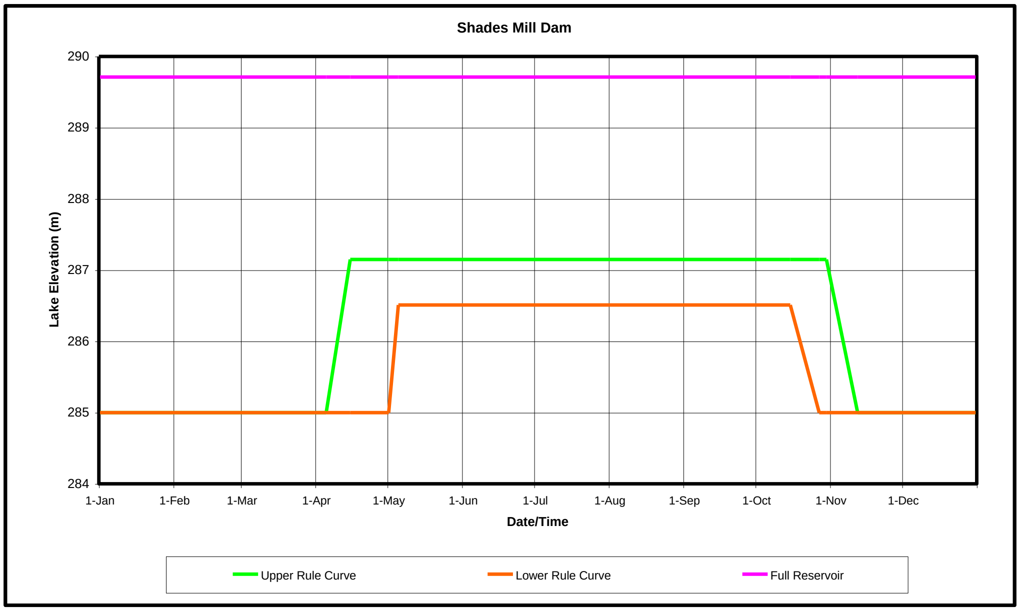

Figure A. 14. Shade's Mills Dam Annual Hydraulic Curves.

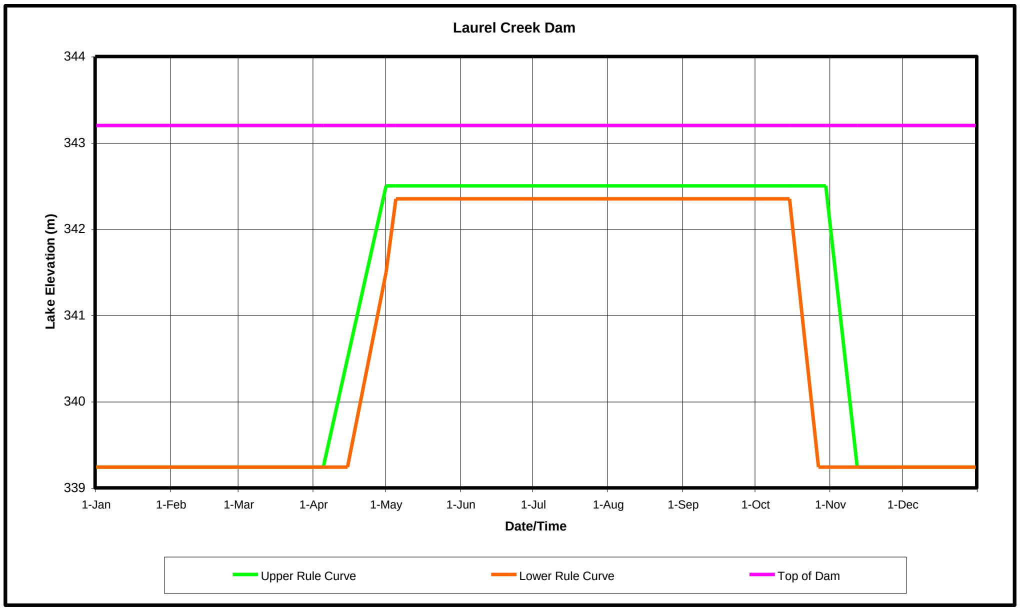

Figure A. 15. Laurel Creek Reservoir Annual Hydraulic Curves.