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Extent, Characteristics and Downstream Effects of Stream Enclosure in Southwestern Ontario

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EXTENT, CHARACTERISTICS AND DOWNSTREAM EFFECTS OF STREAM ENCLOSURES IN SOUTHWESTERN ONTARIO

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ABSTRACT AND KEYWORDS

In a 16000 km^2 area of southwestern Ontario, almost 15% of all streams have been enclosed (buried) largely for agricultural purposes. ArcGIS was used to characterize the natural features of catchments and to calculate enclosedness (proportion of stream network enclosed; $\bar{x}$ = 16.5%, n = 10106). Catchments with the highest enclosedness received >990 mm of precipitation annually and had high drainage density (>1.9 km/km^2), while catchments with the lowest enclosedness received <990 mm of precipitation annually, were characterized by clay and undrumlinized till plains and had relatively shallow water tables (<15 m). These natural features influence where enclosure is likely to occur, however, the final decision to enclose a stream may also be based on social, economic and political factors.

Effects of enclosedness on fish and benthic macroinvertebrate (BMI) assemblages were determined using multiple visit sampling (MVS) at 10 sites in the Ausable River Basin (ARB), and single visit sampling (SVS) at 157 sites across southwestern Ontario (SWO). Among SVS-SWO sites, enclosedness was positively correlated to BMI density (r = 0.195) and estimated abundance (r = 0.266) and the proportion of herbivorous/insectivorous fish species (r = 0.187), and negatively correlated to the proportion of insectivorous/piscivorous fish species (r = -0.167). There were no correlations with enclosedness among the MVS-ARB sites. However, fish species richness and abundance, and BMI richness, density and diversity were most variable with sampling date at these sites, suggesting that multiple visit sampling may provide a more complete description of biotic assemblages in these agricultural streams. For species present at 25 – 75% of SVS-SWO sites, logistic regression showed increased likelihood of finding two fish species (Blacknose Dace, Central Stoneroller) and two BMI taxa (Asellidae, Lebertiidae) and decreased likelihood of finding two BMI taxa (Tabanidae, Physidae) with increasing enclosedness.

Enclosedness has modest effects in streams already impacted by agricultural practices and should be considered a potential threat to aquatic ecosystems. These findings provide a first step toward understanding headwater loss through enclosure and should be taken into consideration in future decisions to enclose streams.

Keywords: stream ecology, agriculture, fishes, invertebrates

CO-AUTHORSHIP STATEMENT

This dissertation will result in three manuscripts. Katie L. Stammler will be the lead author for all three as she played the lead role in defining the research problems, designing the research approach, analyzing the data, interpreting the results and will write the manuscripts. Dr. Robert C. Bailey will be the second author for one of the manuscripts (Downstream habitat and biota correlations to enclosedness) and third author on the other two manuscripts (Extent of enclosure and characteristics of catchments with varying degrees of enclosedness and Spatial and temporal variation of fish and BMI assemblage metrics in stream agro-ecosystems with varying degrees of enclosedness) as he advised on the study designs, analyses and interpretation of data and provided funding for these studies. Dr. Adam G. Yates will be second author for one of the manuscripts (Extent of enclosure and characteristics of catchments with varying degrees of enclosedness) as he contributed to the design of the study and generation of GIS data for analysis. Kari Everett will be the second author for one of the manuscripts (Spatial and temporal variation of fish and BMI assemblage metrics in stream agro-ecosystems with varying degrees of enclosedness) as she contributed to the design of the study and collection and analysis of fish data.

DEDICATION

I dedicate this thesis to my Gramma, the original Kathie Stammler. Her strength, tenacity, faithfulness and courage inspired me to be the best person I can be. None of my life's accomplishments would be possible without the sacrifices she made. Her look of pride when I completed my final draft was one of my greatest rewards.
January 1920 – April 2011

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INTRODUCTION

Headwater Streams

Headwater streams may be intermittent or permanently flowing, and usually comprise up to 50 - 90 % of the total length of a stream network (Dunne and Leopold 1978; Meyer et al. 2007; Nadeau and Rains 2007). They also provide many essential ecosystem services, but are often exposed to the most intense human stressors such as urbanization, agricultural runoff, channelization and enclosure (burial) (Meyer et al. 2003). This is due in part to their close connection with surrounding terrestrial ecosystems, inadequate knowledge of their role within the catchment, and a lack of consistent policy for their management (Gomi et al. 2002; Nadeau and Rains 2007). Although recent research has begun to evaluate and quantify the ecological importance of headwater streams (Meyer and Wallace 2001; Moore 2005; Nadeau and Rains 2007), the effects of human alterations, including enclosure, remain poorly understood.

Headwater streams provide vital services to downstream ecosystems through ecological and hydrological connectivity (Gomi et al. 2002). Invertebrate drift and inputs of energy and nutrients from the riparian zone (e.g. leaf litter and woody debris) make headwater streams an important source of energy and nutrients for downstream areas (Wallace et al. 1997; Wipfli and Gregovich 2002; Wipfli et al. 2007). Organic matter and nutrients from these sources, as well as from surface runoff, are processed in headwater streams, where they are either retained or exported downstream (Bernot et al. 2006; Meyer et al. 2003; Royer et al. 2004). Headwater streams can process and retain >50% of nutrient inputs because of their large surface to volume ratios, which favour rapid uptake. In larger streams, the time and distance required for uptake increases proportionally with depth (Peterson et al. 2001). Headwater streams also play an important role in moderating stream discharge by maintaining a relatively stable water supply and providing some degree of flood control (Allan 1995; Meyer et al. 2003). They can provide recharge areas for groundwater stores throughout the catchment and contribute 40-70% of the mean annual water volume to higher order streams (Alexander et al. 2007).

Headwater streams provide important habitat for stream biota, and are, thus, integral to the biodiversity of the entire stream network (Meyer et al. 2007). Because headwaters occur across a range of natural conditions, their habitats are among the most diverse and unique in river systems (Meyer et al. 2007), allowing them to support several species of invertebrates and fishes, many of which are endemic to headwaters (Clarke et al. 2008; Dietrich and Anderson 2000; Paller 1994). Sampling of small, previously unmapped, headwater streams has identified the presence of several insect genera, including some previously unidentified species (Meyer et al. 2007; Rasmussen 2004; Stout and Wallace 2003; Strayer 2000). Invertebrate assemblages in headwaters tend to be dominated by shredders adapted to feeding on available coarse particulate matter, while fish assemblages tend to be dominated by small, insectivorous species (Schlosser 1990; Smiley et al. 2005; Vannote et al. 1980) . Headwaters also provide spawning and rearing grounds for fishes (e.g. salmonids, darters, pikes) (e.g. Curry et al. 1997) and offer refuge to biota from predation, competition and extremes in flow and temperature (Meyer et al. 2007).

Policy for activities in headwater streams

Headwater streams tend to be poorly studied and under-represented on many topographic maps, making it difficult to enforce policies related to their management, or to accurately determine the effects of disturbance (Meyer and Wallace 2001; Roy et al. 2009). Recent court cases in the United States have shown that enforcement of the United States' *Clean Water Act* (1972) over headwater streams can be difficult (Leibowitz et al. 2008). In Canada, there is no specific protection of headwaters, although they are protected to some extent under Ontario's *Clean Water Act* (2006), which provides protection of source water, and Canada's *Fisheries Act* (1985), which protects fishes and their habitat. This combination of lack of knowledge, enforced policies and consideration of the potential for cumulative effects, leads to a common misconception that the effects of individual alterations to these systems will be negligible and temporary (Meyer and Wallace 2001).

Enclosure of Headwater Streams in Agricultural Areas

In agricultural areas, headwater streams are enclosed to increase the amount of arable land available and provide more efficient access to the land, which together increase the land value of a farm through increased crop yields (Sadler Richards 2004; Sadler Richards 2005). In this context, "enclosure" refers to the practice of headwater stream burial in agricultural areas (Figure 1), while "enclosedness" refers to the proportion of a stream network that is enclosed (enclosed length/total length) (Figure 2). Streams to be enclosed are excavated, and their water diverted through a subsurface drain tile or pipe, which is then buried. Water enters the pipe through a surface inlet and is discharged to a surface drain downstream (Van der Gulik et al. 2000).

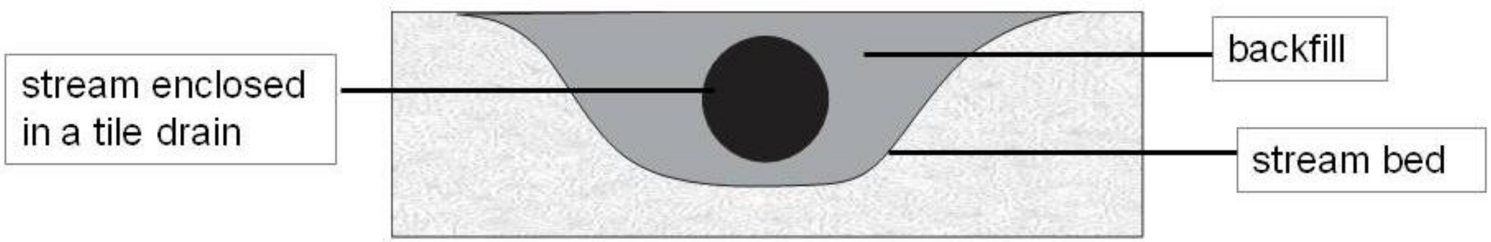

Figure 1 – A representation of a stream enclosure that occurs when the water from an existing open stream is diverted through a subsurface tile drain and the stream bed is backfilled.

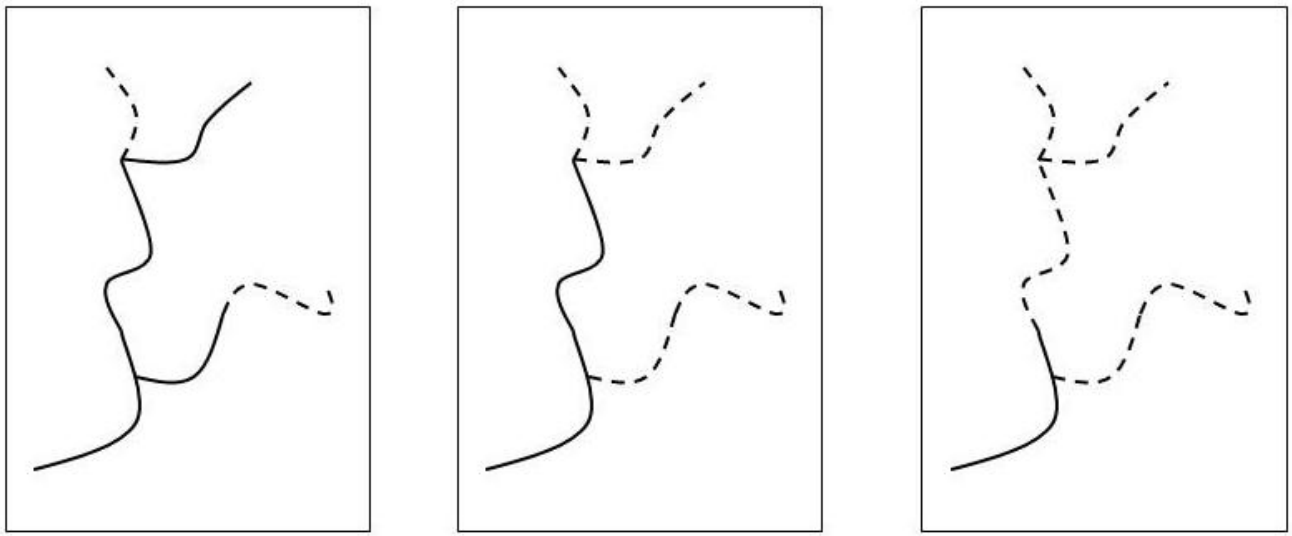

Figure 2 – A representation of enclosedness, which is the proportion of a stream network that is enclosed. Dashed lines represent enclosed streams. The stream networks increase from low to high enclosedness from left to right.

Headwater enclosure has been practiced throughout agricultural history, but has been poorly documented. Recent investigations have shown that enclosedness is occurring at high rates. In Sweden, the surface water area of the Kävlinge River was reduced to 3.4% of its original extent between 1820 and 1950. In two agricultural catchments in Finland, the length of open drainage ditches was reduced by 50% and 91% respectively between 1944 and 1997 (Hietala-Koivu et al. 2004). In Ontario, Veliz and Richards (2005) noted that 38% of the original stream network in the Nairn Creek catchment in the Ausable River Basin and 46% of streams in the middle Thames River catchment had been enclosed.

Enclosure also occurs in urban settings as land is developed for residential and commercial purposes (Elmore and Kaushal 2008; Roy et al. 2009). By 1966, the drainage density (km/km^2 of stream length) in the Rock Creek catchment in Maryland was reduced by 58% due to urbanization (Meyer and Wallace, 2001). Elmore and Kaushal (2008) noted that 66% of streams in Baltimore City were enclosed in contrast to 21% in the entire Gunpowder-Patapsco catchment. They also found that small, headwater streams (<260 ha) were enclosed at a higher rate than larger streams. In light of these examples, headwater enclosure has been acknowledged as a subject requiring further research (Meyer et al. 2005; Roy et al. 2009; Veliz and Sadler Richards 2005; Wenger et al. 2009).

Typically, agricultural drainage is used to remove excess water from cropland to create optimal soil and water conditions for crop growth and is necessary in areas that are naturally poorly drained (i.e. soils remain saturated for long periods following a precipitation event) (Skaggs et al. 1994; Van der Gulik et al. 2000). Drainage improvements are typically not made where the surrounding water table would be lowered to the extent that water would become unavailable, in deep sandy or organic soils, and in soils with low fertility, where drainage may cost more than the increased crop value (Miller and Gardiner 1998). It is likely that stream enclosures would also not occur under similar circumstances. Although the practice of enclosure is never explicitly discussed, the design of any drainage project must take into consideration hydrology, topography, hydraulics, contaminant and nutrient flow, and soil properties. Each drain or drainage scheme is designed for a specific purpose and requires specific calculations to determine the correct size, location and spacing (CSPI 2009; Irwin 1997a; Van Schilfgaarde 1974).

Policy for stream enclosures

In Ontario, headwater enclosure is an aspect of agricultural drainage and, as such, falls under the jurisdiction of Ontario's *Drainage Act* (1990) and is subject to the same process for approval as surface drains (Sid Vander Veen, OMAFRA, pers. comm.). This process includes approval by municipal council, a specific design by a drainage engineer, and final approval by either the local Conservation Authority or Fisheries and Oceans Canada (Irwin 1997b). In a typical year, the Ausable Bayfield Conservation Authority receives 5-10 requests for drain improvements including, on average, one request for a headwater enclosure (Davin Heinbuck, ABCA, pers. comm.). Requests for permission to enclose a stream are often denied because of the potential for harmful alteration, disruption or destruction of fish habitat (HADD), which is prohibited under Canada's *Fisheries Act* (1985). Stream enclosure is the least preferred option and is only considered after options to relocate or redesign the drainage project to avoid the HADD have been explored. An enclosure will only be approved if acceptable fish habitat compensation is implemented (DFO 2010). Many of the existing enclosed streams in Ontario were buried prior to these more recent changes in legislation.

Enclosure has likely occurred in other similar agricultural areas in the United States (R. Wayne Skaggs, North Carolina State University, pers.comm.), although there is no specific protocol for the practice in American jurisdictions (Larry C. Brown, The Ohio State University, pers. comm.). A nationwide permit issued by the United States Army Corps allows up to 300 linear feet (~91 m) of perennial stream to be enclosed provided that less than ½ an acre of non-tidal waters are lost and that the work is conducted to improve agricultural production, relocate a functional drain or construct farm buildings (Department of Defense 2007). The 300 linear foot limit may be waived for intermittent or ephemeral streams (Department of Defense 2007) and permits to enclose navigable waters must be approved under section 404 of the United States' *Clean Water Act* (1972).

Potential effects of headwater enclosure

There remains a paucity of literature directly examining the effects of headwater enclosure (Sadler Richards 2004). However, researchers have begun to speculate about the effects of headwater enclosure on downstream water quality and quantity in terms of the loss of headwater functions such as processing, transport and/or retention of nutrients, contaminants and organic matter; and hydrologic and sediment retention (Klocker et al. 2009; Mayer et al. 2010; Meyer and Wallace 2001). The effects of enclosure may also be predicted using the extensive body of literature available on agricultural subsurface field tile drainage. Field tile drainage uses subsurface drains to collect excess water to control the water table and provide uniform drainage of cropland (Van der Gulik et al. 2000).

The cumulative effect of burying several sections of headwater may produce similar results as systematic field tile drainage.

Streams in areas that rely heavily on field tile drainage typically have higher exports of inorganic nitrogen than those with little or no field tile drainage because water residence time in saturated soils is reduced or eliminated (Evans et al. 1995; McIsaac and Hu 2004) and hydrological alterations limit nutrient uptake (Freeman et al. 2007; Peterson et al. 2001). Excess nitrogen input from agricultural practices can already exceed a stream's capacity for processing nitrogen, causing high exports of inorganic nitrogen from agricultural streams (Bernot et al. 2006; Royer et al. 2004), which could be exacerbated by enclosure. Subsurface drainage can decrease sediment loss relative to surface drainage because the flowing water is confined to drain pipes, which eliminates bank erosion and runoff inputs (Blann et al. 2009; Evans et al. 1995; Skaggs et al. 1994). Consequently, phosphorus can also be reduced in subsurface drains because it is bound to sediment (Eastman et al. 2010), although the benefit of reduced phosphorus is not expected to outweigh the costs of increased nitrate and loss of landscape and species diversity (Hietala-Koivu et al. 2004).

Riparian zones, which are completely eliminated when headwater streams are enclosed, are important for aquatic habitat. They provide organic matter inputs, filter runoff, retain nutrients, and influence cover, in-stream temperature and primary production (Fitch and Adams 1998; Gregory et al. 1991). Changes in channel morphology, instream cover, habitat volume and substrate size are also seen with changes in riparian vegetation (Nerbonne and Vondracek 2001; Stauffer et al. 2000). While riparian zones in agricultural areas are often cleared for crops and grazing (Fitch and

Adams 1998; Nerbonne and Vondracek 2001), the grasses and shrubs present still provide these services.

Alteration of headwater streams causes changes in the natural flow regime by reducing water retention and moving water more quickly downstream, which increases the size and frequency of floods and reduces baseflow discharge levels (Dunne and Leopold 1978; Poff et al. 1997). Lands converted to agricultural use with artificial drainage generally experience higher peak runoff rates and frequency and intensity of flooding than natural areas (Hill 1976; Wiskow and van der Ploeg 2003). However, the addition of subsurface field tile drainage to land already converted to agriculture can decrease peak flows and surface runoff and increase baseflow relative to surface drainage alone (Gilliam and Skaggs 1986; Skaggs et al. 1994).

Enclosure may also affect downstream fish and invertebrate communities because physical stream attributes, hydrologic processes, nutrient dynamics, and the interactions between water, streambed and riparian areas have strong influences on the biota (Allan 1995; Gregory et al. 1991), which may all be altered by headwater enclosure. Human alterations to habitat result in the loss of species unable to tolerate the new conditions (Chapin et al. 2000; Gorman and Karr 1978); in particular, species richness has been shown to decline due to activities related to agricultural practices (Etnier 1972; Stauffer et al. 2000; Walser and Bart 1999).

Several studies have noted significant correlations between stream water quality in agricultural streams and fish and invertebrate assemblage metrics. The metrics measured included fish species richness, abundance, feeding and reproductive guild richness (Smiley et al. 2009), and indices of biotic integrity for both fish and benthic macroinvertebrate assemblages (Fitzpatrick et al. 2001; Miltner and Rankin 1998). In general, these metrics were negatively correlated to concentrations of ammonium, nitrate plus nitrite, total nitrogen and total phosphorus and positively related to dissolved oxygen and pH (Fitzpatrick et al. 2001; Miltner and Rankin 1998; Smiley et al. 2009).

A river's natural flow regime has a strong influence over both the physical habitat and biota inhabiting the stream (Poff et al. 1997). Hydrologic variability is well known to affect body size and shape and community structure in both fish and macroinvertebrates (Poff and Allan 1995; Poff and Ward 1989). Enclosure of headwaters also certainly results in the direct loss of habitat and further changes to the downstream community are expected through the loss of spawning areas and food sources (Meyer et al. 2007).

Importantly, the responses of both water quality and quantity noted above may vary depending on a host of factors including the natural local hydrologic cycle, soil properties, topography and configuration of the drainage network. These factors in turn interact with other agricultural stressors such as livestock, tillage practices and pesticide and herbicide use (Blann et al. 2009; Skaggs et al. 1994; Yates and Bailey 2010b), making it difficult to predict the exact effect of headwater enclosure and how the biological community may respond to these changes.

Testing Ecological Hypotheses

Ecological studies use many different approaches at different spatial and temporal scales to test hypotheses. Observational studies use natural variation in the variable of interest to determine patterns and test hypotheses. They can also be used to generate hypotheses and predictions about causal mechanisms (Gotelli 2004; Werner 1998) and can provide key information to guide further studies and generate additional questions for experimental research. Manipulative experiments, on the other hand, allow the researcher to test a specific hypothesis to identify the causal mechanism behind the observed patterns (Gotelli 2004; Werner 1998), but often lack ecological realism (Clements et al. 2002).

Both approaches can be conducted at different spatial scales. However, if the scale is too small (e.g. microcosms), important features of the abiotic environment, biotic inhabitants or processes with large spatial or temporal scales are often missing (Schindler 1998). This can create misleading results that cannot be extrapolated to the larger scale of natural ecosystems (Carpenter 1996; Englund and Cooper 2003). Large-scale studies (e.g. whole ecosystems) have the main advantage of realism and are the only way to measure the effects of changing variables in the context of natural ecosystem processes, with results that are directly applicable to the ecosystem in question (Carpenter 1998). However, replication of manipulative experiments at this scale is often impossible because of prohibitive costs, ethical constraints or the ability to find a closely matched ecosystem to act as a replicate (Likens 1998; Schindler 1998). In general, manipulative experiments are most practical at small scales, while observational studies are more practical at larger scales. This shift from manipulative to observational studies occurs at the level of the natural catchment scale due to logistic constraints (funding, adequate sampling) and the lack of comparable, unimpaired control sites (Power et al. 1998).

Factors affecting aquatic communities are influential at different scales including habitat, reach, riparian, catchment and region (Johnson and Host 2010; Lammert and Allan 1999; Stewart et al. 2001) and exhibit a hierarchical influence on stream biota (Allan and Johnson 1997; Frissell et al. 1986; Hynes 1975). Catchment scale features, such as geological features and the intensity of varying land uses, have been shown to influence local fish assemblages (Richards et al. 1996; Yates and Bailey 2010b). These features become increasingly influential in catchments that are already highly degraded by human activities such as agriculture (Wang et al. 2003). Advances in GIS techniques now make it possible to define the degree of anthropogenic stress within a catchment and current bioassessment literature encourages the use of these techniques to select sites across a gradient of the stressor of interest (Bailey et al. 2007; Danz et al. 2005).

Many observational studies strive to obtain as many samples as possible across a broad geographic range (Barbour et al. 1999; Reynoldson et al. 1999). This allows for rapid sampling, statistical independence of samples, and increased statistical power (Gotelli 2004), but provides only a snapshot of stream conditions and aquatic communities and may not capture their average response to disturbance or natural range of variation (Cooper et al. 1998). Another approach often used in observational studies is to sample a smaller set of sites with many visits over time (Fore et al. 1996; Ostrand and Wilde 2002). This has the advantage of increased precision, but the ability to generalize the results to a larger geographical scale can be limited (Wiley et al. 1997). It is well documented that both spatial and temporal variation of habitat features influence fish (Horwitz 1978; Meador and Matthews 1992; Ostrand and Wilde 2002) and benthic macroinvertebrate assemblages (Boulton et al. 1992; Kerans et al. 1992). Therefore, a sampling regime that incorporates both single visit sampling over a broad spatial scale and multiple visit sampling over a narrow spatial scale would provide a more complete picture of the effects of human activities on aquatic communities (Cooper et al. 1998; Wiley et al. 1997), although few studies put this into practice (Growns et al. 2006).

Research Objectives

The goal of this research was to create a better understanding of enclosure of headwater streams in southwestern Ontario. This was accomplished using a descriptive GIS study to identify and characterize the extent of headwater enclosure and two studies conducted at different scales to determine the effects of enclosedness on downstream aquatic communities.

The first objective of this study was to quantify enclosedness of streams in southwestern Ontario and to determine the natural and anthropogenic characteristics associated with varying degrees of enclosedness. To achieve this, a large-scale GIS study was undertaken that covered a broad spectrum of natural features and anthropogenic land uses. The information gained from this study is an important starting point for understanding the conditions under which enclosure is likely to occur.

The second objective of this study was to determine the effects of enclosedness on fish and benthic macroinvertebrate (BMI) assemblages in ecosystems downstream of enclosed headwaters. This was based on the hypothesis that enclosing headwaters will alter their natural function, that these effects will be cumulative and will result in changes to water quality and quantity, which will be reflected in the biota. Two sampling regimes were used to carry out this objective. Multiple visit sampling (MVS) was conducted at 10 sites in a small geographical area (Ausable River Basin) over a two year period. Single visit sampling (SVS) was conducted at 157 sites over a broad geographical range (southwestern Ontario). The combination of MVS and SVS sampling regimes accounts for both temporal and spatial variability in the response of biota to enclosedness. A secondary objective addressed with MVS was to determine sources of variability in fish and BMI assemblage metrics, which will inform future study designs in stream agro-ecosystems.

METHODS

Study Area

The extent of headwater enclosure and the characteristics of catchments with varying degrees of enclosedness were described for 29000 km of streams over an area of more than 16000 km^2 in southwestern Ontario, Canada. This area encompasses a variety of land use types including urban areas, pasture, small natural areas of mixed/deciduous broadleaf forest and wetlands, but is dominated by row-crop agriculture of corn, wheat, soy and tobacco (Statistics Canada 2007). The entire study area is underlain by Paleozoic sedimentary bedrock and exhibits surface landforms typical of a glaciated landscape. Although the region is dominated by coarse till deposits with imperfectly drained soils, the Long Point Region and other localized areas are characterized by fine glaciolacustrine deposits of well drained sandy soils (Yates and Bailey 2010b). All streams ultimately drain into Lake St. Clair (Thames and Sydenham rivers), Lake Huron (Ausable, Bayfield and Maitland rivers), or Lake Erie (Catfish Creek, Long Point Region) (Figure 3).

To determine the effects of enclosedness on downstream aquatic ecosystems, two studies were conducted over different time scales. Multiple visit sampling (MVS) was conducted over two years at two sites in each of five streams in the Ausable River Basin (ARB). The ARB is an 1142 km^2, J-shaped catchment with an outlet to Lake Huron near Port Franks, ON, and headwaters near Staffa, ON (Figure 4). The dominant land use is row crop agriculture with some pasture and vegetable farming. Soils consist mainly of clay to silt/clay till with poor to very poor infiltration (Veliz et al. 2006). Much of the area is subjected to artificial drainage including headwater enclosure. The Ausable River Recovery Strategy for Species at Risk (Ausable River Recovery Team 2005) identified

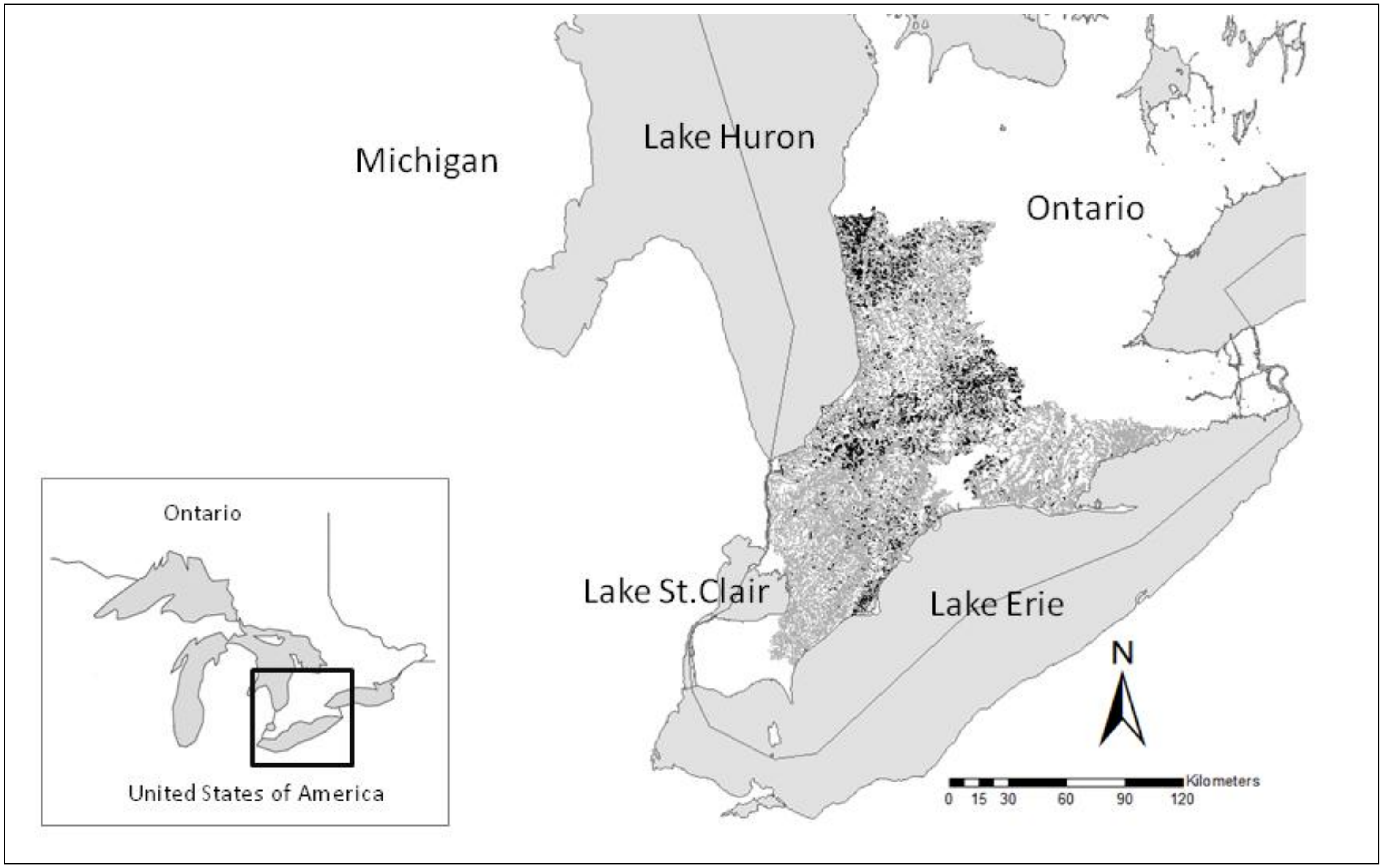

Figure 3 - Location of the study region for determining the extent and characteristics of catchments with varying degrees of enclosedness. The stream network within southwestern Ontario depicts both open (grey) and enclosed (black) streams within the study area. Of the 29000 km of stream length, 4100 km (14%) are enclosed.

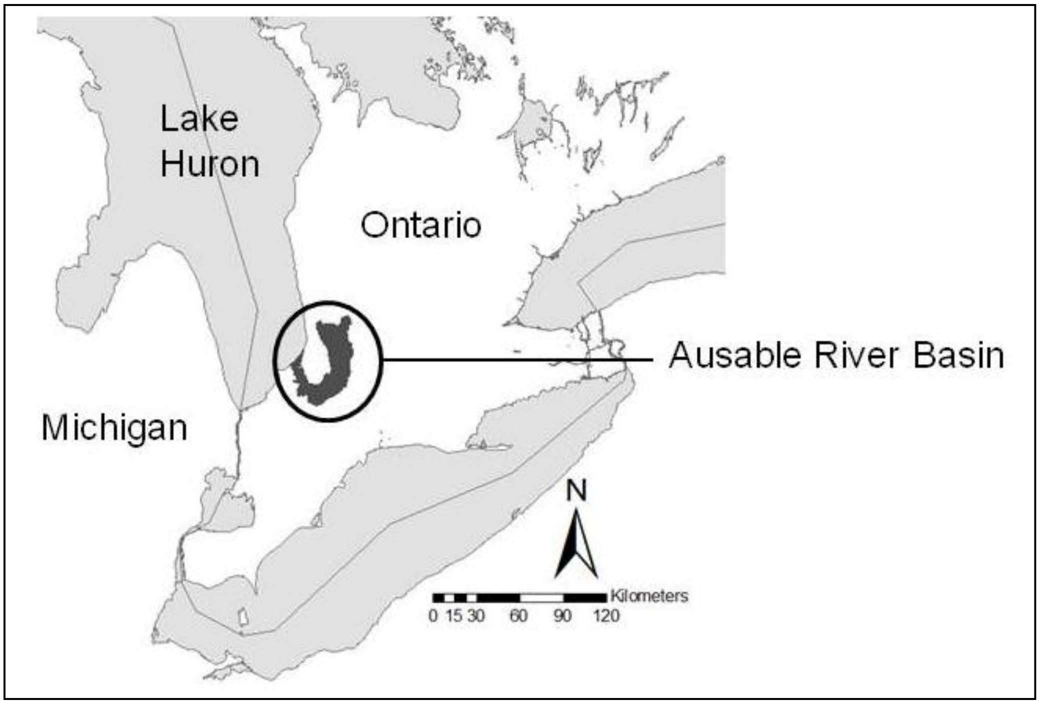

Figure 4 - Location of the Ausable River Basin (ARB) in southwestern Ontario, where multiple visit sampling (MVS) was conducted

the need for better understanding of the effects of enclosure as a necessary objective because it is not uncommon for more than 25% of the stream network in a catchment to be enclosed (Veliz and Sadler Richards 2005).

Single visit sampling (SVS) conducted by Yates (2008) was used to examine the effects of enclosedness on downstream aquatic ecosystems in southwestern Ontario (SWO) catchments. Sampling sites used for this study were located in the Grand River and Thames River catchments, as well as the Long Point Region. These sites are geographically near to the ARB but represent a wider range of physical features under which enclosure tends to occur (Figure 5).

Extent of enclosure and characteristics of catchments with varying degrees of enclosedness

Beginning in 2000, Conservation Authorities (CA) of Ontario classified municipal drains according to guidelines determined by Fisheries and Oceans Canada (DFO) to better manage drain maintenance (DFO 2010). During this project, enclosed streams were identified on the Ontario Stream Network (OSN), a stream layer generated between 1977 and 2000 by the Ontario Ministry of Natural Resources at the 1:10000 scale. Streams were classified as enclosed if they were indentified on the OSN but no open channel could be located during field surveys and drainage outlets indicating the presence of a subsurface tile drain were observed. Locations of enclosures may also have been identified on the OSN by municipal drainage superintendents. In Ontario, this is currently the only information available as a GIS layer that identifies enclosed streams. This information was obtained from participating CAs, including Ausable Bayfield CA, Catfish Creek CA, Long Point Region CA, Lower Thames Valley CA, Maitland Valley CA, St. Clair Region CA and Upper Thames River CA (Figure 6).

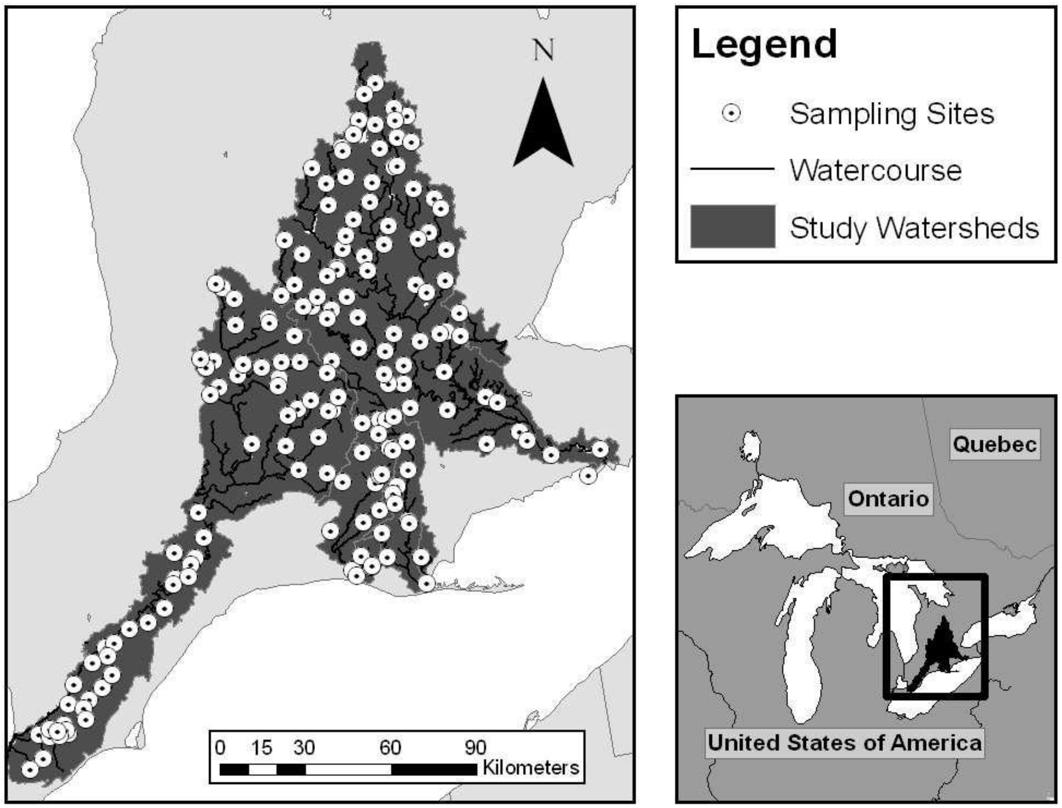

Figure 5 - Location of the southwestern Ontario (SWO) study region and 157 sampling sites used for single visit (SVS) sampling, from Yates (2008).

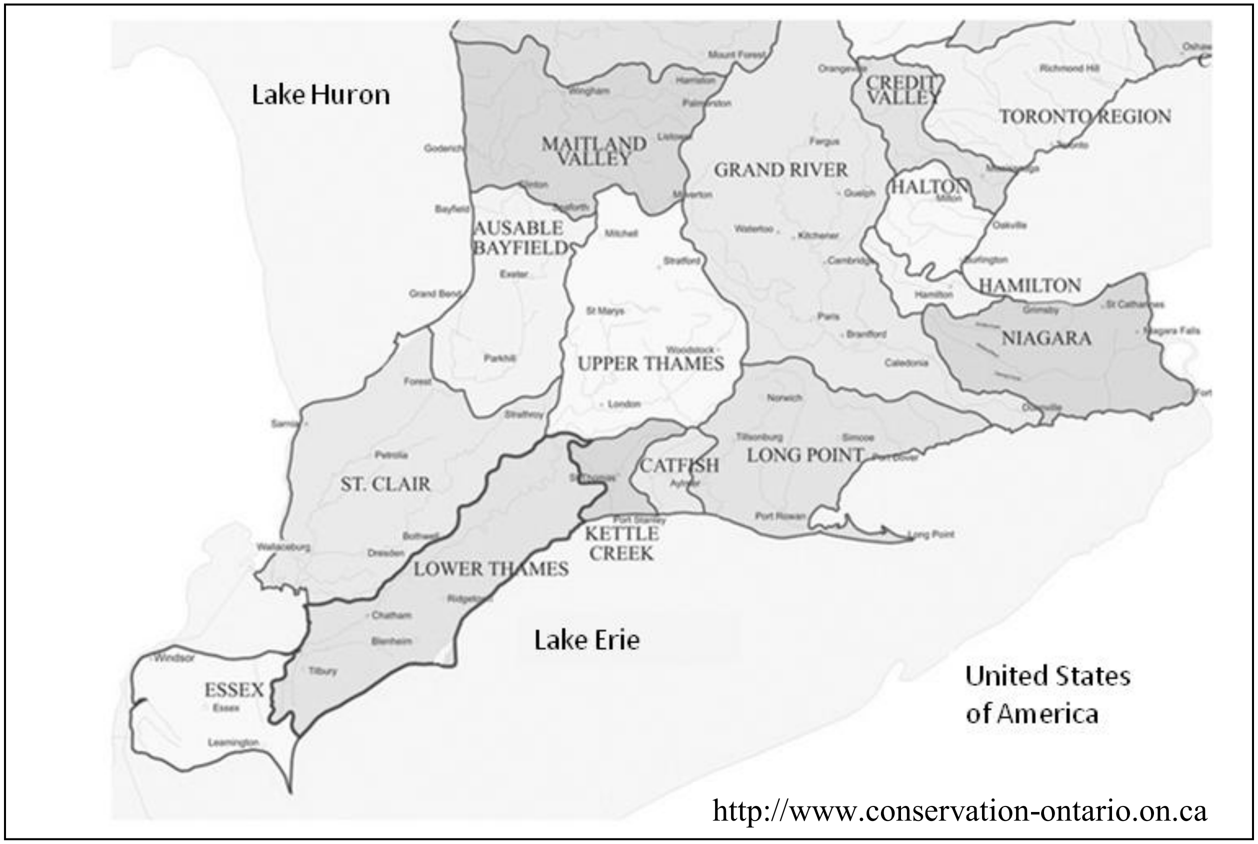

Figure 6 – Conservation Authorities in southwestern Ontario

Catchment delineation and description

Catchment areas were delineated using the ArcHydro 1.3 (ESRI 2009) extension for ArcGIS 9.2 (ESRI 2003) together with a 10 m resolution digital elevation model (DEM) and the Ontario Stream Network obtained from the Ontario Ministry of Natural Resources. The area of each catchment was calculated using the Xtools Pro (Data East 2010) extension. Only catchments greater than 1 km^2 and with more than 250 m of stream were kept for further analysis (n = 10106). Using the OSN, all stream orders were calculated in ArcGIS 9.2 (ESRI 2003) using the Strahler method (Strahler 1952) and the proportion of stream length enclosed by order was determined for all delineated catchments.

The catchments were intersected with provincial and federal GIS layers to determine their characteristics using ArcGIS 9.2 (ESRI 2003). The layers used describe agricultural soil (AGRSOIL) (Canada Soil Information Service 2008a), physiography (PHYS) (Ontario Geological Survey 2007), soil drainage (DRAIN) (Canada Soil Information Service 2008b), surficial geology deposits (SURGEO), which includes permeability (PERM) and primary geological material (PRMGEO) (Ontario Ministry of Northern Development Mines and Forestry 2011), and land cover (LANDCOV) (Ontario MInistry of Natural Resources 2008). Layers provide information about a single descriptor (e.g. PERM) and consist of several categories for that descriptor (e.g. low, medium, high). For each catchment, the area and proportion of each category for each descriptor was calculated using Xtools Pro (Data East 2010).

For each descriptor (AGRSOIL, PHYS, DRAIN, PERM, PRMGEO, SURGEO and LANDCOV), a Principal Component Analysis (PCA) of the covariance matrix was conducted on the proportion of the associated categories to reduce dimensionality using Systat v11.0 (Systat 2004). The number of components retained for each PCA was determined by examining the scree plot (Quinn and Keough 2002). In all cases, at least 85% of the variation in the data was explained by the retained principal components (PC).

Xtools Pro (Data East 2010) was also used to calculate stream length in each catchment, which was used to calculate drainage density (km/km^2) and enclosedness (proportion of stream length enclosed) in each catchment. Mean and standard deviation of depth to water table, annual precipitation and slope (change in elevation) for each catchment were calculated from available raster GIS layers using the ArcGIS 9.3 (ESRI 2008) extension Geospatial Modeling Environment (Beyer 2010).

Determination and testing of a model to describe characteristics of catchments with varying degrees of enclosedness

A regression tree analysis was then conducted with enclosedness as the target variable and 29 candidate predictors, including PC scores, drainage density, depth to water table, precipitation and slope, using Statistica (StatSoft 2010). Regression trees determine the best predictor variable, and the best value of that predictor variable, that splits the observations into groups that are as similar as possible in their value of a response variable (Breiman et al. 1984; De'ath and Fabricius 2000). Ten-fold cross-validation was performed and the 1-SE rule was applied to determine the optimal tree, which is the simplest tree whose estimated error rate is within one standard error of the tree with maximum reduction in error (Breiman et al. 1984).

To test the resulting regression tree model, each of the 10 catchments in the ARB and 157 catchments in southwestern Ontario used to study the effect of enclosedness on downstream biota were assigned to a terminal node based upon the splitting rules. One sample t-tests were then conducted to determine if the actual mean enclosedness of the catchments assigned to each node differed significantly from that predicted by the regression tree (Zar 1999).

Measurements of fish and benthic macroinvertebrate assemblages and physical habitat in catchments with varying degrees of enclosedness

Multiple visit sampling in the Ausable River Basin – site selection

To select sites for repeated sampling within the Ausable River Basin, all catchments within the ARB were delineated using the ArcHydro 1.2 (ESRI 2009) extension for ArcGIS 9.2 (ESRI 2003) with a 10m resolution DEM and the Ontario stream network. The area of each catchment was calculated using the extension Xtools Pro (Data East 2010). A total of 300 catchments were delineated that ranged in size from 2.2 - 1104.6 km^2 with an average size of 170.5 km^2. Catchments with drainage areas of 10 - 50 km^2 were then selected to ensure that watercourses were permanently flowing and wadeable, resulting in 82 candidate catchments.

Each candidate catchment was then characterized according to bedrock geology, ecoregion, climate zone, soil and surficial geology in ArcGIS 9.2 (ESRI 2003) using available federal and provincial shapefiles. The ArcGIS 9.2 extension Hawth's tools v3.24 (Beyer 2006) was used to calculate the areas and proportions of each descriptor in each of the candidate catchments. Hawth's tools was also used to calculate the total length of stream in each catchment, which was used to calculate drainage density (km/km^2) and enclosedness.

Sampling sites were selected to be similar in their natural variation based on characteristics of the natural landscape to isolate the effects of enclosedness. All 82 candidate catchments were characterized by bedrock of Paleozoic sedimentary rock;

therefore, bedrock type was excluded as a selection criterion. Selection criteria were chosen to be representative of the dominant classes of ecoregion, climate, soil and surficial geology found within the ARB. This included selecting catchments primarily within the Manitoulin-Lake Simcoe Ecoregion (dominant landuse: cropland and mixed forest; glacial till deposits; rolling topography with low relief) and climate district 567 (annual precipitation: 177 cm snow, 793.3 mm rain; mean temperature: January -6.4^{o}C, July 20.6^{o}C), and consisting mostly of Brookston soil (medium texture, well structured, poorly drained) and glacial till.

These candidate sites were then surveyed by local experts to assess physical features (e.g. stream width, accessibility, riparian land use) and were further described by human activities (road density, quarries, field tile design, and land cover) using ArcGIS 9.2 (ESRI 2003). Following this analysis, 10 sites were chosen from five streams: Bear, Centralia, Colwell, Duncrief, and Vail. In each stream, two sampling locations separated by 1-3 km (river length) were chosen. These sites were chosen to ensure that the types and amount of human activity varied as little as possible except enclosedness. Centralia was not among the candidate catchments but, based on expert opinion, was considered a suitable comparison for the other selected stream catchments (Table 1, Figure 7).

Single visit sampling across southwestern Ontario (SVS-SWO) – site description

Rural headwater catchments (n=157, Figure 5) were selected to represent the range of variation in surficial geology and rural anthropogenic activities in the study region (see Yates (2008) for full site descriptions), with catchment sizes ranging from 6 – 30 km^2. Enclosed streams in the Grand River catchments were identified by examining aerial photographs of the region overlaid by the OSN. Streams were considered enclosed

Table 1. Description of the land use in the ARB study catchments. Site names indicate the stream and location (1 = downstream, 2 = upstream)

Name	Area (km^2)	%Water	%Agriculture	%Forest	%Quarry	%Developed	Road Density (m/ha)	%Enclosedness
Bear 1	35.54	0.01	91.11	8.88	0.00	0.00	11.09	26.64
Bear 2	23.29	0.02	89.95	10.03	0.00	0.00	10.45	20.99
Centralia 1	37.20	0.36	88.20	4.14	0.00	7.30	17.00	5.93
Centralia 2	24.11	0.00	96.72	1.90	0.00	1.37	12.23	6.78
Colwell 1	34.22	0.08	92.25	7.67	0.00	0.00	9.92	15.39
Colwell 2	16.14	0.00	96.62	3.38	0.00	0.00	11.35	29.02
Duncrief 1	12.43	0.93	79.79	18.86	0.42	0.00	14.39	48.04
Duncrief 2	21.63	0.53	85.69	13.53	0.24	0.00	14.34	37.27
Vail 1	23.16	0.02	84.90	15.08	0.00	0.00	10.96	13.51
Vail 2	11.04	0.00	88.08	11.92	0.00	0.00	10.22	22.81

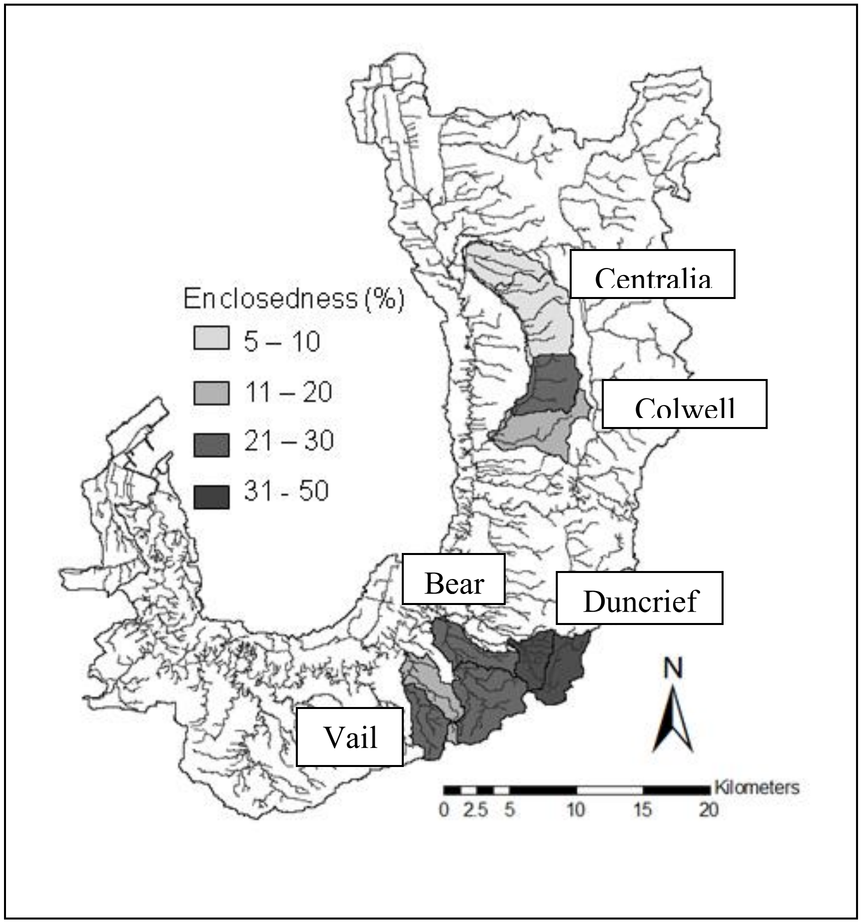

Figure 7 – Study catchments in the Ausable River Basin with enclosedness of the catchments indicated.

if no open channel was visible in close proximity to a water line on the OSN. Enclosedness was then determined as stated in the previous section using this information and the available GIS layers for the Thames River catchment and Long Point Region. Enclosedness could not be determined for three of the original catchments used by Yates (2008).

Biological sampling

Fishes and benthic macroinvertebrates (BMI) were sampled following the same protocol for both multiple visit sampling in the Ausable River Basin (MVS-ARB) and single visit sampling in southwestern Ontario (SVS-SWO). The MVS-ARB sites were sampled from 2007 to 2009 (summer and fall 2007, spring, summer and fall 2008, spring 2009). The SVS-SWO sites were sampled once in early fall of either 2006 or 2007.

Backpack electrofishing (pulsed DC current at 150-250 volts, 60 Hz) was carried out at a rate of approximately 10 sec/m^2 for a minimum shocking time of 600 seconds with a minimum site length of 30 m. The entire length of the site was sampled with a systematic single pass moving upstream through the site. All fishes were identified to species and counted. Fishes not readily identified in the field were either photographed or preserved in 70% ethanol for later laboratory identification. All other fishes were returned to the stream. BMI were collected in the same stream reach following fish collection using a 3-minute travelling kick sample with a 500 µm D-frame net in representative habitats, and all collected material was preserved in 70% ethanol.

In the laboratory, benthic samples were washed in a 500 µm sieve to remove large debris (e.g. leaves, rocks). The remaining material was then spread evenly in a gridded pan with 56 cells. A random number table was used to select cells for sub-sampling. All

BMI in selected cells were separated from other benthic material and counted under a dissecting microscope. This procedure was repeated until a minimum of 300 individuals were counted or the entire sample was processed. These were then identified by a taxonomist to the family level except for Amphipoda, Hirudinea, Lepidoptera and Megaloptera, which were identified to order, Odonata, which were identified to sub-order (Anisoptera and Zygoptera) and Chironomidae, which were identified to subfamily.

Habitat sampling

Habitat at MVS-ARB and SVS-SWO sites was assessed using the United States Enivronmental Protection Agency rapid bioassessment protocol for low gradient streams (Barbour et al. 1999) (Appendix 1). This protocol visually scores habitat quality based on 10 physical habitat parameters. Scores range from 0 to 20 for each parameter, where higher scores indicate better habitat quality. The parameters assessed included substrate characteristics, flow status, channel structure, bank stability and riparian vegetation. For each study (MVS-ARB and SVS-SWO), one person was assigned to conduct this qualitative assessment.

The physical attributes of the ARB sites were also measured during each biological sampling visit. Wetted width and water depth were measured at the upstream end, middle and downstream end of each site. Three measurements of depth were taken at each of these points. The percentage of riffle, run and pool habitat was also recorded. Beginning in spring 2008, Secchi depth was measured at each site using a turbidity tube.

Water quality sampling

At the MVS-ARB sites, water samples were collected monthly from July to November 2007. At the time of sampling, two replicate grab samples were taken from the water column and analyzed for total N, nitrate-N, NH_4^+, total P, K^+, Mg^{2+}, Ca^{2+} and Na^+. At the same sampling times and when biota were collected, a YSI multi-meter was used to determine physicochemical properties (temperature (°C), pH, DO (mg/L), and specific conductivity (µS/cm)).

Calculation of fish and BMI assemblage metrics

Biotic indices for fishes and invertebrates were calculated for both MVS-ARB and SVS-SWO sites. These included species richness (species count), abundance (total number of individuals per site), and the proportion of the total richness of tolerant, very tolerant and intolerant species for fishes (Coker et al. 2001; Halliwell et al. 1999). For BMI, species richness, diversity, density (number of individuals per cell of benthos picked) and estimated total abundance (density * number of cells picked) were calculated. A Family Biotic Index (FBI) was also calculated for BMI using tolerance values for each taxonomic group (Barbour et al. 1999; Hilsenhoff 1988a). Feeding guild was assigned for both fishes (Insectivore, Herbivore/Insectivore, Insectivore/Piscivore) (Coker et al. 2001) and BMI (Collector-filterer, Collector-gatherer, Predator, Piercer-herbivore, Scraper-grazer, Shredder) (Barbour et al. 1999; Merritt and Cummins 1984; Vieira et al. 2006) and the proportion of the total richness was calculated for each.

Spatial and temporal variation of fish and BMI assemblage metrics in stream agro-ecosystems with varying degrees of enclosedness

Variability in basic biotic indices at the MVS-ARB sites may be due to differences between streams, within streams or among seasons. Nested 3-factor analyses of variance (ANOVA) were conducted with stream (5 levels - Bear, Centralia, Colwell, Duncrief, Vail) and date (6 levels – Summer 07, Fall 07, Spring 08, Summer 08, Fall 08, Spring 09) as fixed factors, site (2 levels – Site 1, Site 2) as a nested factor within stream and the interaction of date with the nested term using Systat v11.0 (Systat 2004). For fish data, ANOVAs were performed with species richness, abundance and proportion of intolerant, tolerant and very tolerant species as response variables. For invertebrate data, species richness, density, estimated total abundance, diversity and FBI were used as response variables. The proportion of variance explained by each of the factors (η^2) was estimated as the ratio of the factor Sum of Squares (SS) to the total SS, i.e. $\eta^2 = SS_{factor} / SS_{total}$ (Quinn and Keough 2002).

Nonmetric Multidimensional Scaling (NMDS) was used to describe the patterns in the assemblage structure of fishes and invertebrates in the MVS sites in terms of both spatial and temporal variability using PC-ORD (McCune and Mefford 1999). The Jaccard distance measure was used for presence/absence fish data and the Sorensen distance measure was used for relative abundance BMI data with random starting coordinates for 250 runs with real data. Sites were ordinated in species space using weighted average scores for species and a Monte Carlo test was performed on 100 runs of randomized data to evaluate the significance of the resulting ordination (McCune and Grace 2002).

Downstream habitat and biota correlations to enclosedness

For each of the 10 sites in the ARB, the data from seasonal biota sampling were pooled to create a single observation. For fish and invertebrate assemblages, a new matrix for presence/absence was created to reflect whether a species occurred in any sampling period. The biotic indices described above were recalculated using these pooled data. The average value was used for habitat variables.

Principal Component Analyses (PCA) of the covariance matrix was conducted to reduce dimensionality of each of four groups of habitat descriptors for the MVS-ARB sites (US-EPA qualitative habitat scores, physical features, water chemistry and physicochemical properties) and for US-EPA qualitative habitat scores for SVS-SWO data using Systat v11.0 (Systat 2004). The number of components retained was determined by examining the generated scree plot (Quinn and Keough 2002). The calculated scores for each catchment and PC were used as variables in subsequent analysis.

Preliminary analyses revealed that the majority of the data did not meet the assumptions of linear models, and standard transformations did not lead them to better satisfy the major assumptions of linear models (normality of residuals, homoscedasticity). Spearman rank correlations with $\alpha = 0.05$ were performed to assess the correlation between enclosure level and indices of habitat and biotic assemblages using Systat v11.0 (Systat 2004). Dependent variables included the PC scores generated from the preceding analysis of habitat data, % agricultural land use, eight indices of fish assemblages and 10 indices of BMI assemblages with enclosedness as the independent variable.

Logistic regressions were conducted to determine whether the likelihood of observing a particular species was correlated with enclosedness (Quinn and Keough 2002) using Systat v11.0 (Systat 2004). For these tests, presence or absence for each fish species (n = 35 (MVS-ARB), 33 (SVS-SWO)) and each BMI taxa (n = 74 (MVS-ARB), 94 (SVS-SWO)) was the response variable and enclosedness was the predictor variable.

RESULTS

Extent of enclosure and characteristics of catchments with varying degrees of enclosedness

A total of 10106 catchments were delineated ranging in size from 1 - 5725 km^2 ($\bar{x}$ = 212 km^2; median = 5.6 km^2; Figure 8). Enclosedness ranged from 0-100% ($\bar{x}$ = 16.5%; median = 2.8%; Figure 9). Catchments with 100% enclosedness ranged in area from 1-41 km^2 (n = 253, $\bar{x}$ = 2.6 km^2, median = 1.6 km^2), while catchments with 0% enclosedness ranged from 1 – 487 km^2 ($\bar{x}$ = 11.0 km^2, median = 3.2 km^2) **(**Figure 10). Of the 29000 km of streams in the study area across all 10106 catchments, 4100 km (14%) were enclosed. Of these, 23% (by length) of all 1st order streams, 10% of 2nd order streams, 2% of 3rd order streams and less than 1% of 4th and 5th order streams were enclosed (Figure 11).

The most common categories of agricultural soils in the catchments were those most suited for growing crops or for pasture land (class 1-3 and 5). The PCA of agricultural soil types (AGRSOIL) produced two principal components (PCs) that explained 91% of the total variation among catchments. The first component was a gradient that distinguished catchments with the two best classes of agricultural soil for crop lands. The second component was a gradient that distinguished catchments with the third best class for crop land from the two best classes (Table 2).

Although the catchments in this area were dominated by till (including bevelled till plain, till moraine and drumlized and undrumlinized till plains), clay and sand plains were also common, along with meltwater spillways. The PCA of physiography (PHYS) produced five PCs that explained 93% of the total variation among catchments. The first component distinguished catchments with a high proportion of undrumlinized till plains

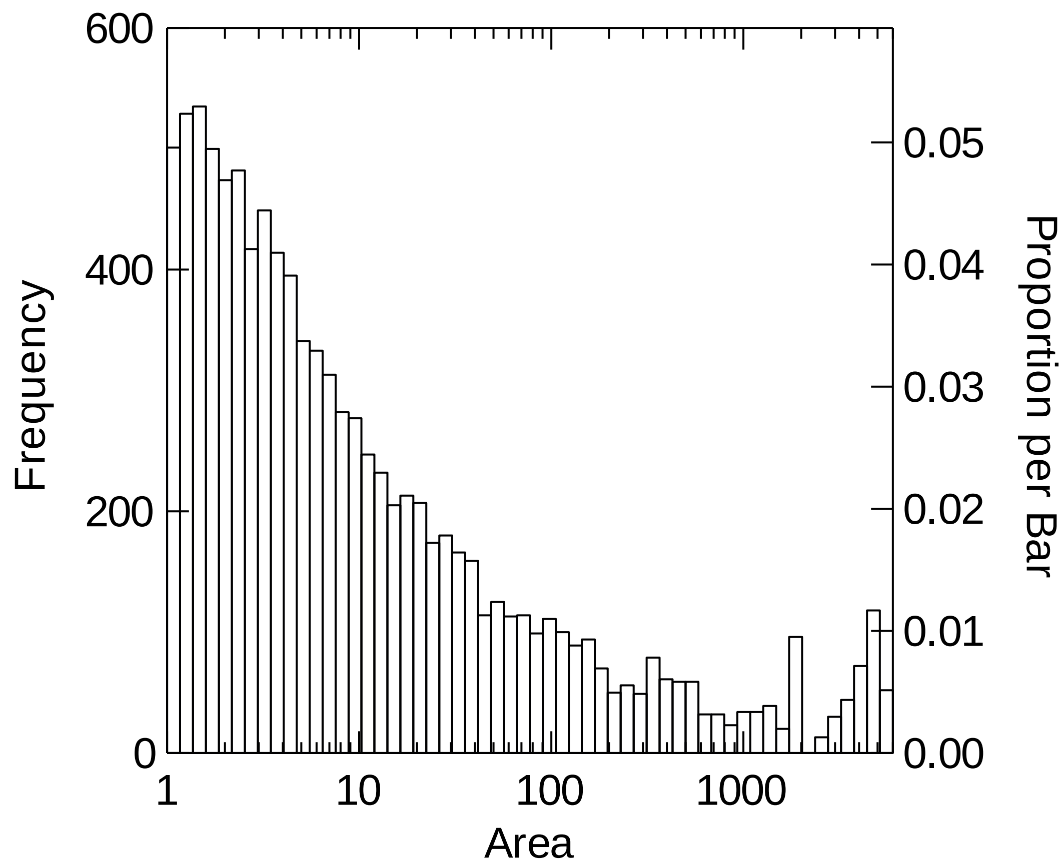

Figure 8 - Catchment area (km^2) of southwestern Ontario catchments (n=10106).

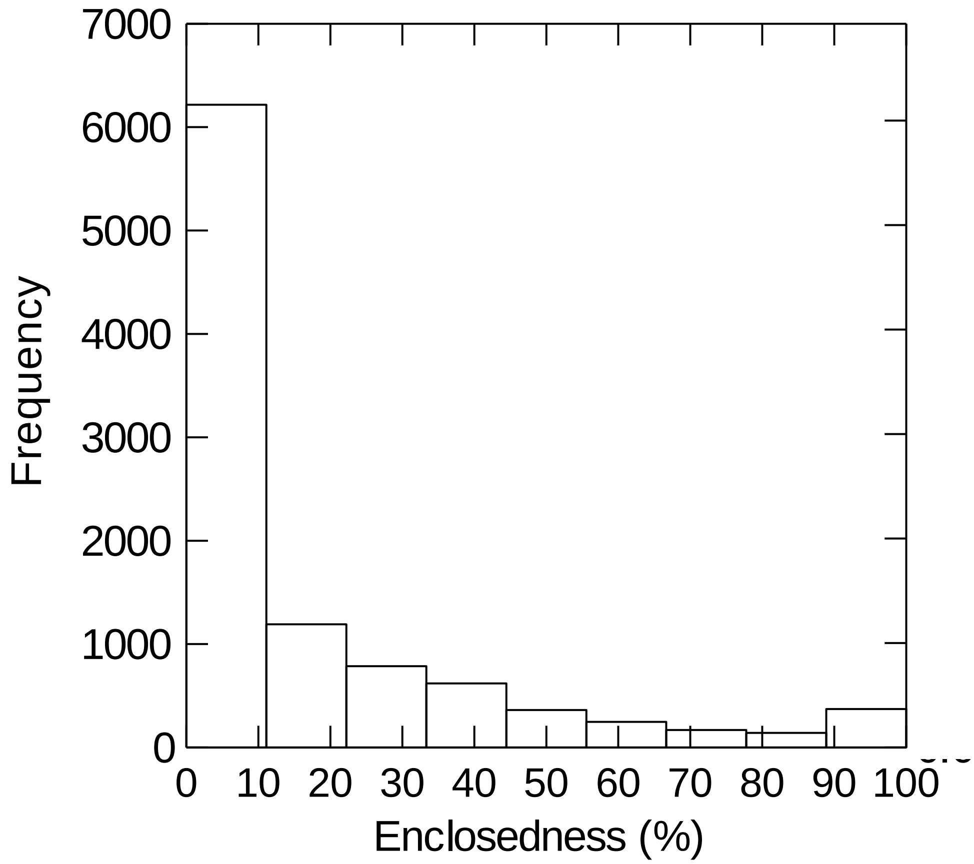

Figure 9 – Frequency distribution of enclosedness (%) in southwestern Ontario catchments (n = 10106).

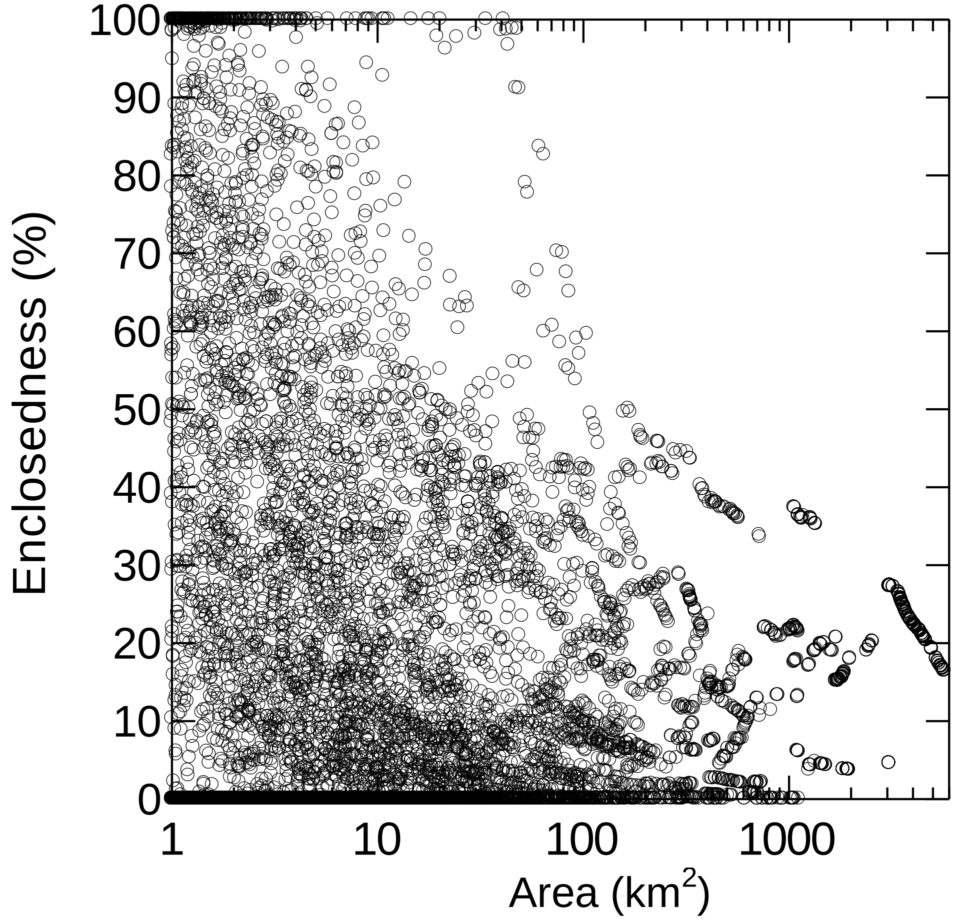

Figure 10 – Enclosedness (%) versus catchment size (km^2), n = 10106

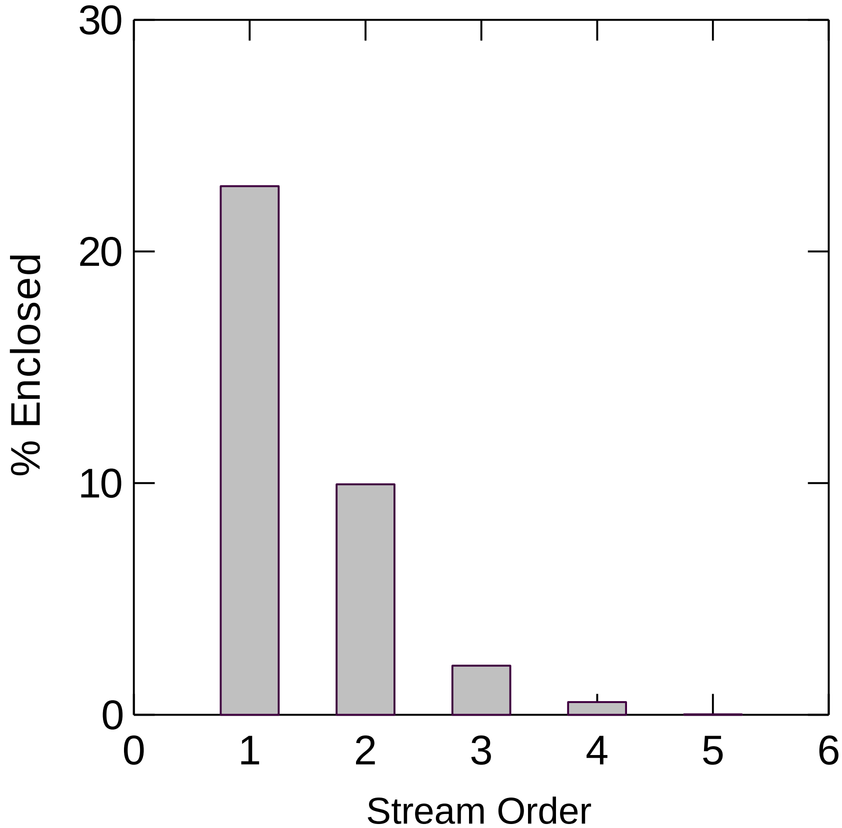

Figure 11 - Proportion of stream length enclosed versus stream order for streams in southwestern Ontario

Table 2 – Occurrence, proportion and Principal Component loadings of agricultural soil categories (AGRSOIL) in southwestern Ontario catchments (n=10106).

Category	Description	% Occurrence (n=10106)	Mean % Area of Basin(sd)	AGRSOIL_PC1	AGRSOIL_PC2
UNCL	Unclassified	31	2.4 (8.2)	0.01	0.01
1	No limitations for crop use	74	32.8 (34.0)	-0.32	-0.10
2	Moderate limitations for crop use, require moderate conservation practices	92	44.8 (33.0)	0.30	-0.14
3	Moderate – severe limitations for crop use, require special conservation practices	57	13.8 (23.2)	0.04	0.22
4	Severe limitation for crop use, low productivity except for specially adapted crops	25	0.8 (3.0)	0.00	0.00
5	Not suitable for crop land, used for perennial forage plants and grazing	43	1.8 (4.9)	-0.01	0.00
6	Not suitable for crop land, used for perennial forage plants and some grazing	20	1.1 (5.2)	-0.01	0.00
7	Not suitable for crop land or pasture	16	0.8 (4.3)	0.00	0.00
ORGANIC	Organic soils	31	1.7 (5.4)	-0.01	0.00
WATER	Water area	22	0.2 (1.0)	-0.00	0.00

from those with a high proportion of sand plains. The second component distinguished catchments with a high proportion of drumlinized till plains from those with a high proportion of bevelled till plains. The third component distinguished catchments with a high proportion of clay plains from those with a high proportion of undrumlinized till plains. The fourth component distinguished catchments with a high proportion of drumlinized till plain from those with a high proportion of clay plains and undrumlinized till plains. The fifth component distinguished catchments with a high proportion of drumlinized till plains from those with a high proportion of till moraine (Table 3).

Soils in the catchments were predominantly poor or imperfectly drained with some areas being well drained. The PCA of soil drainage (DRAIN) produced two PCs that explained 89% of the total variation among catchments. The first component distinguished catchments with a high proportion of poorly drained soils from those with a high proportion of imperfectly drained soils. The second component distinguished catchments with a high proportion of well or imperfectly drained soils from those with a high proportion of poorly drained soils (Table 4).

The majority of catchments in the study area contained areas of both low and high permeability. The PCA of permeability (PERM) produced two PCs that explained 99% of the total variation among catchments. The first component was a gradient from low to high permeability. The second component distinguished catchments with a high proportion of low/medium permeability from those with high permeability (Table 5).

The most common primary geological material in the catchments was diamicton, however, sand, clay/silt/sand, sand/gravel and clay/silt materials were also common. The PCA of primary geological material (PRMGEO) produced five PCs that explained 97%

Table 3 – Occurrence, proportion and Principal Component loadings of physiography categories (PHYS) in southwestern Ontario catchments (n=10106).

Category	% Occurrence (n=10106)	Mean % Area of Basin(sd)	PHYS_PC1	PHYS_PC2	PHYS_PC3	PHYS_PC4	PHYS_PC5
Beaches	19	0.6 (2.6)	0.00	0.00	0.00	0.00	0.00
Bevelled till plains	23	14.9 (32.7)	0.07	-0.30	-0.09	-0.06	0.01
Clay plains	29	12.7 (28.7)	-0.03	0.00	0.25	-0.13	0.00
Drumlin	11	0.6 (3.0)	0.00	0.00	0.00	0.01	0.00
Eskers	8	0.2 (1.2)	0.00	0.00	0.00	0.00	0.00
Kame moraine	12	2.2 (11.6)	0.01	0.00	0.00	0.01	0.00
Lime	1	0.2 (3.4)	0.00	0.00	0.00	0.00	0.00
Peat	11	0.6 (4.4)	0.00	0.00	0.00	0.00	0.00
Till moraine	46	15.4 (25.6)	0.05	0.04	-0.01	0.08	-0.23
Drumlinized till plain	18	9.7 (25.0	0.03	0.01	0.05	0.19	0.14
Undrumlinized till plain	34	17.8 (31.1)	0.17	0.18	-0.13	-0.12	0.06
Sand plain	36	19.2 (34.5)	-0.33	0.04	-0.10	-0.03	0.01
Spillway	33	6.0 (13.2)	0.02	0.02	0.01	0.05	0.00

Table 4 - Occurrence, proportion and Principal Component loadings of soil drainage categories (DRAIN) in southwestern Ontario catchments (n=10106).

Category	Description	% Occurrence (n=10106)	Mean % Area of Basin(sd)	DRAIN_PC1	DRAIN_PC2
Very Poor	Excess water in soil, water Table at or near surface	28	1.3 (4.9)	0.00	0.00
Poor	Excess water in soil	90	28.9 (29.7)	-0.25	0.15
Imperfect	Slow drainage, range of storage water capacity	90	41.7 (29.7)	0.27	0.11
Moderately well	Intermediate to high storage water capacity	23	1.3 (4.5)	0.00	0.00
Well	Intermediate storage water capacity	62	19.5 (27.2)	-0.03	-0.27
Rapid	Low storage water capacity	22	2.7 (9.5)	0.00	0.00
Other	Water, alluvium or urban land	59	4.0 (8.7)	0.01	0.00
Unclassified		17	0.6 (3.0)	0.00	0.00

Table 5 – Occurrence, proportion and Principal Component loadings of surficial geology permeability categories (PERM) in southwestern Ontario catchments (n=10106).

Category	**% Occurrence (n=10106)**	**Mean % Area of Catchment(sd)**	**PERM_PC1**	**PERM_PC2**
Low	89	57.3 (38.0)	-0.38	0.02
Low-Medium	31	15.1 (28.5)	0.16	-0.234
Medium-High	10	0.2 (2.1)	0.00	0.00
High	79	25.2 (30.3)	0.22	0.21
Variable	58	2.1 (4.6)	0.00	0.00
Unclassified	1	0.0 (0.0)	0.00	0.00

of the total variation among catchments. The first component distinguished catchments with a high proportion of sand and from those with a high proportion of diamicton. The second component distinguished catchments with high proportions of clay/silt/sand and/or clay/silt from those with high proportions of sand and/or diamicton. The third component distinguished catchments with a high proportion of clay/silt from those with a high proportion of clay/silt/sand. The fourth component distinguished catchments with high proportions of clay and/or sand/gravel those with high proportions of clay/silt/sand and/or clay/silt. The fifth component distinguished catchments with a high proportion of sand/gravel those with a high proportion of clay (Table 6).

The most common surficial geology type in the catchments was till, followed by glaciolacustrine material and fine sand/silt. The PCA of surficial geology (SURGEO) produced two PCs that explained 87% of the total variation among catchments. The first component distinguished catchments with high proportions of fine and/or coarse glaciolacustrine deposits from those with a high proportion of till deposits. The second component was a gradient from coarse to fine glaciolacustrine deposits (Table 7).

Almost all catchments contained at least some road, forest, wetland and agriculture. The PCA of land cover (LANDCOV) produced three PCs that explained 97% of the total variation among catchments. The first component was a gradient from agricultural to non-agricultural (wetland, forest, impervious urban) land cover. The second component distinguished catchments with a high proportion of impervious urban land cover from those with a high proportion of wetlands. The third component distinguished catchments with a high proportion of forest from those with high proportions of wetland and/or impervious urban land cover (Table 8).

Table 6 – Occurrence, proportion and Principal Component loadings of primary geological material categories (PRMGEO) in southwestern Ontario catchments (n=10106).

Category	% Occurrence (n=10106)	Mean Areal % of Basin(sd)	PRMGEO_ PC1	PRMGEO_ PC2	PRMGEO_ PC3	PRMGEO_ PC4	PRMGEO_ PC5
Clay	13	2.3 (12.4)	0.01	0.02	0.01	0.07	-0.10
Clay silt	40	8.4 (19.2)	0.03	0.09	0.16	-0.05	0.01
Clay silt sand	51	9.6 (20.3)	0.06	0.13	-0.14	-0.06	0.00
Clay silt sand gravel	10	0.2 (0.9)	0.00	0.00	0.00	0.00	0.00
Diamicton	85	52.5 (35.1)	-0.34	-0.08	-0.01	-0.03	0.00
Fill	4	0.0 (0.6)	0.00	0.00	0.00	0.00	0.00
Gravel	15	0.5 (3.2)	0.00	0.00	0.00	0.00	0.00
Organic	27	0.9 (3.8)	0.00	0.00	0.00	0.01	0.00
Paleological material	8	0.1 (1.2)	0.00	0.00	0.00	0.00	0.00
Sand	58	17.5 (28.2)	0.21	-0.18	-0.01	-0.04	0.01
Sand gravel	47	5.3 (12.9)	0.01	0.02	-0.01	0.10	0.08
Silt	18	0.9 (4.1)	0.00	0.00	0.00	0.00	0.00
Silt sand	11	0.9 (5.8)	0.00	0.01	0.00	0.00	0.00
Silt sand gravel	37	1.0 (2.7)	0.00	0.00	0.00	0.00	0.00
Unclassified	1	0.0 (0.0)	0.00	0.00	0.00	0.00	0.00

Table 7 - Occurrence, proportion and Principal Component loadings of surficial geology categories (SURGEO) in southwestern Ontario catchments (n=10106).

Category	% Occurrence (n=10106)	Mean Areal % of Basin(sd)	SURGEO_PC1	SURGEO_PC2
Limestone, dolomite, sandstone; bedrock	8	0.1 (1.2)	0.00	0.00
Glacial Till (sandy to silty)	85	52.5 (35.1)	-0.35	-0.02
Boulders, sand, gravel; kames; glaciofluvial	32	3.1 (10.0)	0.00	0.00
Sand, gravel, boulder; dunes; glaciofluvial	32	3.9 (10.4)	0.00	0.00
Silt and clay; glaciolacustrine	72	20.0 (27.0)	0.15	0.22
Sand and gravel; glaciolacustrine	57	16.8 (28.4)	0.19	-0.21
Medium to coarse gravel; fluvial gravel	13	0.4 (3.1)	0.00	0.00
Silt and clay, fine sand; lacustrine	0	0.0 (0.1)	0.00	0.00
Sand, minor gravel; modern lacustrine	5	0.6 (5.6)	0.00	0.00
Stabilized dunes, sand deposits	10	0.2 (2.1)	0.00	0.00
Fine sand, silt, organics, muck; alluvium	56	1.6 (2.9)	0.00	0.00
Peat, muck; swamp and bog deposits	27	0.9 (3.8)	0.00	0.00
Tailings	4	0.0 (0.6)	0.00	0.00
Unclassified	1	0.0 (0.0)	0.00	0.00

Table 8 – Occurrence, proportion and Principal Component loadings of land cover categories (LANDCOV) in southwestern Ontario catchments (n=10106).

Category	% Occurrence (n=10106)	Mean Areal % of Basin(sd)	LANDCOV_PC1	LANDCOV_PC2	LANDCOV_PC3
Open tallgrass prairie	5	0.0 (0.1)	0.00	0.00	0.00
Tallgrass woodland	1	0.0 (0.1)	0.00	0.00	0.00
Coniferous, mixed and/or deciduous forest	97	6.1 (5.6)	0.04	0.01	-0.04
Plantations – tree cultivations	49	0.4 (1.1)	0.00	0.00	0.00
Hedge rows	79	0.5 (0.6)	0.00	0.00	0.00
Road/railway	100	2.7 (1.6)	0.01	-0.01	0.00
Extraction	26	0.2 (1.0)	0.00	0.00	0.00
Urban – permeable	42	0.6 (1.8)	0.01	-0.01	0.00
Urban – impermeable	44	1.3 (4.8)	0.02	-0.04	0.01
Swamp, fen, bog, and/or marsh	95	6.2 (7.0)	0.05	0.04	0.02
Open water	55	0.2 (0.8)	0.00	0.00	0.00
Agriculture	100	82.0 (12.6)	-0.13	0.01	0.00

The regression tree created using the derived PC scores and additional physical features as predictor variables (Table 9) had four splits resulting in five possible terminal nodes (Figure 12). The splits were made on the basis of differences in mean annual precipitation, PHYS_PC4, mean depth to water table and drainage density. The catchments in the terminal node containing the highest average level of enclosedness (Figure 12; ID = 33) received greater than 990.4 mm of precipitation annually and had drainage densities greater than 1.84 km/km^2. This is in contrast to catchments in the terminal node with the lowest average level of enclosedness (Figure 12; ID = 6), which received less than 990.4 mm of precipitation annually, were physiographically composed of clay or undrumlinized till planes and had shallow depth to water table (< 15.2 m). The remaining three terminal nodes contained catchments with average values of enclosedness between these two extreme nodes.

Of the 167 catchments assigned to terminal nodes of the regression tree, over half were assigned to the node with the lowest enclosedness (n = 86, Figure 12, ID = 6) and few were assigned to the node with the highest enclosedness (n = 4, Figure 12, ID = 33). The t-tests showed that actual enclosedness differed from predicted enclosedness for catchments assigned to nodes ID = 5 ($t = -4.579$, $df = 26$, $p < 0.001$) and ID = 6 ($t = 5.124$, $df = 85$, $p < 0.001$). Whereas, actual enclosedness did not differ from predicted enclosedness for catchments assigned to nodes ID = 7 ($t = 1.685$, $df = 14$, $p = 0.114$), ID = 32 ($t = -0.446$, $df = 34$, $p = 0.558$) and ID = 33 ($t = -0.409$, $df = 3$, $p = 0.710$) (Figure 13).

Table 9 – Minimum, mean, maximum and standard deviation of the physical features of catchments in southwestern Ontario (n = 10106)

Variable	**Min**	**Mean**	**Max**	**SD**
Log area	0.00	1.02	3.76	0.898
Drainage density (km/km^2)	0.02	1.41	7.18	0.631
Mean slope	0.00	0.87	6.03	0.556
Variability in slope	0.00	1.00	7.51	0.709
Mean annual precipitation (mm)	826.65	965.22	1038.82	50.297
Variability in annual precipitation (mm)	0.00	3.36	51.90	7.050
Mean depth to water table (m)	-19.62	15.21	108.74	15.202
Variability in depth to water table (m)	0.00	5.39	31.06	5.132

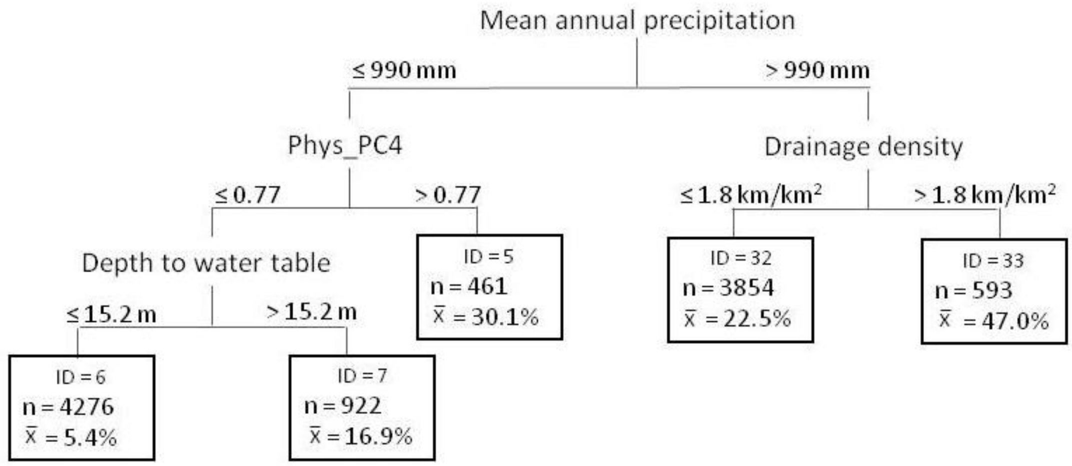

Figure 12 - Regression tree showing natural features associated with enclosedness in catchments in southwestern Ontario (n = 10106). The tree shows four different variables and their thresholds used for splitting the catchments into terminal nodes. Each terminal node is indicated by a unique identification number (ID). The number of catchments (n) and average enclosedness ($\bar{x}$) are also indicated for each node.

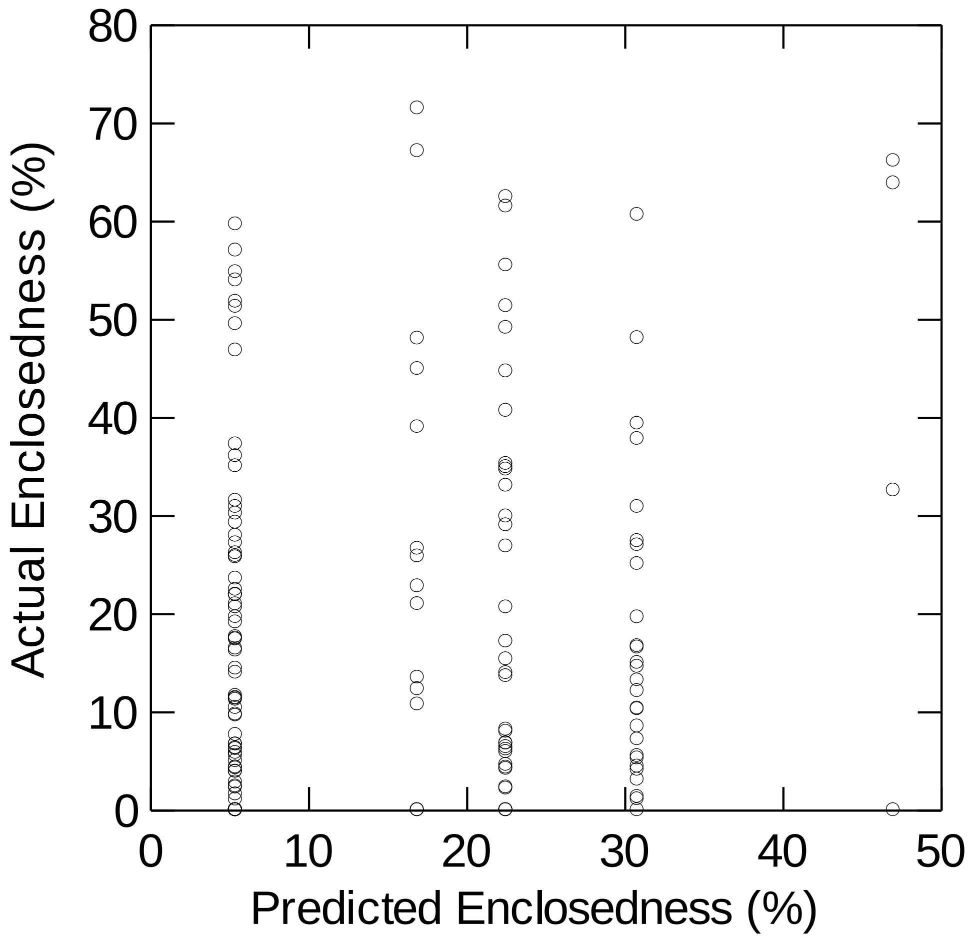

Figure 13 – Predicted enclosedness (%) versus actual enclosedness (%) for catchments assigned to regression tree nodes (N = 167)

Measurements of fish and benthic macroinvertebrate assemblages and physical habitat in catchments with varying degrees of enclosedness

Enclosedness in the MVS-ARB sites ranged from 5.9% to 48.0% ($\bar{x}$ = 64%) (Table 1) and in the SVS-SWO sites from 0% to 71.6% ($\bar{x}$ = 17.8%).

Summary of biota collected at both the MVS-ARB and SVS-SWO sites

Across the 10 sites and two years of MVS-ARB sampling, a total of 7124 fishes representing 35 species were collected (Table 10). Fish species richness ranged from 4 to 14 species per sampling visit ($\bar{x}$ = 9.1; sd = 2.3). The total abundance of fishes caught per sampling visit ranged from 14 to 658 individuals ($\bar{x}$ = 118.7; sd = 120.1). A total of 74 BMI taxa were collected at the MVS-ARB sites (Table 11). BMI richness per sampling visit ranged from 10 to 31 taxa ($\bar{x}$ = 18.2; sd = 4.8), and density of BMI ranged from 3.8 to 316 individuals per sampling cell ($\bar{x}$ = 61.4; sd = 51.2).

Among the 157 SVS-SWO sites, a total of 21318 fishes were collected representing 33 species (Appendix 2). Fish species richness per site ranged from 1 to 16 species ($\bar{x}$ = 7.3; sd = 3.3). The total abundance of fishes caught per site ranged from 1 to 822 individuals ($\bar{x}$ = 135.8; sd = 141.7). A total of 94 taxonomic groups of BMI were also collected at the SVS-SWO sites (Appendix 3). BMI richness per site ranged from 7 to 36 taxa ($\bar{x}$ = 19.4; sd = 5.4). Density of BMI ranged from 1.4 to 167.5 individuals per sampling cell ($\bar{x}$ = 39.3; sd = 33.4).

Table 10 - Fish species present in the Ausable River Basin (MVS-ARB) and the number of times each was found out of all visits (n = 60) or out of all sites (n = 10).

Family	Species	Common Name	Occurrence out of 60 visits over 10 sites	Occurrence out of 10 sites over 6 visits
Catostimidae	*Catostomus commersoni*	White sucker	49	10
	Hypentelium nigricans	Northern hogsucker	5	2
	Moxostoma macrolepidotum	Shorthead redhorse	2	2
Centrarchidae	*Ambloplites rupestris*	Rock bass	26	6
	Lepomis cyanellus	Green sunfish	4	2
	Lepomis gibbosus	Pumpkinseed	11	4
	Lepomis macrochirus	Bluegill	4	2
	Lepomis megalotis	Longear sunfish	5	1
	Micropterus dolomieu	Smallmouth bass	7	3
Cottidae	*Cottus bairdii*	Mottled sculpin	6	1
Cyprinidae	*Campostoma anomalum*	Central stoneroller	37	9
	Hybognathus hankinsoni	Brassy minnow	1	1
	Luxilus chrysocephalus	Striped shiner	11	5
	Luxilus cornutus	Common shiner	35	10
	Lythrurus umbratilis	Redfin shiner	5	3
	Nocomis biguttatus	Hornyhead chub	6	2
	Notropis heterolepis	Blacknose shiner	2	1
	Notropis rubellus	Rosyface shiner	4	4
	Notropis volucellus	Mimic shiner	4	3

	Phoxinus eos	Northern redbelly dace	11	4
	Pimephales notatus	Bluntnose minnow	49	9
	Pimephales promelas	Fathead minnow	24	7
	Rhinichthys atratulus	Blacknose dace	18	5
	Semotilus atromaculatus	Creek chub	52	10
Esocidae	*Esox lucius*	Northern pike	3	1
Gasterosteidae	*Culaea inconstans*	Brook stickleback	27	7
Ictaluridae	*Ameiurus melas*	Black bullhead	6	4
	Noturus flavus	Stonecat	1	1
Percidae	*Etheostoma blennioides*	Greenside darter	8	4
	Etheostoma flabellare	Fantail darter	2	2
	Etheostoma microperca	Least darter	38	8
	Etheostoma nigrum	Johnny darter	54	10
	Percina maculata	Blackside darter	15	5
Salmonidae	*Oncorhynchus mykiss*	Rainbow trout	8	3
	Salvelinus fontinalis	Brook trout	4	1

Table 11 - Benthic macroinvertebrate species present in the Ausable River Basin (MVS-ARB) and the number of times each was found out of all visits (n = 60) or out of all sites (n = 10).

Order	Suborder	Family	Subfamily	Occurrence out of 60 visits over 10 sites	Occurrence out of 10 sites over 6 visits
Hirudinea				16	7
Lumbriculida		Lumbriculidae		11	5
Tubificida		Naididae		19	10
		Tubificidae		26	10
Acariformes		Lebertidae		24	7
		Limnesiidae		12	6
		Sperichonidae		11	7
		Torrenticolidae		1	1
Prostigmata		Arrenuridae		4	3
		Hydrozetidae		40	10
		Mideopsidea		30	9
		Pionidae		13	7
		Torrenticolidae		1	1
		Unionicolidae		25	9
Amphipoda		Gammaridae		3	1
		Talitridae		30	8

Decapoda	Cambaridae		17	7
Isopoda	Asellidae		12	4
Coleoptera	Chrysomelidae		1	1
	Curculionidae		1	1
	Dryopidae		1	1
	Dytiscidae		15	8
	Elmidae		59	10
	Gyrinidae		2	2
	Haliplidae		27	9
	Hydroporinae		16	8
	Hydraenidae		4	4
	Hydrophilidae		15	6
	Psephenidae		3	3
	Staphylinidae		2	1
Diptera	Ceratopogonidae		41	10
	Chironomidae	Chironominae	60	10
		Orthocladiinae	38	10
		Diamesinae	2	2
		Tanypodinae	49	10
	Culicidae		3	3
	Dixidae		4	3
	Empididae		14	7
	Ephydridae		2	1
	Psychodidae		3	3
	Simuliidae		8	5
	Tabanidae		16	7

		Tipulidae	8	4
Ephemeroptera		Baetidae	27	10
		Caenidae	48	10
		Ephemerellidae	3	3
		Ephemeridae	4	4
		Heptageniidae	20	6
		Leptophlebiidae	13	5
		Leptohyphidae	2	2
Hemiptera		Corixidae	45	9
Lepidoptera			3	3
Megaloptera			16	8
Odonata	Zygotera		34	9
	Anisoptera		10	6
Plecoptera		Capniidae	16	6
		Nemouridae	1	1
		Perlodidae	4	3
		Taeniopterygidae	1	1
Trichoptera		Helicopsychidae	2	1
		Hydroptilidae	7	4
		Hydropsychidae	7	2
		Leptoceridae	17	6
		Limnephilidae	7	5

	Phryganeidae	2	2
	Philopotamidae	3	1
	Polycentropodidae	6	4
Lamellibranchia	Sphaeriidae	41	10
Prosobranchia	Hydrobiidae	1	1
	Valvatidae	2	1
Pulmonata	Lymnaeidae	5	4
	Physidae	30	9
	Planorbidae	17	7
Tricladida	Dugesiidae	10	6

Habitat description of MVS-ARB and SVS-SWO sites

On a scale from 0 (poorest habitat) to 200 (best habitat), USEPA habitat scores ranged from 80 to 149 for MVS-ARB sites (Table 12), and from 67 to 177 ($\overline{x}$ = 128.9) for SVS-SWO sites. In the MVS-ARB sites, wetted width ranged from 130 to 820 cm and depth ranged from 6 to 95 cm. The average percent Pool ranged from 12.7 to 67.8%, average percent Run ranged from 32.2 to 82.2% and average percent Riffle ranged from 0 to 25.3%. Values at each site varied across these ranges (Table 13).

In the MVS-ARB sites, total N ranged from 0.10 to 3.11 µg/ml, nitrate-N ranged from 1.00 to 46.60 µg/ml, Ca^{2+} ranged from 42.20 to 298.20 µg/ml, Mg^{2+} ranged from 15.25 to 109.60, total P ranged from 0.10 to 8.46 µg/ml, K^{+} ranged from 1.96 to 8.46 µg/ml and Na^{2+} ranged from 6.57 to 67.80 µg/ml (Table 14). Water temperature ranged from 5.2 to 24.5^{o}C, pH ranged from 6.87 to 8.84, conductivity ranged from 0.48 to 1.73 µS, dissolved oxygen ranged from 6.96 to 22.75 mg/L and Secchi depth ranged from 19 to 120 cm (Table 15). Values at each site varied across these ranges.

Spatial and temporal variation of fish and BMI assemblages in stream agro-ecosystems with varying degrees of enclosedness

The nested ANOVAs for fish indices showed that the interaction of date and sites within a stream had the largest effect size for species richness and total abundance. In contrast, either stream or sites within a stream accounted for the largest amount of variation in the proportion of intolerant, very tolerant and tolerant species (Table 16). The nested ANOVAs for BMI indices showed that date accounted for the most variation in taxon richness, density and diversity, while stream accounted for the most variation in estimated abundance and FBI (Table 17).

Table 12 - Habitat scores for sites in the Ausable River Basin (MVS-ARB). See Appendix 1 for a full description of each qualitative variable score.

		Pool				**Channel**		**Bank**			
site	**Instream cover**	**substrate**	**variability**	**Sediment deposition**	**flow**	**alteration**	**sinuosity**	**stability**	**cover**	**Riparian width**	**Total**
Bear 1	13	13	14	10	18	13	7	14	17	16	135
Bear 2	14	12	14	9	18	14	11	14	19	20	143
Centralia 1	9	9	12	8	16	13	7	9	13	6	100
Centralia 2	13	11	13	11	17	14	9	12	13	6	120
Colwell 1	10	8	14	9	18	17	11	12	16	15	128
Colwell 2	12	14	6	8	18	13	7	16	13	8	115
Duncrief 1	8	4	14	10	18	14	9	13	17	14	118
Duncrief 2	16	17	16	16	18	13	7	16	14	7	138
Vail 1	7	3	13	12	18	14	9	10	18	20	123
Vail 2	11	10	12	10	18	13	8	14	16	7	117
Overall Mean	11	10	13	10	18	14	8	13	15	12	124

Table 13 – Physical habitat measures (mean ± standard deviation) for sites in the Ausable River Basin (MVS-ARB) sites.

Site	Width (cm)	Depth (cm)	% Pool	% Run	% Riffle
Bear 1	522.2 ± 175.0	36.7 ± 17.1	61.7 ± 29.2	20.0 ± 29.7	18.3 ± 27.3
Bear 2	461.7 ± 106.6	30.6 ± 11.9	40.6 ± 36.5	33.3 ± 40.9	26.1 ± 38.8
Centralia 1	439.2 ± 75.7	37.6 ± 20.2	21.1 ± 16.8	54.4 ± 42.3	24.4 ± 36.2
Centralia 2	319.4 ± 111.7	43.2 ± 21.4	34.4 ± 26.2	45.0 ± 32.2	20.6 ± 32.6
Colwell 1	295.3 ± 98.5	39.8 ± 18.7	58.9 ± 39.7	26.7 ± 35.2	14.4 ± 23.8
Colwell 2	572.2 ± 161.0	43.3 ± 22.0	67.8 ± 26.9	21.1 ± 29.3	11.1 ± 19.1
Duncrief 1	395.3 ± 45.5	46.7 ± 15.9	26.0 ± 13.0	44.0 ± 39.1	30.0 ± 38.4
Duncrief 2	425.3 ± 76.0	24.5 ± 8.1	12.7 ± 14.4	58.7 ± 37.2	28.7 ± 29.2
Vail 1	357.3 ± 50.2	40.8 ± 20.5	27.3 ± 17.9	40.0 ± 37.0	32.7 ± 41.8
Vail 2	362.7 ± 95.1	34.8 ± 18.1	41.3 ± 14.6	37.3 ± 34.1	22.9 ± 27.9
Overall Mean	415.1	37.8	38.8	56.8	4.4

Table 14 – Water chemistry parameters (µg/mL) (mean ± standard deviation) for sites in the Ausable River Basin (MVS-ARB) sites.

Site	Total N			Nitrate-N			Ca^{2+}			Mg^{2+}		
Bear 1	0.23	±	0.29	1.00	±	0.00	60.10	±	18.60	20.91	±	5.56
Bear 2	0.26	±	0.36	1.50	±	0.87	76.34	±	16.84	25.24	±	5.08
Centralia 1	0.20	±	0.14	22.52	±	10.54	180.40	±	53.10	63.69	±	25.53
Centralia 2	0.73	±	1.33	34.59	±	12.80	203.82	±	67.49	71.11	±	30.82
Colwell 1	0.19	±	0.12	3.32	±	5.02	65.09	±	24.79	18.20	±	2.32
Colwell 2	0.43	±	0.33	3.21	±	4.94	65.07	±	26.87	20.24	±	2.38
Duncrief 1	0.69	±	0.96	2.51	±	1.16	91.38	±	9.16	23.11	±	4.13
Duncrief 2	0.29	±	0.27	2.23	±	0.90	94.64	±	7.10	23.11	±	3.90
Vail 1	0.50	±	0.55	2.73	±	1.97	69.34	±	16.98	23.11	±	5.32
Vail 2	0.17	±	0.16	1.63	±	0.76	70.67	±	8.22	22.78	±	3.93
Overall Mean	0.37			7.52			97.68			31.15		

Site	Total P			K^{+}			Na^{+}		
Bear 1	0.10	±	0.00	4.49	±	1.29	18.26	±	2.52
Bear 2	0.11	±	0.01	3.47	±	0.88	19.35	±	9.10
Centralia 1	0.55	±	0.60	60.50	±	36.12	28.71	±	4.17
Centralia 2	1.31	±	1.07	74.06	±	39.72	37.71	±	17.99
Colwell 1	0.10	±	0.00	4.25	±	1.50	11.83	±	6.28
Colwell 2	0.10	±	0.00	2.83	±	0.73	12.79	±	7.28
Duncrief 1	0.10	±	0.00	2.50	±	0.54	9.19	±	0.61
Duncrief 2	0.10	±	0.00	2.42	±	0.25	9.52	±	0.84
Vail 1	0.10	±	0.00	2.93	±	0.44	9.01	±	1.31
Vail 2	0.10	±	0.00	2.98	±	0.83	13.48	±	1.92
Overall Mean	0.27			16.04			16.98		

Table 15 – Physicochemical water parameters (mean ± standard deviation) for sites in the Ausable River Basin (MVS-ARB).

site	Temperature (°C)	pH	Conductivity (μS)	DO(mg/L)	Secchi depth (cm)
Bear 1	14.2 ± 6.9	8.02 ± 0.30	0.55 ± 0.05	12.87 ± 2.27	82 ± 24
Bear 2	13.5 ± 6.1	7.81 ± 0.33	0.59 ± 0.05	11.62 ± 2.44	96 ± 21
Centralia 1	14.3 ± 6.6	8.14 ± 0.31	0.91 ± 0.30	12.42 ± 2.75	96 ± 21
Centralia 2	19.2 ± 6.0	8.18 ± 0.42	0.91 ± 0.21	17.84 ± 3.92	60 ± 43
Colwell 1	14.4 ± 4.2	7.65 ± 0.63	0.74 ± 0.19	11.85 ± 3.12	108 ± 23
Colwell 2	15.0 ± 5.0	7.72 ± 0.70	0.96 ± 0.52	12.11 ± 3.08	86 ± 28
Duncrief 1	15.3 ± 4.6	8.08 ± 0.53	0.73 ± 0.31	11.16 ± 2.24	85 ± 24
Duncrief 2	12.4 ± 3.9	7.79 ± 0.56	0.60 ± 0.06	10.16 ± 1.57	96 ± 28
Vail 1	13.2 ± 7.3	7.97 ± 0.26	0.56 ± 0.06	13.31 ± 1.99	69 ± 25
Vail 2	13.7 ± 7.5	7.84 ± 0.41	0.55 ± 0.04	13.18 ± 1.91	41 ± 19
Overall Mean	14.5	6.87	0.710	12.65	79.7

Table 16 - Nested ANOVA results used to determine spatial and temporal variation of fish assemblage metrics at the multiple visit sampling sites in the Ausable River Basin (MVS-ARB).

Response variable and source of variation	df	F	SS	R^2
Species richness				
Stream	4	2.939	62.57	0.194
Date	5	0.403	10.73	0.033
Site(Stream)	5	1.773	47.12	0.147
Date*Site(Stream)	25	0.713	94.83	0.295
Error	20		106.43	
Total abundance				
Stream	4	4.806	146926.40	0.173
Date	5	2.512	96009.93	0.113
Site(Stream)	5	2.304	88036.33	0.103
Date*Site(Stream)	25	1.924	367709.67	0.432
Error	20		851545.73	
Proportion of intolerant species				
Stream	4	21.450	0.03	0.336
Date	5	0.992	0.28	0.019
Site(Stream)	5	18.358	0.30	0.360
Date*Site(Stream)	25	2.109	0.90	0.207
Error	20		0.39	
Proportion of very tolerant species				
Stream	4	39.743	0.59	0.393
Date	5	1.345	0.03	0.017
Site(Stream)	5	34.612	0.64	0.428
Date*Site(Stream)	25	1.827	0.17	0.113
Error	20		0.07	
Proportion of tolerant species				
Stream	4	39.220	1.13	0.647
Date	5	0.998	0.04	0.021
Site(Stream)	5	6.945	0.25	0.143
Date*Site(Stream)	25	1.030	0.19	0.106
Error	20		0.14	

Table 17 - Nested ANOVA results used to determine spatial and temporal variation of benthic macroinvertebrate assemblage metrics at the multiple visit sampling sites in the Ausable River Basin (MVS-ARB).

Response variable and source of variation	df	F	SS	R^2
Richness				
Stream	4	2.807	169.43	0.126
Date	5	5.640	425.48	0.317
Site(Stream)	5	2.928	220.92	0.165
Date*Site(Stream)	25	0.590	222.58	0.166
Error	20		301.77	
Density				
Stream	4	3.920	16644.75	0.107
Date	5	8.785	46630.89	0.301
Site(Stream)	5	3.299	17513.52	0.113
Date*Site(Stream)	25	1.990	52814.75	0.341
Error	20		21231.78	
Estimated abundance				
Stream	4	3.874	5.00	0.275
Date	5	8.754	1.41	0.078
Site(Stream)	5	2.391	3.86	0.212
Date*Site(Stream)	25	1.822	1.47	0.081
Error	20		6.46	
Family biotic index				
Stream	4	7.654	12.83	0.335
Date	5	2.620	5.49	0.143
Site(Stream)	5	1.974	4.14	0.108
Date*Site(Stream)	25	0.715	7.50	0.195
Error	20		8.38	
Diversity				
Stream	4	1.732	15.60	0.063
Date	5	8.346	93.89	0.380
Site(Stream)	5	2.414	27.16	0.110
Date*Site(Stream)	25	1.158	65.13	0.264
Error	20		45.0	

The NMDS ordination for fish composition resulted in a relatively stable, two-dimensional solution, with a final stress of 19.03 and final instability of 0.0001 over 133 iterations (Figure 14). Results of the Monte Carlo test show that the ordination produces stronger axes than expected by chance ($p = 0.001$). Fish assemblages at each site tended to occupy a specific space on the ordination plot, but also displayed variability in time, with the exception of Colwell, which varied little with time. There was a high degree of overlap between sites and times, with the exception of the upstream site of Duncrief (Du2), which occupied a unique space on the plot.

The NMDS ordination for invertebrates resulted in a relatively stable, two-dimensional solution, with a final stress of 17.10 and final instability of 0.0001 over 68 iterations (Figure 15). Results of the Monte Carlo test show that the ordination produces stronger axes than expected by chance ($p = 0.001$). BMI community composition tended to be more variable in both time and space than fish community composition, and sites were not clearly distinguishable. Across all sites and years, spring and summer samples tended to occupy opposite ends of axis 2, though both were variable on axis 1, while fall samples were more evenly distributed across both axes.

Downstream habitat and biota correlations to enclosedness

The PCA of US-EPA habitat scores produced three PCs for both the MVS-ARB and SVS-SWO sites. These components together explained more than 70% of the variation in habitat scores for each dataset. For MVS-ARB, the first PC mainly involved pool substrate composition and riparian width, the second PC mainly involved instream cover complexity, pool substrate composition and riparian width and the third PC mainly involved pool variability and sediment deposition (Table 18). Many of the habitat

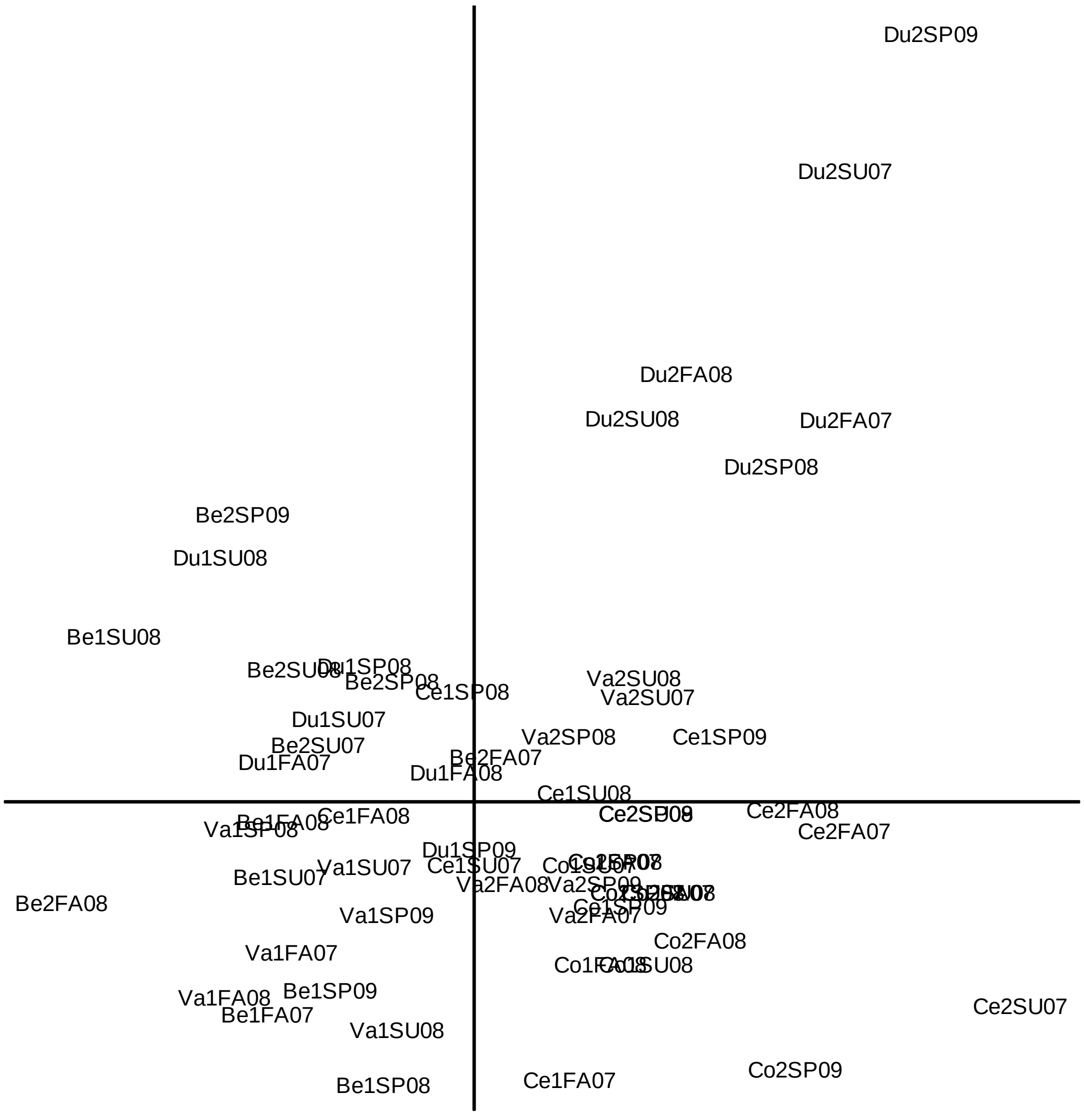

Figure 14 - Ordination plot of a non-metric multidimensional scaling for fish presence/absence in sampling sites in the Ausable River Basin (MVS-ARB). Site codes include a two character code for site (Be = Bear, Ce = Centralia, Co = Colwell, Du = Duncrief, Va = Vail), a number (1 = downstream, 2 = upstream), a two character code for sampling season (SU = summer, FA = fall, SP = spring) and the last two digits of the year sampled.

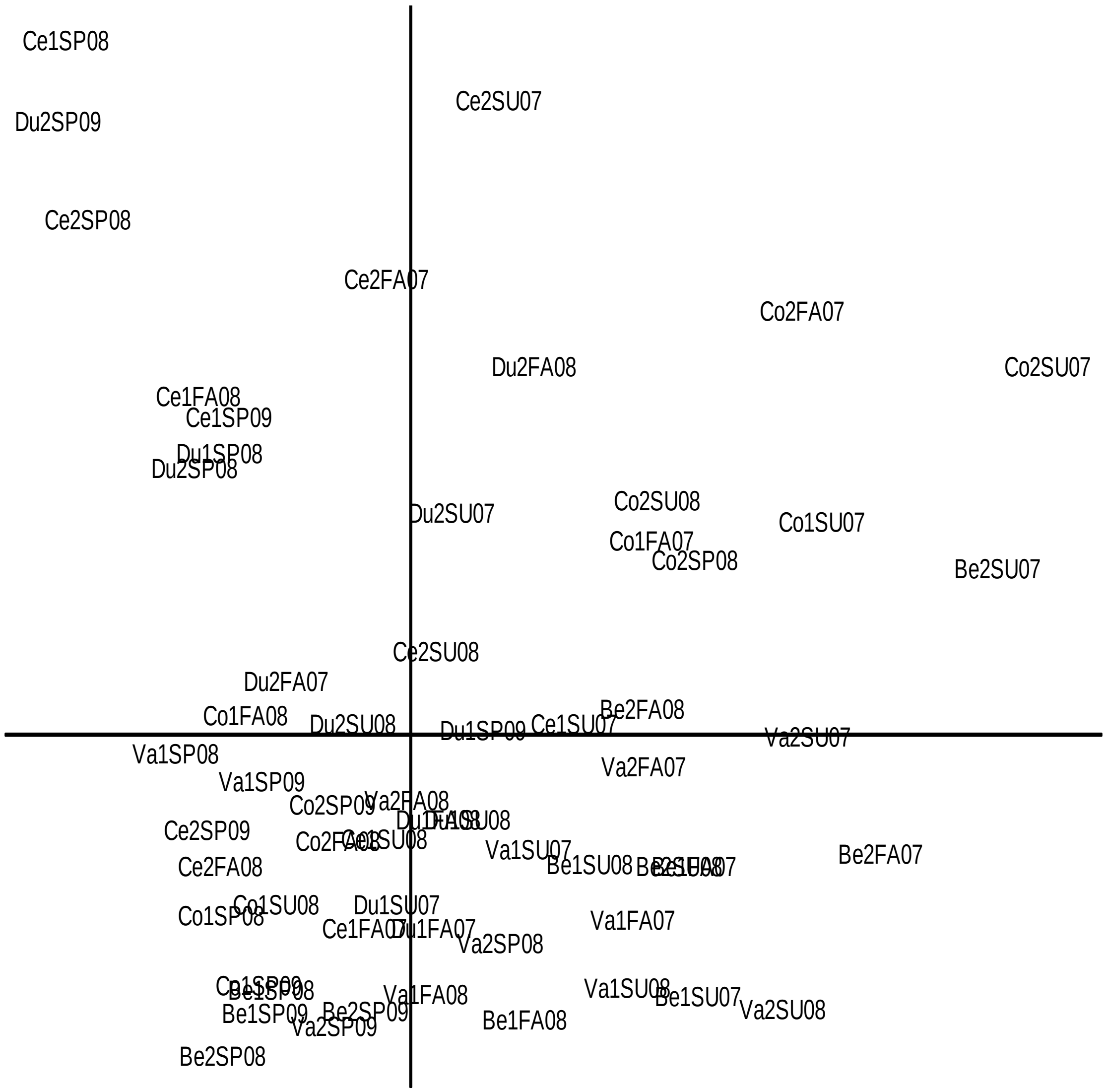

Figure 15 - Ordination plot of a non-metric multidimensional scaling for benthic macroinvertebrate relative abundance in sampling sites in the Ausable River Basin (MVS-ARB). Site codes include a two character code for site (Be = Bear, Ce = Centralia, Co = Colwell, Du = Duncrief, Va = Vail), a number (1 = downstream, 2 = upstream), a two character code for sampling season (SU = summer, FA = fall, SP = spring) and the last two digits of the year sampled.

Table 18 - Principal component loadings for qualitative habitat descriptors at the Ausable River Basin sites (MVS-ARB) (n = 10).

Category	EPA_PC1	EPA_PC2	EPA_PC3
Instream cover	-1.859	2.202	0.076
Pool substrate	-3.155	2.648	0.621
Pool variability	0.668	1.015	-2.119
Sediment deposition	-0.424	1.007	-1.946
Channel flow	0.206	0.433	0.079
Channel Alteration	0.541	-0.056	0.125
Channel sinuosity	0.964	0.361	0.059
Bank stability	-1.191	1.358	0.59
Bank cover	1.833	0.891	-0.056
Riparian width	5.137	2.292	0.653

variables loaded moderately on the PCs for habitat scores at SVS-SWO sites (Table 19). For the MVS-ARB sites, the PCA of physical habitat features produced two PCs, which together explain 73% of the total variation and all variables loaded moderately on the PCs (Table 20). The PCA for water chemistry produced one PC, which explained 81% of the total variation and all variables loaded moderately on the PC (Table 21). The PCA of water quality produced two PCs, which together explained 75% of the total variation and most variables loaded moderately on the PCs (Table 22).

Spearman rank correlations showed that the second PC of US-EPA habitat scores for the SVS-SWO sites was significantly and negatively correlated to enclosedness (r_s = -0.333, n = 157, p < 0.05). All other habitat correlations were not significant (Table 23). Spearman rank correlations showed that the proportion of fish that feed on plants and invertebrates was significantly and positively correlated to enclosedness (r_s = 0.187, n= 157, p < 0.05), while the proportion of fish that feed on invertebrates and fishes was significantly and negatively correlated to enclosedness (r_s = -0.167, n = 157, p < 0.05) for the SVS-SWO sites. All other correlations for fish indices were not significant (Table 24).

Spearman rank correlations showed that the density of invertebrates (r_s = 0.195, n = 157, α = 0.05) and the estimated abundance of invertebrates (r_s = 0.266, n = 157, α = 0.05) in SVS-SWO samples were significantly and positively correlated to enclosedness. All other correlations for invertebrate indices were not significant (Table 25).

Table 19 – Principal component loadings for qualitative habitat descriptors at the southwestern Ontario sites (SVS-SWO) (n = 157).

Category	EPA_PC1	EPA_PC2	EPA_PC3
Instream cover	3.673	-3.216	1.015
Pool substrate	2.522	-2.96	1.237
Pool variability	3.923	-1.499	-2.626
Sediment deposition	2.627	-1.741	0.543
Channel flow	0.276	0.12	-0.624
Channel Alteration	4.672	1.144	-1.24
Channel sinuosity	3.042	1.041	-1.093
Bank stability	0.216	-0.854	1.776
Bank cover	1.794	1.748	0.784
Riparian width	5.09	3.583	1.872

Table 20 – Principal component loadings for physical habitat measurements at the Ausable River Basin sites (MVS-ARB) (n = 10).

Category	PHYSC_PC1	PHYSC_PC2
Mean Width	0.479	0.278
Mean Depth	0.452	-0.749
Mean % Pool	0.96	0.183
Mean % Run	-0.777	-0.526
Mean % Riffle	-0.553	0.685

Table 21 – Principal component loadings for water chemistry parameters at the Ausable River Basin sites (MVS-ARB) (n = 10).

Category	Chem_PC1
Total N (mean)	0.425
Total N (SD)	0.572
Nitrate-N (mean)	0.996
Nitrate-N (SD)	0.926
Ca^{2+} (mean)	0.957
Ca^{2+} (SD)	0.955
Mg^{2+} (mean)	0.975
Mg^{2+}(SD)	0.978
Total P (mean)	0.976
Total P (SD)	0.996
K^{+} (mean)	0.984
K^{+} (SD)	0.972
Na^{+} (mean)	0.931
Na^{+} (SD)	0.751

Table 22 – Principal component loadings for physicochemical water parameters at the Ausable River Basin sites (MVS-ARB) (n = 10).

Category	Qual_PC1	Qual_PC2
Temperature (mean)	0.947	-0.118
Temperature (SD)	-0.005	-0.864
pH (mean)	0.47	-0.595
pH (SD)	0.171	0.887
Conductivity (mean)	0.833	0.426
Conductivity (SD)	0.602	0.631
DO (mean)	0.765	-0.573
DO (SD)	0.896	0.150
Secchi depth (mean)	-0.386	0.504

Table 23 – Spearman rank correlation of habitat variables to enclosedness for Ausable River Basin (MVS-ARB) and southwestern Ontario (SVS-SWO) sites. *Spearman rank correlations are significant for MVS-ARB when $|r_s| >$ 0.648 (n = 10, α = 0.05) and for SVS-SWO when $|r_s| >$ 0.154 (n = 157, α = 0.05) (Zar, 1999).

Variable	MVS-ARB r_s	SVS-SWO r_s
% Agriculture	-0.176	0.049
EPA_PC1	-0.164	0.140
EPA_PC2	0.285	-0.333*
EPA_PC3	0.358	
PHYSC_PC1	0.261	
PHYSC_PC2	0.273	
CHEM_PC1	-0.455	
QUAL_PC1	-0.261	
QUAL_PC2	0.479	

Table 24 – Spearman rank correlation of fish assemblage metrics to enclosedness for Ausable River Basin (MVS-ARB) and southwestern Ontario (SVS-SWO) sites. *Spearman rank correlations are significant for MVS-ARB when $|r_s| > 0.648$ (n = 10, α = 0.05) and for SVS-SWO when $|r_s| > 0.154$ (n = 157, α = 0.05) (Zar, 1999).

Variable	MVS-ARB r_s	SVS-SWO r_s
Species richness	-0.261	0.043
Insectivore	-0.261	-0.048
Herbivore/Insectivore	0.024	0.187*
Insectivore/Piscivore	0.456	-0.167*
Herbivore	-0.164	0.123
% tolerant	-0.91	-0.115
% very tolerant	0.139	-0.026
% intolerant	0.169	0.071

Table 25 – Spearman rank correlation of benthic macroinvertebrate assemblage metrics to enclosedness for Ausable River Basin (MVS-ARB) and southwestern Ontario (SVS-SWO) sites. *Spearman rank correlations are significant for MVS-ARB when $|r_s| > 0.648$ (n = 10, α = 0.05) and for SVS-SWO when $|r_s| > 0.154$ (n = 157, α = 0.05) (Zar, 1999).

Variable	MVS-ARB r_s	SVS-SWO r_s
Richness	0.164	0.094
Density	0.261	0.282*
Estimated Abundance	0.309	0.266*
Collector-filterer	0.382	-0.062
Collector-gatherer	0.164	0.079
Predator	-0.195	0.086
Piercer-herbivore	-0.200	0.008
Scraper-grazer	0.164	-0.015
Shredder	-0.359	0.026
Family biotic index	0.139	-0.139

Logistic regressions for fish species at the MVS-ARB sites showed that increasing enclosedness was related to an increased likelihood of occurrence for one of the 35 species, Stonecat, which had a non-significant, negative relationship to enclosedness at the SVS-SWO sites. In contrast, enclosedness significantly affected the likelihood of occurrence for 10 of the 33 species in the SVS-SWO sites. Of these, the likelihood of finding seven species decreased significantly (Green Sunfish, Bluegill, Rainbow Trout, Common Carp, Central Mudminnow, Brown Trout and Gizzard Shad), while three (Hornyhead Chub, Blacknose Dace and Central Stoneroller) increased significantly with enclosedness. The direction of the relationship was the same for five of seven common species in both the MVS-ARB and SVS-SWO sites (Table 26).

Logistic regressions for BMI species showed that enclosedness significantly affected the likelihood of occurrence for four of the 74 taxa in the MVS-ARB sites and ten of the 94 taxa in the SVS-SWO sites. At the MVS-ARB sites, the likelihood of finding three taxa (Diamesinae, Megaloptera and Torrenticolidae (Acariformes)) increased significantly, while one (Unioncolidae) decreased significantly with enclosedness. At the SVS-SWO sites, the likelihood of finding seven taxa (Diamesinae, Helicopsychidae, Torrenticolidae (Prostigmata), Psephenidae, Asellidae, Leuctridae and Lebertiidae) increased significantly, while three (Tabanidae, Physidae and Ptychopteridae) decreased significantly with enclosedness. The direction of the relationship was the same for five of nine common species in both the MVS-ARB and SVS-SWO sites (Table 27).

Table 26 – Logistic regressions of occurrence of fish species in relation to enclosedness in both Ausable River Basin (MVS-ARB) and southwestern Ontario (SVS-SWO) sites. Only significant relationships and the corresponding relationships from the second study are reported. # indicates species found only at SVS-SWO sites.

	MVS-ARB			SVS-SWO		
Species	**Occurrence (out of 10 sites)**	**Odds ratio**	**p value**	**Occurrence (out of 157 sites)**	**Odds ratio**	**p value**
Stonecat	1	10.090	0.011	1	0.946	0.488
Hornyhead Chub	2	1.108	0.133	9	1.034	0.036
Blacknose Dace	5	1.020	0.700	96	1.039	< 0.001
Green Sunfish	2	0.994	0.926	20	0.970	0.046
Bluegill	2	0.961	0.561	8	0.795	< 0.001
Rainbow Trout	3	0.971	0.603	11	0.956	0.046
Central Stoneroller	9	0.900	0.228	43	1.013	0.044
Common Carp#				13	0.943	0.010
Central Mudminnow#				30	0.972	0.025
Brown Trout#				10	0.933	0.013
Gizzard Shad#				5	0.711	0.002

Table 27 – Logistic regressions of occurrence of benthic macroinvertebrate species in relation to enclosedness in both Ausable River Basin (MVS-ARB) and southwestern Ontario (SVS-SWO) sites. Only significant relationships and the corresponding relationships from the second study are reported. [#] indicates species found only at SVS-SWO sites, [##] indicates species found only at MVS-ARB sites.

	MVS-ARB			SVS-SWO		
Species	**Occurrence (out of 10 sites)**	**odds ratio**	**p value**	**Occurrence (out of 157 sites)**	**odds ratio**	**p value**
Diamesinae	2	21.178	0.002	11	1.031	0.041
Megaloptera	8	37.855	0.002	30	0.998	0.824
Unionicolidae	9	0.099	0.011	13	1.008	0.580
Helicopsychidae	1	1.111	0.228	11	1.034	0.022
Torrenticolidae (prostigmata)	1	1.111	0.228	8	1.034	0.047
Psephenidae	3	1.119	0.080	14	1.027	0.047
Tabanidae	7	0.993	0.900	43	0.977	0.026
Physidae	9	1.012	0.889	81	0.974	0.003
Asellidae	4	0.986	0.784	74	1.024	0.007
[##]Torrenticolidae (acariformes)	1	10.09	0.011			
[#]Leuctridae				1	24.121	0.001
[#]Lebertiidae				91	1.025	0.006
[#]Ptychopteridae				2	0.013	0.014

DISCUSSION

This study showed that a substantial proportion of streams in agricultural southwestern Ontario have been enclosed, suggesting that this practice should be considered a potential threat to aquatic habitat. The natural features of catchments influence, at least partially, the degree of enclosedness; however the decision to enclose a stream is also influenced by social, political and economic factors. Multiple visit sampling of fish and benthic macroinvertebrate (BMI) assemblages may provide a more complete description of biotic assemblages in the Ausable River Basin than single visit sampling (snapshot). The degree of upstream enclosedness had modest effects on fish and BMI assemblages in downstream reaches, however future studies are required to determine the cause of these effects, which may include reduced sedimentation and altered water quality.

Extent of enclosure and characteristics of catchments with varying degrees of enclosedness

Almost 15% of all streams in agricultural southwestern Ontario have been enclosed. This value is slightly lower than those reported in previous studies (e.g. Hietala-Koivu et al. 2004, Elmore and Kaushal 2008); however, this study is the first to describe headwater enclosure over a broad geographical scale with intensive agriculture as the dominant land use. Other studies have been limited to a single, relatively small catchment (e.g. Hietala-Koivu et al. 2004; Meyer and Wallace 2001; Veliz and Sadler Richards 2005) or focused on enclosure in an urban context (e.g. Elmore and Kaushal 2008; Roy et al. 2009). This estimate for southwestern Ontario is also conservative because not all enclosed streams have been located and noted, particularly in larger Conservation Authorities or those with limited resources for field surveys.

A higher proportion of headwater streams (1st and 2nd order) were enclosed than higher-order streams, but there was no relationship between catchment size and enclosedness in southwestern Ontario. Elmore and Kaushal (2008) also found that headwaters were buried more extensively than higher-order streams and that enclosedness decreased with catchment size throughout the Gunpowder-Patapsco catchment. They found that all catchments less than 2.6 km^2 in Baltimore City were ~70% enclosed. In contrast, catchments smaller than 2.6 km^2 in southwestern Ontario ranged from 0 – 100% enclosed. This is likely because these catchments cover a broader range of natural and anthropogenic variation, which is indicative of the complexity of the relationship of enclosedness to the natural landscape and human activity. It is also indicative of the higher occurrence of enclosure in urban areas than in agricultural areas. There are very few large city centers in this area of southwestern Ontario and none as large as Baltimore City. Because it was created after the settlement of most urban centers, the Ontario Stream Network does not display streams that may have been historically enclosed, making it difficult to incorporate urban enclosures resulting in an underestimation of enclosedness.

Catchments with the highest degrees of enclosedness tended to occur in areas with high annual precipitation and high drainage density (km of stream/km^2). Depending upon soil drainage characteristics, enclosing streams in areas with high precipitation may be beneficial because subsurface drainage can reduce peak flows and runoff in response to precipitation events (Robinson 1999). Catchments with high drainage density have more open channels bisecting crops and roadways, which hinders access to cropland. Thus, from a landowner's perspective, enclosing streams in these areas may be beneficial because it creates continuous tracts of accessible cropland (Sadler Richards 2004). The loss of some open channels in these areas may also not be perceived as problematic simply because of their abundance. Depending on the distribution and frequency of precipitation events, there may be periods in which there is insufficient overland flow to keep all of the channels flowing. Landowners may choose to enclose stream channels that tend to run dry because they can create safety risks for tractors and the benefit of accessibility may outweigh the costs of maintaining an open channel (Sadler Richards 2004).

Catchments with the lowest degree of enclosedness tended to occur in areas that receive less annual precipitation, are dominated by either clay plains or undrumlinized till plains, and have shallower water tables. Subsurface drainage has been shown to lower shallow water tables (Hill 1976), which could make water unavailable to plants (Miller and Gardiner 1998) or decrease base flows, particularly in areas that receive less precipitation (Skaggs et al. 2005) making it less beneficial to enclose these streams. Additionally, enclosing a stream in an area with a shallow water table may not be feasible if the water is too near the soil surface. Clay soils have poor infiltration and poor natural drainage, which often makes them unsuitable for high-value crops (Rycroft and Amer 1995). Enclosure in these soils would be less economically justifiable than on more productive lands because the gain in crop yield and accessibility would not outweigh the costs of enclosure (Rycroft and Amer 1995). These areas are also better suited for rearing livestock (Webber and Hoffman 1970) where enclosure would be unnecessary.

Both clay plains and undrumlinized till plains are found in flat landscapes, which, in combination with low rainfall, would result in a lower drainage density. Because there would be fewer streams bisecting croplands, there would be less need to enclose streams. These results are similar to Yates and Bailey (2006), who found that the proportion of enclosed channels in the Upper Thames catchment were positively associated with drumlinized till plains and high drainage density.

Catchments classified using the decision rules determined by the regression tree analysis were moderately well assigned to terminal nodes based upon actual enclosedness values. This suggests that the regression tree only accounts for a proportion of the total variation in catchment characteristics and enclosedness. The remaining unexplained variation may be accounted for by fine scale landscape, economic, social and/or political factors. Large, higher order streams may simply be more expensive to enclose than smaller streams or may be better protected by environmental legislations (Elmore and Kaushal 2008), regardless of the characteristics of the catchment. Additionally, land use is described at a very coarse, regional scale, which may miss detail that is critical in determining whether or not streams are enclosed (Yates and Bailey 2010b). This readily available information reports the amount of land used for agriculture, but does not account for differences such as crop type, management practices or individual field size and shape, which are better indicators of the intensity of agricultural practices in general (Herzog et al. 2006) and can influence the decision to enclose a stream (Sadler Richards 2005; Yates and Bailey 2006).

There are also many socio-economic factors that may affect the decision to enclose a stream (Hietala-Koivu 2002). Landowners may consider the costs associated with enclosing a stream compared to regular open drain maintenance, where the initial costs for construction may be higher but ongoing maintenance costs much lower (Sadler Richards 2004; Van der Gulik et al. 2000) . Enclosing a stream may also result in savings associated with reduced regulatory requirements under Ontario's *Nutrient Management Act* (2002), which restricts the application of fertilizer and pesticides near surface water. Enclosed streams are considered an indicator of agricultural modernization (Hietala-Koivu 2002) because they allow easier access to otherwise segregated fields and increase land value through larger crop size and the use of grassed waterways (Sadler Richards 2005). Hietala-Koivu et al. (2004) determined that further enclosures in their study site in Finland would not be warranted because the social costs of decreased landscape and species diversity and increased nutrient runoff would be greater than the increased private profit. There are many additional site specific physical factors (Irwin 1997a), and socio-political factors (Irwin 1997b; Walters and Shrubsole 2003) that determine whether an enclosure can occur. Landowners' personal decisions to enclose streams are an important source of variability (Roy et al. 2009) that have not been accounted for in our descriptive model.

This study adds to the growing body of literature documenting the extent of headwater enclosure, which is now recognized as a potentially serious threat to aquatic ecosystems (Meyer and Wallace 2001; Wenger et al. 2009). It also provides new information about the physical conditions under which enclosure is likely to occur, although socio-economic factors may also influence enclosure decisions. Much of the recent focus on enclosure has occurred in urban areas (Elmore and Kaushal 2008; Roy et al. 2009; Wenger et al. 2009), where it is even more common than in areas with intense agricultural land use. Small urban streams are enclosed at virtually the same extent as dry land (Elmore and Kaushal 2008), whereas, enclosedness is highly variable in agricultural areas. Studying agricultural streams provides an opportunity to determine the effects of enclosedness on aquatic biota and their habitats before more streams are lost to urbanization.

Spatial and temporal variation of fish and BMI assemblages in stream agro-ecosystems with varying degrees of enclosedness

Streams in agricultural southwestern Ontario experience distinct seasons, often freezing in the winter, flooding in the spring and fall and having reduced flows in the summer to which fish and BMI assemblages are known to respond (Gorman and Karr 1978; Mazor et al. 2006). They are also exposed to a variety of agricultural stressors including crops (e.g. corn, wheat, soy), livestock rearing (e.g. cattle, chicken, hogs), herbicide, pesticide and fertilizer application, agricultural drainage and stream enclosure. Each stream catchment is described by a unique set of these stressors, which can affect fish and BMI assemblages (Yates and Bailey 2010b). To assess the effects of enclosedness on aquatic biota, it is necessary to first understand the spatial and temporal variation of assemblages in streams in this area. Enclosedness is a static value for a catchment, but varies spatially among streams and sites; however, the potential effects may differ seasonally.

At the sites in the Ausable River Basin (MVS-ARB), fish richness and abundance and BMI richness, density and diversity varied more with sampling date than among streams, suggesting that multiple visit sampling does provide additional, important information about stream assemblages than single visit sampling. Differences in these metrics across sampling dates may be explained by several factors. Fish species richness was lowest in the spring and fall of 2007, which was a relatively dry year, and highest in the spring of 2008. Fish abundance tended to be highest at Colwell, which was dominated by small-bodied cyprinid species, but showed no consistent seasonal trend among sites. Spring BMI samples were dominated by the chironomid sub-families Chironominae and Orthocladiinae, which made up, on average, 75% of the total number of BMI sampled. This could account for low BMI richness, diversity and high density in spring samples (Merritt and Cummins 1984).

BMI assemblages based on relative abundances showed seasonal variation regardless of sampling site. The BMI assemblages collected in spring and summer were distinct and showed very little overlap with one another. The BMI assemblages collected in fall occupied the ordination space between spring and summer assemblages and overlapped greatly with both. There was also considerable assemblage variation between sampling years at some sites. This suggests that sampling throughout and across years provides a more complete description of the entire possible BMI assemblage in these streams. If sampling is restricted to one sample per year, it should be taken in the fall to get the best representation of the assemblage. These results are similar to several other studies that have shown high inter- and intra-annual variability in BMI assemblages and descriptive metrics (Beche et al. 2006; Bonada et al. 2006; Mazor et al. 2009). This is likely reflective of the various life histories of BMI, with emergence and oviposition occurring at different times during the year (Merritt and Cummins 1984).

Fish assemblages based on occurrence also showed considerable variation with time, although not in a predictable seasonal manner. Instead, the assemblage in each stream appeared to have a central composition about which the assemblage fluctuated.

There was some overlap of assemblages especially for Vail and Bear, which are closest in physical proximity to one another, and have similar physical characteristics and similar degrees of enclosedness. The upstream site on Ducrief occupies a unique space on the ordination plot, which is indicative of the coldwater species (Brook Trout and Mottled Sculpin) found only at this site. The lack of variation at the sites in Colwell may be reflective of their consistently poor water quality (see below). Other recent studies have also found that, while fish assemblages have high annual and seasonal variation, more variation occurs between streams and sites (Adams et al. 2004; Growns et al. 2006; Meador and Matthews 1992), which suggests that the underlying differences between streams are stronger determinants of fish assemblage compositions than sampling time.

Stammler et al. (in prep) sampled fishes at the MVS-ARB sites once every three weeks from May-July 2009. Their results show that fish species richness and abundance were most variable between streams and sites, with sampling time contributing little to the overall variation. These results suggest that sampling during different seasons does provide a more complete assessment of the fish assemblages at a site, but that sampling once per season may be adequate. In a similar study that sampled fish assemblages monthly in the summer and fall, Growns et al. (2006) found that sampling at any time in a single season is sufficient to describe fish assemblages; however, they did not include spring sampling in their study, which may exclude species using streams for spawning habitat.

Measures of tolerance in both fish and BMI assemblages varied most among streams or sites nested within streams. Family Biotic Index (FBI), a measure of BMI tolerance to organic pollution (Hilsenhoff 1988a) was highest in Colwell and lowest in

Centralia and Duncrief. The proportion of very tolerant and tolerant fishes was highest in Colwell and Centralia. The highest proportion of intolerant fishes was found in Duncrief and few intolerant species were found in any of the other sites. This indicates that water quality is consistently poorest in Colwell and highest in Duncrief. According to Hilsenhoff (1987), the water quality at Colwell would be considered 'Fairly Poor' and the upstream site of Duncrief would be considered 'Good'. All other sites would be classified as 'Fair'. Kosnicki and Sites (2011) and Mazor et al. (2009) also found that Family Biotic Index had low temporal variability, even though BMI assemblages showed some seasonal variation. In contrast, Hilsenhoff (1988b) and Linke et al. (1999) found that FBI varied seasonally, mainly due to the increased presence of coldwater species in the fall, which indicate good water quality.

These results suggest that the FBI and proportion of tolerant fishes are indicative of the local conditions at each site and that the species living there are adapted to the specific conditions at each site (Collier 2008). The surrounding land use at all sites was row crop agriculture consisting mainly of corn, wheat or soy. Colwell also had a small amount of pasture, which may have contributed to poor water quality and impacted biotic assemblages (Fitch and Adams 1998). In Centralia, high nutrient concentrations may have been due to vegetable crops (e.g. red peppers, cabbage), which require high inputs of fertilizer, and the presence of a large greenhouse upstream of both sampling sites, which may have contributed effluent to the stream. Although Duncrief had the highest degree of enclosedness, there was also a higher proportion of forest in the catchments than most of the other sites and it is the only stream in which coldwater indicator fish species were consistently found.

It has long been known that fish and BMI assemblages display seasonal and inter-annual variability (Gorman and Karr 1978; Mazor et al. 2006). However, most bioassessment and biomonitoring studies only take one sample per site and if replicates are taken, they are often only used for Quality Assurance / Quality Control and are not included in data analysis (Carter and Resh 2001). Recent studies suggest that single "snapshot" sampling may not be sufficient to accurately determine the effects of a particular anthropogenic stressor (Collier 2008; Linke et al. 1999; Mazor et al. 2009). Linke et al. (1999) studied streams in the Upper Thames catchment of southwestern Ontario and found that snapshot sampling may miss important variation in a BMI community at a site and that the predictive power of models was increased by including season. The results from the current study support these findings, and suggest that when using fishes and BMI for assessing the effects of human activity on agricultural streams, multiple sampling visits to a site are necessary to obtain a more complete picture of the fish and BMI species assemblages, but that single visit sampling may be sufficient to determine metrics of tolerance.

Downstream habitat and biota correlations to enclosedness

At small and large spatial scales, with either multiple or single visit sampling, there were few effects of enclosedness on the downstream biota or measured habitat features in agricultural streams. Enclosedness was expected to affect stream discharge, nutrient processing and physical habitat, which would be reflected by detectable changes in measures of the fish and BMI assemblages in downstream reaches. However, there were no significant correlations with enclosedness at the MVS-ARB sites for any of the habitat (qualitative, physical, chemical or physicochemical), fish assemblage or BMI assemblage metrics tested. The majority of variables tested (3 of 4 qualitative habitat features, 6 of 8 fish assemblage metrics and 8 of 10 invertebrate metrics) for the southwestern Ontario sites (SVS-SWO) also showed no significant correlations with enclosedness. Additionally, the odds of finding most fish and invertebrate taxa showed no significant relationship to enclosedness in MVS-ARB sites (104 of 109 taxa) or SVS-SWO sites (106 of 127 taxa).

Among SVS-SWO sites, enclosedness was negatively correlated with one qualitative measure of habitat (EPA-PC2), which implies that catchments with high enclosedness have narrow riparian corridors at the reach scale and have high quality instream habitat with a mixture of substrate materials (e.g. gravel, firm sand, aquatic vegetation). EPA_PC2 is counterintuitive because large, diverse riparian areas are typically associated with heterogeneous, high quality instream habitat (Gregory et al. 1991). Enclosedness was positively correlated to BMI density and total estimated abundance. If enclosure reduces sedimentation as predicted (Sadler Richards 2004), instream habitat quality could be higher in streams with high enclosedness because reduced sedimentation generally allows for more heterogeneous habitat (Karr and Schlosser 1978; Walser and Bart 1999). BMI density and abundance may increase as a result of this increased habitat quality through sediment reduction (Doledec et al. 2006; Walser and Bart 1999). These BMI metrics are also known to increase with increasing nutrient inputs associated with high agricultural land use (Evans et al. 1995; Rader and Richardson 1994; Royer et al. 2004), which may be exacerbated by enclosedness.

The proportion of fish species that feed upon plants and invertebrates was positively correlated to enclosedness while the proportion of fish species that feed upon invertebrates and fishes was negatively correlated to enclosedness. Similar shifts in fish feeding guilds have been seen in other studies in agricultural areas. Rashleigh (2004) found a shift from specialized to generalized insectivores with increasing agriculture and found piscivorous fishes (those that eat other fishes) only in areas with a high degree of forest cover. Piscivorous fishes have also been found to decrease with increasing agricultural eutrophication (Egertson and Downing 2004) and with increasing sedimentation (Shields et al. 2007). The increase in smaller-bodied plant/invertebrate feeders could be due to a release from predation by piscivorous fishes. Therefore, it is possible that piscivores, and consequently, prey fishes are responding to the overall effects of agricultural stressors rather than to a particular effect of enclosedness.

Although significant logistic relationships were observed between enclosedness and some fish and invertebrate taxa at the MVS-ARB sites, these results were likely influenced by small sample size (n = 10) and the capture of rare species. Stonecat (*Noturus flavus)* was found on only one occasion at the site with the highest enclosedness (Duncrief downstream), and significant relationships for BMI taxa can be accounted for by the presence or absence of the taxa from either Centralia or Duncrief, which have the lowest and highest enclosedness, respectively. Similarly, many of the fish and BMI taxa with significant logistic relationships to enclosedness at the SVS-SWO sites were found at a small percentage of sites, making it difficult to report with confidence that these relationships are, in fact, due to enclosedness. For these reasons, only the significant relationships for species found at 25 - 75% of SVS-SWO sites and the corresponding relationships from the MVS-ARB sites will be further discussed.

The likelihood of finding Blacknose Dace (*Rhinichthys atratulus)* and Central Stoneroller (*Campostoma anomalum)* increased with increasing enclosedness at the SVS-SWO sites. Although the relationships were not significant, the likelihood of finding Blacknose Dace also increased with enclosedness in MVS-ARB sites, but the likelihood of finding Central Stoneroller decreased. Both Blacknose Dace and Central Stoneroller prefer streams with high gradients and cool, clear water and are benthivorous fishes tolerant of water quality variations. Blacknose Dace is insectivorous while Central Stoneroller is herbivorous (McKee and Parker 1982; Scott and Crossman 1973).

In the SVS-SWO sites, the likelihood of finding Tabanidae (Diptera) and Physidae (Gastropoda) decreased with increasing enclosedness at the SVS-SWO sites and the likelihood of finding Asellidae (Isopoda) and Lebertiidae (Prostigmata) increased with increasing enclosedness. Although the relationships were not significant, the likelihood of finding Tabanidae also decreased with enclosedness in MVS-ARB sites, but the likelihood of finding Asellidae decreased. Identifying BMI to the level of family is sufficient for bioassessment (Bailey et al. 2001); however, the ability to explain the observed trends based on species-specific traits is difficult (Lenat and Resh 2001). Based on available family-level information, Tabanidae and Physidae both prefer silt habitat and tend to live in pools, while Asellidae and Lebertiidae prefer cool, headwater streams with rocky habitat and tend to live in riffles (Vieira et al. 2006).

In their analysis of SVS-SWO data, Yates and Bailey (2010a) found that both fish species and all four BMI taxa tended to be found in streams with moderately high agricultural activity, but did not show a distinct response to an environmental gradient largely determined by surficial geology. Similar to the correlations of enclosedness to habitat quality and BMI density and estimated abundance, these logistic relationships may be in response to agricultural land use or to the predicted reduction of sediment by enclosure (Sadler Richards 2004). If sedimentation is reduced by enclosing streams, the benthivorous fishes and BMI preferring rocky substrate may prefer streams with high enclosedness, while the BMI species preferring silty habitat may prefer streams with low enclosedness. Additionally, the increased likelihood of finding Central Stoneroller could be due to an increase in algal density as a result of increased nutrient inputs commonly associated with agricultural inputs (Evans et al. 1995; Rashleigh 2004) and potentially exacerbated by enclosedness.

Although the determination of causal links between enclosedness and its ecosystem effects is beyond the scope of this study, many of the observed correlations to enclosedness point to a role of sediment reduction. Enclosing a stream may decrease sedimentation because the stream no longer passes through an open channel and, thus, does not erode stream banks or receive surface runoff (Sadler Richards 2004). There is no direct evidence to support this prediction to date; however, subsurface tile drains have been shown to decrease sedimentation in agricultural areas (Blann et al. 2009; Evans et al. 1995; Skaggs et al. 1994). While sediment is the biggest source of pollution in agricultural streams (Walser and Bart 1999), enclosing a stream should not be viewed as an acceptable manner to reduce sedimentation because of the direct loss of habitat, which is a violation of the *Fisheries Act* (1985). There are also many unknown effects of enclosedness on water quality and discharge which could outweigh the benefits of sediment reduction. Stream rehabilitation is well studied and several methods have been shown to increase instream habitat and reduce downstream sedimentation without causing detrimental habitat loss (Makarewicz et al. 2009; Shields et al. 2007). Additionally, many other benefits of sediment reduction, such as the return of specialist fishes and an increase in intolerant BMI species (Doledec et al. 2006; Shields et al. 2007), were not observed in relation to an increase in enclosedness in this study.

The effects of agricultural practices on water quantity and quality, and the consequent effect on fish and BMI assemblages, are well known (Blann et al. 2009; Doledec et al. 2006; Skaggs et al. 1994; Smiley et al. 2009; Walser and Bart 1999). For example, intolerant species are replaced by resilient species able to tolerate the highly altered conditions of agricultural streams (Lenat 1984; Miltner and Rankin 1998), fish assemblages become dominated by habitat and feeding generalists (Doledec et al. 2006; Rashleigh 2004) and shredders are lost from BMI assemblages (Dance and Hynes 1980). The streams in both the MVS-ARB and SVS-SWO study were located in catchments with an average of 80% agricultural land use. It is, therefore, possible that all of these streams have been exposed to agricultural activity (e.g. loss of riparian vegetation, drainage, inputs of fertilizers and pesticides) for a sufficient period of time that the fish and BMI assemblages have adjusted to the resulting conditions (Chambers et al. 2010), making it difficult to detect the signal of enclosedness as a single factor in an area with intense agricultural activity. Other recent studies were also unable to detect differences in fish or invertebrate assemblages due to specific stressors in agricultural streams (e.g. Bt toxins, local upland land use, agricultural drainage) (Chambers et al. 2010; Nerbonne and Vondracek 2001; Stammler et al. 2008).

Enclosure occurs as part of a suite of agricultural activities that together define the stressors to aquatic ecosystems (Blann et al. 2009; Cuffney et al. 2000; Yates 2008).

Although agricultural land use was not correlated to enclosedness, high enclosedness always occurred in catchments with a high proportion of agricultural land use. Therefore, it is necessary to study the effects of enclosure in the greater context of an agricultural setting (Stammler et al. 2008).

While an experiment with a before-after-control-impact design (Underwood 1992) may be able to isolate the effects of enclosure from other stressors**,** the results of such studies would be applicable only in the specific conditions under which they were conducted (Breitburg et al. 1998). Additionally, this study used a catchment-scale measurement of enclosedness, which necessitates an observational approach because of logistic restraints (Power et al. 1998). The catchments in the current study have varying degrees of enclosedness and cover a broad range of agricultural activities (e.g. crop type, management practices, field tile drainage). While the broader effects of agricultural practices may confound the effects of enclosedness, this design allows for a realistic analysis of the effects of enclosedness within an agricultural landscape and the results are broadly applicable to other agricultural areas in temperate climates.

This study is the first to explicitly test the effects of stream enclosure on downstream aquatic biota. By using a combination of multiple visit sampling of sites in a relatively small area and single visit sampling of sites over a much broader geographical area, I was able to use the complementary strengths of both datasets to counteract their weaknesses. Multiple visit sampling accounts for the potential for seasonal effects of enclosedness, but has low explanatory power, while single visit sampling is more broadly applicable and has higher explanatory power, but cannot account for temporal variability. While it is difficult to determine conclusive causal relationships from observational studies (Clements et al. 2002), this is an appropriate starting point and will help to focus further research into the effects of stream enclosure, such as those on discharge, sedimentation and water quality.

SUMMARY

Almost 15% of the total length of agricultural streams in southwestern Ontario has been enclosed. This conservative estimate of the degree of enclosedness will increase as further enclosures occur in agricultural areas and urban development continues to encroach upon agricultural land. Although the decision to bury headwater streams depends partially on features of the natural landscape, individual landowners may ultimately decide whether or not to enclose a stream based on a variety of social, economical and political factors.

Based on sets of sites in southwestern Ontario visited multiple times (ARB) or only once (SWO), it was found that enclosedness had modest effects on fish and BMI assemblages in downstream reaches. Although it was difficult to isolate specific effects of enclosedness on downstream ecosystems, it may act in synergy with a larger suite of agricultural stressors and, thus, should be considered a potential threat to aquatic ecosystems. This study assessed only a subset of the possible ecosystem effects of enclosedness and, therefore, should be seen as a first step towards a better understanding. Further studies are required to determine whether enclosedness does effect sedimentation and nutrient concentrations and to determine the trade-offs of these effects versus direct habitat loss. Multiple visit sampling in streams in the Ausable River Basin also demonstrated that fish and BMI assemblages vary substantially both spatially and temporally, suggesting that studies assessing anthropogenic effects on streams in agricultural areas should incorporate multiple sampling visits across several seasons.

Currently, Ontario is the only jurisdiction in North America with a management protocol in place for considering proposals for stream enclosure. It is recommended that these continue to be assessed on a case-by-case basis, given the unique natural and anthropogenic conditions under which each enclosure occurs. Other agricultural jurisdictions in which enclosure occurs should also consider adopting a similar strategy. However, approval of any enclosure project must also include consideration of the proportion of the stream network already enclosed in a catchment. Although the downstream effects noted in this study were modest, it is clear that enclosedness does affect downstream biotic assemblages. Stronger emphasis should be placed on protecting headwater streams from enclosure, not only because of the direct loss of habitat, but also because the effects on downstream water quality and quantity, and the ecosystem consequences of these effects, are unknown and may be significant. It is, therefore, necessary in this case to adopt the precautionary principle until such information becomes available.

CONCLUSIONS AND RECOMMENDATIONS

The enclosure of headwater streams in southwestern Ontario is common and likely to increase with continued agricultural intensification and expansion of urban areas. Enclosedness is associated with certain features of the physical landscape, but is also strongly influenced by socio-economic factors. Using a combined multiple visit and single visit sampling approach provided the best evidence of the effects of enclosedness by accounting for both spatial and temporal variation. Enclosedness does have discernable effects on downstream fish and benthic macroinvertebrate assemblages and is, therefore, a potential threat to aquatic ecosystems beyond the direct loss of headwater habitat. Management decisions of future applications for stream enclosure should consider the physical landscape features and current degree of enclosedness in order to minimize the effects of this practice.

LITERATURE CITED

Adams, S. B., M. L. Warren & W. R. Haag, 2004. Spatial and temporal patterns in fish assemblages of upper coastal plain streams, mississippi, USA. Hydrobiologia 528(1-3):45-61.

Alexander, R. B., E. W. Boyer, R. A. Smith, G. E. Schwarz & R. B. Moore, 2007. The role of headwater streams in downstream water quality. Journal of the American Water Resourses Association 43:41-59.

Allan, J. D., 1995. Stream ecology: Structure and function of running waters. Chapman & Hall, London, England.

Allan, J. D. & L. B. Johnson, 1997. Catchment-scale analysis of aquatic ecosystems. Freshwater Biology 37(1):107-111.

Ausable River Recovery Team, 2005. Recovery strategy for species at risk in the ausable river: An ecosystem approach, 2005-2010. Draft recovery strategy submitted to Renew Secretariat. ABCA, Exeter, ON.

Bailey, R. C., R. H. Norris & T. B. Reynoldson, 2001. Taxonomic resolution of benthic macroinvertebrate communities in bioassessments. Journal of the North American Benthological Society 20:280-286.

Bailey, R. C., T. B. Reynoldson, A. G. Yates, J. Bailey & S. Linke, 2007. Integrating stream bioassessment and landscape ecology as a tool for land use planning. Freshwater Biology 52(5):908-917.

Barbour, M. T., J. Gerritsen, B. D. Snyder & J. B. Stribling, 1999. Rapid bioassessment protocols for use in streams and wadeable rivers: Periphyton, benthic macroinvertebrates and fish (EPA 841-b-99-002). 2nd edition, U.S. Environmental Protection Agency, Office of Water, Washington, DC.

Beche, L. A., E. P. McElravy & V. H. Resh, 2006. Long-term seasonal variation in the biological traits of benthic-macroinvertebrates in two mediterranean-climate streams in California, USA. Freshwater Biology 51(1):56-75.

Bernot, M. J., J. L. Tank, T. V. Royer & M. B. David, 2006. Nutrient uptake in streams draining agricultural catchments of the midwestern united states. Freshwater Biology 51:499-509.

Beyer, H. L., 2006. Hawth's analysis tools v3.24.

Beyer, H. L., 2010. Geospatial modelling environmental, version 0.3.4 beta. Spatial Ecology LLC.

Blann, K. L., J. L. Anderson, G. R. Sands & B. Vondracek, 2009. Effects of agricultrual drainage on aquatic ecosystems: A review. Critical Reviews in Environmental Science and Technology 39:909-1001.

Bonada, N., M. Rieradevall, N. Prat & V. H. Resh, 2006. Benthic macroinvertebrate assemblages and macrohabitat connectivity in Mediterranean-climate streams of northern California. Journal of the North American Benthological Society 25(1):32-43.

Boulton, A. J., C. G. Peterson, N. B. Grimm & S. G. Fisher, 1992. Stability of an aquatic macroinvertebrate community in a multiyear hydrologic disturbance regime. Ecology 73(6):2192-2207.

Breiman, L., J. H. Friedman, R. A. Olshen & C. J. Stone, 1984. Classification and regression trees. Wadsworth International Group, Belmont, CA.

Breitburg, D. L., J. W. Baxter, C. A. Hatfield, T. W. Howarth, C. G. Jones, G. M. Lovett & C. Wigand, 1998. Understanding effects of multiple stressors: Ideas and challenges. In Pace, M. L. & P. M. Groffman (eds) Successes, limitations, and frontiers in ecosystem science. Springer-Verlag, New York, NY.

Canada Soil Information Service, 2008a. Canada land inventory, level 1. Soil capability for agriculture. Available at: Http://sis.Agr.Gc.Ca/cansis/nsdb/cli/class.Html.

Canada Soil Information Service, 2008b. Soil drainage class. Available at: Http://sis.Agr.Gc.Ca/cansis/nsdb/detailed/name/drainage.Html.

Carpenter, S. R., 1996. Microcosm experiments have limited relevance for community and ecosystem ecology. Ecology 77:677-680.

Carpenter, S. R., 1998. The need for large-scale experiments to assess and predict the response of ecosystems to perturbation. In Pace, M. L. & P. M. Groffman (eds) Successes, limitations, and frontiers in Ecosystem Science. Springer-Verlag Inc., New York, NY.

Carter, J. L. & V. H. Resh, 2001. After site selection and before data analysis: sampling, sorting, and laboratory procedures used in stream benthic macroinvertebrate monitoring programs by USA state agencies. Journal of the North American Benthological Society 20(4):658-682.

Chambers, C. P., M. R. Whiles, E. J. Rosi-Marshall, J. L. Tank, T. V. Royer, N. A. Griffiths, M. A. Evans-White & A. R. Stojak, 2010. Responses of stream macroinvertebrates to bt maize leaf detritus. Ecological Applications 20(7):1949-1960.

Chapin, F. S., E. S. Zavaleta, V. T. Eviner, R. L. Naylor, P. M. Vitoursek, H. L. Reynolds, D. U. Hooper, S. Lavorel, O. E. Sala, S. E. Hobbie, M. C. Mack & S. Diaz, 2000. Consequences of changing biodiversity. Nature 405:234-242.

Clarke, A., R. Mac Nally, N. Bond & P. S. Lake, 2008. Macroinvertebrate diversity in headwater streams: A review. Freshwater Biology 53(9):1707-1721.

Clements, W. H., D. M. Carlisle, L. A. Courtney & E. A. Harrahy, 2002. Integrating observational and experimental approaches to demonstrate causation in stream biomonitoring studies. Environmental Toxicology and Chemistry 21(6):1138-1146.

Coker, G. A., C. B. Portt & C. K. Minns, 2001. Morphological and ecological characteristics of Canadian freshwater fishes. Canadian manuscript report of fisheries and aquatic sciences 2554.

Collier, K. J., 2008. Temporal patterns in the stability, persistence and condition of stream macroinvertebrate communities: Relationships with catchment land-use and regional climate. Freshwater Biology 53(3):603-616.

Cooper, S. D., S. Diehl, K. Kratz & O. Sarnelle, 1998. Implications of scale for patterns and processes in stream ecology. Australian Journal of Ecology 23(1):27-40.

CSPI, 2009. Handbook of steel drainage & highway construction. Corrugated Steel Pipe Institute, Cambridge, ON.

Cuffney, T. F., M. R. Meador, S. D. Porter & M. E. Gurtz, 2000. Responses of physical, chemical, and biological indicators of water quality to a gradient of agricultural land use in the Yakima River Basin, Washington. Environmental Monitoring and Assessment 64(1):259-270.

Curry, R. A., C. Brady, D. L. G. Noakes & R. G. Danzmann, 1997. Use of small streams by young brook trout spawned in a lake. Transactions of the American Fisheries Society 126:77-83.

Dance, K. W. & H. B. N. Hynes, 1980. Some effects of agricultural land-use on stream insect communities. Environmental Pollution Series a-Ecological and Biological 22(1):19-28.

Danz, N. P., R. R. Regal, G. J. Niemi, V. J. Brady, T. Hollenhorst, L. B. Johnson, G. E. Host, J. M. Hanowski, C. A. Johnston, T. Brown, J. Kingston & J. R. Kelly, 2005. Environmentally stratified sampling design for the development of Great Lakes environmental indicators. Environmental Monitoring and Assessment 102(1-3):41-65.

Data East, L., 2010. Xtools pro for arcgis desktop, version 7.0.0. Novosibirsk, Russia.

De'ath, G. & K. E. Fabricius, 2000. Classification and regression trees: A powerful yet simple technique for ecological data analysis. Ecology 81(11):3178-3192.

Department of Defense, 2007. Federal Register (72 fr 11092), reissuance of nationwide permits. Us army corps of engineers.

DFO, 2010. What you should know about fish habitat and a class authorization system for the maintenance of agricultural municipal drains in Ontario. Available at http://www.Dfo-mpo.Gc.Ca/regions/central/pub/factsheets-feuilletsinfos-on/l2-eng.Htm.

Dietrich, M. & N. H. Anderson, 2000. The invertebrate fauna of sumer-dry streams in western Oregon. Archiv Fur Hydrobiologie 147:273-295.

Doledec, S., N. Phillips, M. Scarsbrook, R. H. Riley & C. R. Townsend, 2006. Comparison of structural and functional approaches to determining landuse effects on grassland stream invertebrate communities. Journal of the North American Benthological Society 25(1):44-60.

Drainage Act, 1990. R.S.O., c. D.17. Available at http://www.E-laws.Gov.On.Ca/dblaws/statutes/english/90d17_e.Htm.

Dunne, T. & L. B. Leopold, 1978. Water in environmental planning. W.H Freeman and Company, San Francisco, CA.

Eastman, M., A. Gollamudi, N. Stampfli, C. A. Madramootoo & A. Sarangi, 2010. Comparative evaluation of phosphorus losses from subsurface and naturally drained agricultural fields in the Pike River watershed of Quebec, Canada. Agricultural Water Management 97:596-604.

Egertson, C. J. & J. A. Downing, 2004. Relationship of fish catch and composition to water quality in a suite of agriculturally eutrophic lakes. Canadian Journal of Fisheries and Aquatic Sciences 61(9):1784-1796.

Elmore, A. J. & S. S. Kaushal, 2008. Disappearing headwaters: Patterns of stream burial due to urbanization. Frontiers in Ecology and the Environment 6(6):308-312.

Englund, G. & S. D. Cooper, 2003. Scale effects and extrapolation in ecological experiments. Advances in Ecological Research 33:161-213.

ESRI, 2003. Arcmap 9.2. Redlands, California.

ESRI, 2008. Arcmap 9.3. Redlands, California.

ESRI, 2009. Arc hydro, version 1.3. Redlands, CA.

Etnier, D. A., 1972. The effect of annual rechanneling on a stream fish population. Transaction of the American Fisheries Society 2:372-375.

Evans, R. O., R. W. Skaggs & J. W. Gilliam, 1995. Controlled versus conventional drainage effects on water-quality. Journal of Irrigation and Drainage Engineering-Asce 121(4):271-276.

Fisheries Act, 1985. R.S., c. F-14. Available at http://laws.Justice.Gc.Ca/en/f-14/text.Html

Fitch, L. & B. W. Adams, 1998. Can cows and fish co-exist? Canadian Journal of Plant Science 78:191-198.

Fitzpatrick, F. A., B. C. Scudder, B. N. Lenz & D. J. Sullivan, 2001. Effects of multi-scale environmental characteristics on agricultural stream biota in eastern Wisconsin. Journal of the American Water Resources Association 37(6):1489-1507.

Fore, L. S., J. R. Karr & R. W. Wisseman, 1996. Assessing invertebrate responses to human activities: Evaluating alternative approaches. Journal of the North American Benthological Society 15(2):212-231.

Freeman, M. C., C. M. Pringle & C. R. Jackson, 2007. Hydrologic connectivity and the contribution of stream headwaters to ecological integrity at regional scales. Journal of the American Water Resourses Association 43:5-14.

Frissell, C. A., W. J. Liss, C. E. Warren & M. D. Hurley, 1986. A hierarchical framework for stream habitat classification viewing streams in a watershed context. Environmental Management 10(2):199-214.

Gilliam, J. W. & R. W. Skaggs, 1986. Controlled agricultural drainage to maintain water-quality. Journal of Irrigation and Drainage Engineering-Asce 112(3):254-263.

Gomi, T., R. C. Sidle & J. S. Richardson, 2002. Understanding processes and downstream linkages of headwater systems. BioScience 52(10):905-916.

Gorman, O. T. & J. R. Karr, 1978. Habitat structure and stream fish communities. Ecology 59:507-515.

Gotelli, N. J., 2004. A primer of ecological statistics. Sinauer Associates, Inc., Sunderland, MA.

Gregory, S. V., F. J. Swanson, W. A. McKee & K. W. Cummings, 1991. An ecosystem perspective of riparian zones: Focus on links between land and water. BioScience 41:540-551.

Growns, I., K. Astles & P. Gehrke, 2006. Mulitscale spatial and small-scale temporal variation in the composition of riverine fish communities. Environmental Monitoring and Assessment 114:553-571.

Halliwell, D. B., R. W. Langdon, R. A. Daniels, J. P. Kurtenbach & R. A. Jacobson, 1999. Classification of freshwater fish species of the northeastern United States for use in the development of indices of biological integrity, with regional applications. In Simon, T. P. (ed) Assessing the sustainability and biological integrity of water resources using fish communities. CRC Press, Boca Raton, FL.

Herzog, F., B. Steiner, D. Bailey, J. Baudry, R. Billeter, R. Bukacek, G. De Blust, R. De Cock, J. Dirksen, C. F. Dormann, R. De Filippi, E. Frossard, J. Liira, T. Schmidt, R. Stockli, C. Thenail, W. van Wingerden & R. Bugter, 2006. Assessing the intensity of temperate European agriculture at the landscape scale. European Journal of Agronomy 24(2):165-181.

Hietala-Koivu, R., 2002. Landscape and modernizing agriculture: A case study of three areas in Finland in 1954-1998. Agriculture Ecosystems & Environment 91(1-3):273-281.

Hietala-Koivu, R., J. Lankoski & S. Tarmi, 2004. Loss of biodiversity and its social cost in an agricultural landscape. Agriculture Ecosystems & Environment 103(1):75-83.

Hill, A. R., 1976. Environmental impacts of agricultural land drainage. Journal of Environmental Management 4(3):251-274.

Hilsenhoff, W. L., 1987. An improved biotic index of organic stream pollution. Great Lakes Entomologist 20(1):31-39.

Hilsenhoff, W. L., 1988a. 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., 1988b. Seasonal correction factors for the biotic index. Great Lakes Entomologist 21(1):9-13.

Horwitz, R. J., 1978. Temporal variability patterns and the distributional patterns of stream fishes. Ecological Monographs 48(3):307-321.

Hynes, H. B. N., 1975. The stream and its valley. Verhhandlungen der Internationalen Vereinigung für Theoretische und Angewandte Limnologie 19:1-15.

Irwin, R. W., 1997a. Drainage guide for Ontario - publication 29. Ontario ministry of agriculture food and rural affairs.

Irwin, R. W., 1997b. Drainage legislation. Ontario ministry of agriculture food & rural affairs.

Johnson, L. B. & G. E. Host, 2010. Recent developments in landscape approaches for the study of aquatic ecosystems. Journal of the North American Benthological Society 29(1):41-66.

Karr, J. R. & I. J. Schlosser, 1978. Water resources and the land-water interface. Science 201:229-234.

Kerans, B. L., J. R. Karr & S. A. Ahlstedt, 1992. Aquatic invertebrate assemblages - spatial and temporal differences among sampling protocols. Journal of the North American Benthological Society 11(4):377-390.

Klocker, C. A., S. S. Kaushal, P. M. Groffman, P. M. Mayer & R. P. Morgan, 2009. Nitrogen uptake and denitrification in restored and unrestored streams in urban Maryland, USA. Aquatic Sciences 71(4):411-424.

Kosnicki, E. & R. W. Sites, 2011. Seasonal predictability of benthic macroinvertebrate metrics and community structure with maturity-weighted abundances in a Missouri Ozark stream, USA. Ecological Indicators 11(2):704-714 doi:10.1016/j.ecolind.2010.04.008.

Lammert, M. & J. D. Allan, 1999. Assessing biotic integrity of streams: Effects of scale in measuring the influence of land use/cover and habitat structure on fish and macroinvertebrates. Environmental Management 23(2):257-270.

Leibowitz, S. G., P. J. Wigington, M. C. Rains & D. M. Downing, 2008. Non-navigable streams and adjacent wetlands: Addressing science needs following the Supreme Court's Rapanos Decision. Frontiers in Ecology and the Environment 6(7):366-373.

Lenat, D. R., 1984. Agriculture and stream water quality: A biological evaluation of erosion control practices. Environmental Management 8:333-334.

Lenat, D. R. & V. H. Resh, 2001. Taxonomy and stream ecology : The benefits of genus- and species-level identifications. Journal of the North American Benthological Society 20:287-298.

Likens, G. E., 1998. Limitations to intellectual progress in ecosystem science. In Pace, M. L. & P. M. Groffman (eds) Successes, limitations, and frontiers in ecosystem science. Springer-Verlag, New York, NY.

Linke, S., R. C. Bailey & J. A. Schwindt, 1999. Temporal variability of stream bioassessments using benthic macroinvertebrates. Freshwater Biology 42:575-584.

Makarewicz, J. C., T. W. Lewis, I. Bosch, M. R. Noll, N. Herendeen, R. D. Simon, J. Zollweg & A. Vodacek, 2009. The impact of agricultural best management practices on downstream systems: Soil loss and nutrient chemistry and flux to Conesus Lake, New York, USA. Journal of Great Lakes Research 35:23-36.

Mayer, P. M., P. M. Groffman, E. A. Striz & S. S. Kaushal, 2010. Nitrogen dynamics at the groundwater-surface water interface of a degraded urban stream. Journal of Environmental Quality 39(3):810-823.

Mazor, R. D., A. H. Purcell & V. H. Resh, 2009. Long-term variability in bioassessments: A twenty-year study from two northern California streams. Environmental Management 43(6):1269-1286.

Mazor, R. D., T. B. Reynoldson, D. M. Rosenberg & V. H. Resh, 2006. Effects of biotic assemblage, classification, and assessment method on bioassessment performance. Canadian Journal of Fisheries and Aquatic Sciences 63(2):394-411.

McCune, B. & J. B. Grace, 2002. Analysis of ecological communities. MjM Software Design, Gleneden Beach, OR, USA.

McCune, B. & M. J. Mefford, 1999. Multivariate analysis of ecological data, version 4.27. MjM Software, Gleneden Beach, OR, USA.

McIsaac, G. F. & X. Hu, 2004. Net N input and riverine N export from Illinois agricultural watersheds with and without extensive tile drainage. Biogeochemistry 70:251-271.

McKee, P. M. & B. J. Parker, 1982. The distribution, biology, and status of the fishes *campostoma anomalum, clinostomus elongatus, notropis photogenis* (cyprinidae), and *fundulus notatus* (cyprinodontidae) in Canada. Canadian Journal of Zoology 60(6):1347-1358.

Meador, M. R. & W. J. Matthews, 1992. Spatial and temporal patterns in fish assemblage structure of an intermittent texas stream. American Midland Naturalist 127(1):106-114.

Merritt, R. W. & K. W. Cummins, 1984. An introduction to the aquatic insects of North America. Kendall/Hunt Publishing Company, Dubuque, IA, USA.

Meyer, J. L., L. A. Kaplan, D. Newbold, D. L. Strayer & C. J. Woltemade, 2003. Where rivers are born: The scientific imperative for defending small streams and wetlands. American Rivers and Sierra Club.

Meyer, J. L., G. C. Poole & K. L. Jones, 2005. Buried alive: Potential consequences of burying headwater streams in drainage pipes. Proceedings of the 2005 Georgia Water Resources Conference, Athens, GA.

Meyer, J. L., D. L. Strayer, J. B. Wallace, S. L. Eggert, G. S. Helfman & N. E. Leonard, 2007. The contribution of headwater streams to biodiversity in river networks. Journal of the American Water Resourses Association 43:86-103.

Meyer, J. L. & J. B. Wallace, 2001. Lost linkages and lotic ecology: Rediscovering small streams. In Press, M. C., H. N. J. & L. S. (eds) Ecology: Achievement and challenge. 295-317.

Miller, R. W. & D. T. Gardiner, 1998. Soils in our environment, 8 edn. Prentice Hall, Upper Saddle River, NJ.

Miltner, R. J. & E. T. Rankin, 1998. Primary nutrients and the biotic integrity of rivers and streams. Freshwater Biology 40(1):145-158.

Moore, R. D., 2005. Small stream channels and their riparian zones in forested catchments of the pacific northwest: Introduction. Journal of the American Water Resources Association 41(4):759-761.

Nadeau, T. L. & M. C. Rains, 2007. Hydrological connectivity between headwater streams and downstream waters: How science can inform policy. Journal of the American Water Resources Association 43(1):118-133.

Nerbonne, B. A. & B. Vondracek, 2001. Effects of local land use on physical habitat, benthic macroinvertebrates, and fish in the Whitewater River, Minnesota, USA. Environmental Management 28:87-99.

Ontario Geological Survey, 2007. Physiography of southern ontario.

Ontario MInistry of Natural Resources, 2008. Southern ontario land resource information system (solris) land classification data. .

Ontario Ministry of Northern Development Mines and Forestry, 2011. Surficial geology. Available at: Http://www.Mndmf.Gov.On.Ca/mines/ogs/topics/surficial_geology_e.Asp.

Ostrand, K. G. & G. R. Wilde, 2002. Seasonal and spatial variation in a prairie stream-fish assemblage. Ecology of Freshwater Fish 11(3):137-149.

Paller, M. H., 1994. Relationships between fish asemblage structure and steram order in South Carolina coastal plain streams. Transactions of the American Fisheries Society 123:150-161.

Peterson, B. J., W. M. Wollheim, P. J. Mulholland, J. R. Webster, J. L. Meyer, J. L. Tank, E. Marti, W. B. Bowden, H. M. Valett, A. E. Hershey, W. H. McDowell, W. K. Dodds, S. K. Hamilton, S. Gregory & D. D. Morrall, 2001. Control of nitrogen export from watersheds by headwater streams. Science 292:86-90.

Poff, N. L. & J. D. Allan, 1995. Functional organization of stream fish assemblages in relation to hydrological variability. Ecology 76:606-627.

Poff, N. L., J. D. Allan, M. B. Bain, J. R. Karr, K. L. Prestegaard, B. D. Richter, R. E. Sparks & J. C. Stromber, 1997. The natural flow regime: A paradigm for river conservation and restoration. BioScience 47:769-783.

Poff, N. L. & J. V. Ward, 1989. Implications of streamflow variability and predictability for lotic community structure: A regional analysis of streamflow patterns. Canadian Journal of Fisheries and Aquatic Science 46:1805-1815.

Power, M. E., W. E. Dietrich & K. O. Sullivan, 1998. Experimentation, observation, and infernce in river and watershed investigations. In Resetarits, W. J. & J. Bernardo (eds) Experimental ecology - issues and perspectives. Oxford University Press, New York, NY.

Quinn, G. & M. Keough, 2002. Experimental design and data analysis for Biologists Cambridge University Press, Cambridge, UK.

Rader, R. B. & C. J. Richardson, 1994. Response of macroinvertebrates and small fish to nutrient enrichment in the northern Everglades. Wetlands 14(2):134-146.

Rashleigh, B., 2004. Relation of environmental characteristics to fish assemblages in the Upper French Broad River Basin, North Carolina. Environmental Monitoring and Assessment 93(1-3):139-156.

Rasmussen, A. K., 2004. Species diveristy and ecology of Trichoptera (caddisflies) and Plecoptera (stoneflies) in ravine ecosystems of northern Florida. Phd thesis, university of florida, Gainsville, FL.

Reynoldson, T. B., C. Logan, T. Pascoe & S. P. Thompson, 1999. Methods manual : I. Creation of reference-condition databases for benthic invertebrate community structure. NWRiReport no.99-211.

Richards, C., L. B. Johnson & G. E. Host, 1996. Landscape-scale influences on stream habitats and biota. Canadian Journal of Fisheries and Aquatic Sciences 53:295-311.

Robinson, M., 1999. The impact of drainage on stream flows. Institute of hydrology report no. 113. Center for Ecology and Hydrology, Edinburgh, U.K.

Roy, A. H., A. L. Dybas, K. M. Fritz & H. R. Lubbers, 2009. Urbanization affects the extent and hydrologic permanence of headwater streams in a midwestern US metropolitan area. Journal of the North American Benthological Society 28(4):911-928.

Royer, T. V., J. L. Tank & M. B. David, 2004. Transport and fate of nitrate in headwater agricultural streams in Illinois. Journal of Environmental Quality 33:1296-1304.

Rycroft, D. W. & M. H. Amer, 1995. Propects for the drainage of clay soils. Food and Agriculture Organization of the United Nations, Rome, Italy.

Sadler Richards, J., 2004. A review of the enclosure of watercourses in agricultural landscapes and river headwater functions. Cordner Science, Ailsa Craig, ON.

Sadler Richards, J., 2005. The economic importance of agricultural drainage with emphasis on the enclosure of open drains in Ontario. Cordner Science, Aisla Craig, ON.

Schindler, D. W., 1998. Replication versus realism: The need for ecosystem-scale experiments. Ecosystems 1:323-334.

Schlosser, I. J., 1990. Environmental variation, life-history attributes, and community structure in stream fishes - implications for environmental-management and assessment. Environmental Management 14(5):621-628.

Scott, W. B. & E. J. Crossman, 1973. Freshwater fishes of Canada. Fisheries Research Board of Canada, Ottawa, Canada.

Shields, F. D., S. S. Knight & C. M. Cooper, 2007. Can warmwater streams be rehabilitated using watershed-scale standard erosion control measures alone? Environmental Management 40(1):62-79.

Skaggs, R. W., M. A. Breve & J. W. Gilliam, 1994. Hydrologic and water quality impacts of agricultural drainage. Critical Reviews in Environmental Science and Technology 24:1-32.

Skaggs, R. W., G. M. Chescheir & B. D. Phillips, 2005. Methods to determine lateral effect of a drainage ditch on wetland hydrology. Transactions of the American Society of Agricultural Engineers 48(2):577-584.

Smiley, P. C., E. D. Dibble & S. H. Schoenholtz, 2005. Fishes of first-order streams in north-central Mississippi. Southeastern Naturalist 4(2):219-236.

Smiley, P. C., R. B. Gillespie, K. W. King & C. H. Huang, 2009. Management implications of the relationships between water chemistry and fishes within channelized headwater streams in the midwestern United States. Ecohydrology 2(3):294-302.

Stammler, K. L., R. L. McLaughlin & N. E. Mandrak, 2008. Streams modified for drainage provide fish habitat in agricultural areas. Canadian Journal of Fisheries and Aquatic Sciences 65(3):509-522.

Statistics Canada, 2007. Land cover by category, by major drainage area and sub-drainage area, 2000 - St. Lawrence Ottawa, ON, Canada.

StatSoft, 2010. Satistica, version 10. Tulsa, OK.

Stauffer, J. C., R. M. Goldstein & R. M. Newman, 2000. Relationship of wooded riparian zones and runoff potential to fish community composition in agricultural streams. Canadian Journal of Fisheries and Aquatic Sciences 57:307-316.

Stewart, J. S., L. Z. Wang, J. Lyons, J. A. Horwatich & R. Bannerman, 2001. Influences of watershed, riparian-corridor, and reach-scale characteristics on aquatic biota in agricultural watersheds. Journal of the American Water Resources Association 37(6):1475-1487.

Stout, B. & J. B. Wallace, 2003. A survey of eight aquatic insect orders associated with small headwater streams subject to valley fills from mountaintop mining. Appendix in mountaintop mining/valley fills in appalachia. Final programmatic environmental impact statement. Us environmental protection agency, Philadelphia, PA.

Strahler, A. N., 1952. Hypsometric (area-altitude) analysis of erosional tophography. Bulletin of the Geological Society of America 63:1117-1142.

Strayer, D. L., 2000. North America's freshweater invertebrates: A research priority. In Abell, R. A., et al. (eds) Freshwater ecoregions of North America, a conservation assessment. Island, Press, Washington, DC.

Systat, 2004. Systat for windows 11.00.01 edn, Chicago, IL.

The Clean Water Act, 1972. 33 u.S.C. §1251 et seq. Available at http://epw.Senate.Gov/water.Pdf.

The Clean Water Act, 2006. C. 33, sched. 15, s. 2. Available at http://www.E-laws.Gov.On.Ca/html/statutes/english/elaws_statutes_06c22_e.Htm.

The Nutrient Management Act, 2002. Ontario regulation 338/09 available at http://www.E-laws.Gov.On.Ca/html/regs/english/elaws_regs_030267_e.Htm.

Underwood, A. J., 1992. Beyond baci - the detection of environmental impacts on populations in the real, but variable, world. Journal of Experimental Marine Biology and Ecology 161(2):145-178.

Van der Gulik, T. W., L. H. Christl, D. R. Coote, C. A. Madramootoo, T. J. Nyvall & T. J. V. Sopuck, 2000. Managing excess water. In Coote, D. R. & L. J. Gregorich (eds) The health of our water - toward sustainable agriculture in canada. Research Planning and Coordination Directorate, Research Branch, Agriculture and Agri-Food Canada, Ottawa, ON, 121-129.

Van Schilfgaarde, J., 1974. Drainage for agriculture. American Society of Agronomy, Inc., Madison, WI.

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

Veliz, M., H. Brock & J. Neary, 2006. Ausable Bayfield Conservation Authority watershed report card Exeter, ON.

Veliz, M. & J. Sadler Richards, 2005. Enclosing surface drains: What's the story? Journal of Soil and Water Conservation 60(3):70-73.

Vieira, N. K. M., N. L. Poff, D. M. Carlisle, S. R. Moulton, M. L. Koski & B. C. Kondratieff, 2006. A database of lotic invertebrate traits for north america: U.S. Geological survey data series 187.

Wallace, J. B., S. L. Eggert, J. L. Meyer & J. R. Webster, 1997. Multiple trophic levels of a forest stream linked to terrestrial litter inputs. Science 277(5322):102-104.

Walser, C. A. & H. L. Bart, 1999. Influence of agriculture on in-stream habitat and fish community structure in Piedmont watersheds of the Chattahoochee River system. Ecology of Freshwater Fish 8:237-246.

Walters, D. & D. Shrubsole, 2003. Agricultural drainage and wetland management in Ontario. Journal of Environmental Management 69(4):369-379.

Wang, L. Z., J. Lyons, P. Rasmussen, P. Seelbach, T. Simon, M. Wiley, P. Kanehl, E. Baker, S. Niemela & P. M. Stewart, 2003. Watershed, reach, and riparian influences on stream fish assemblages in the northern lakes and forest ecoregion, USA. Canadian Journal of Fisheries and Aquatic Sciences 60(5):491-505.

Webber, L. R. & D. W. Hoffman, 1970. Origin, classification and use of Ontario soils. Publication 51. Ontario department of agriculture and food. Ontario Agricultural College, University of Guelph Guelph, ON.

Wenger, S. J., A. H. Roy, C. R. Jackson, E. S. Bernhardt, T. L. Carter, S. Filoso, C. A. Gibson, W. C. Hession, S. S. Kaushal, E. Marti, J. L. Meyer, M. A. Palmer, M. J. Paul, A. H. Purcell, A. Ramirez, A. D. Rosemond, K. A. Schofield, E. B. Sudduth & C. J. Walsh, 2009. Twenty-six key research questions in urban stream ecology: An assessment of the state of the science. Journal of the North American Benthological Society 28(4):1080-1098.

Werner, E. E., 1998. Ecological experiments and a research program in community ecology. In Resetarits, W. J. & J. Bernardo (eds) Experimental ecology: Issues and perspectives. Oxford University Press, Inc., New York, NY.

Wiley, M. J., S. L. Kohler & P. W. Seelbach, 1997. Reconciling landscape and local views of aquatic communities: Lessons from Michigan trout streams. Freshwater Biology 37(1):133-148.

Wipfli, M. S. & D. P. Gregovich, 2002. Export of invertebrates and detritus from fishless headwater streams in southeastern Alaska: Implications for downstream salmonid production. Freshwater Biology 47:957-969.

Wipfli, M. S., J. S. Richardson & R. J. Naiman, 2007. Ecological linkages between headwaters and downstream exosystems: Transport of organic matter, invertebrates, and wook down headwater channels. Journal of the American Water Resourses Association 43:72-85.

Wiskow, E. & R. R. van der Ploeg, 2003. Calculation of drain spacings for optimal rainstorm flood control. Journal of Hydrology 272(1-4):163-174.

Yates, A. G., 2008. Development, testing and application of stressor gradients in rural, headwater streams in southwestern Ontario. Phd thesis. University of Western Ontario, London, ON.

Yates, A. G. & R. C. Bailey, 2006. The stream and its altered valley: Integrating landscape ecology into environmental assessments of agro-ecosystems. Environmental Monitoring and Assessment 114(1-3):257-271.

Yates, A. G. & R. C. Bailey, 2010a. Covarying patterns of macroinvertebrate and fish assemblages along natural and human activity gradients: Implications for bioassessment. Hydrobiologia 637(1):87-100.

Yates, A. G. & R. C. Bailey, 2010b. Improving the description of human activities potentially affecting rural stream ecosystems. Landscape Ecology 25(3):371-382.

Zar, J. H., 1999. Biological analysis, 4th ed. Prentice Hall, Inc., Upper Saddle River, NJ, USA.

Appendix 1 – United States Environmental Protection Agency Rapid Habitat Assessment Protocol data sheets for low gradient streams (Barbour et al. 1999).

Habitat parameter	**Condition category**			
	Excellent	**Good**	**Fair**	**Poor**
1. Epifaunal substrate / available cover	*Greater than 50% of substrate favourable for epifaunal colonisation and fish cover; mix of snags, submerged logs, undercut banks, cobble or other stable habitat and at stage to allow full colonisation potential (i.e. logs/snags that are not new fall and not transient).*	*30-50% mix of stable habitat; well-suited for full colonisation potential; adequate habitat for maintenance of populations; presence of additional substrate in the form of newfall, but not yet prepared for colonisation (may rate at high end of scale).*	*10-30% mix of stable habitat; habitat availability less than desirable; substrate frequently disturbed or removed.*	*Less than 10% stable habitat; lack of habitat is obvious; substrate unstable or lacking.*
SCORE	**20 19 18 17 16**	**15 14 13 12 11**	**10 9 8 7 6**	**5 4 3 2 1 0**
2. Pool substrate characterization	Mixture of substrate materials, with gravel and firm sand prevalent; root mats and submerged vegetation common.	Mixture of soft sand, mud or clay; mud may be dominant; some root mats and submerged vegetation present.	All mud or clay or sand bottom; little or no root mat; no submerged vegetation.	Hard-pan clay or bedrock; no root mat or vegetation.
SCORE	**20 19 18 17 16**	**15 14 13 12 11**	**10 9 8 7 6**	**5 4 3 2 1 0**
3. Pool variability	Even mix of large-shallow, large-deep, small-shallow, small-deep pools present.	Majority of pools large-deep; very few shallow.	Shallow pools much more prevalent than deep pools.	Majority of pools small-shallow or pools absent.
SCORE	**20 19 18 17 16**	**15 14 13 12 11**	**10 9 8 7 6**	**5 4 3 2 1 0**
4. Sediment deposition	Little or no enlargement of islands or point bars and less than 20% of the bottom affected by sediment deposition.	Some new increase in bar formation, mostly from gravel, sand or fine sediment; 20-50% of the bottom affected; slight deposition in pools.	Moderate deposition of new gravel, sand or fine sediment on old and new bars; 50-80% of the bottom affected; sediment deposits at obstructions, constrictions and bends; moderate deposition in pools prevalent.	Heavy deposits of fine material, increased bar development; more than 80% of the bottom changing frequently; pools almost absent due to substantial sediment deposition.
SCORE	**20 19 18 17 16**	**15 14 13 12 11**	**10 9 8 7 6**	**5 4 3 2 1 0**
5. Channel flow status	Water reaches base of both lower banks, and minimal amount of channel substrate is exposed.	Water fills >75% of the available channel; or <25% of channel substrate is exposed.	Water fills 25-75% of the available channel, and/or riffle substrates are mostly exposed.	Very little water in channel and mostly present as standing pools.
SCORE	**20 19 18 17 16**	**15 14 13 12 11**	**10 9 8 7 6**	**5 4 3 2 1 0**

6. Channel alteration	Channelization or dredging absent or minimal; stream with normal pattern.					Some channelization present, usually in areas of bridge abutments; evidence of past channelization, i.e. dredging (greater than 20 yr) may be present, but recent channelization is not present.					Channelization may be extensive; embankments or shoring structures present on both banks; and 40 to 80% of stream reach channelized and disrupted.					Banks shored with gabion or cement; over 80% of the stream reach channelized and disrupted. Instream habitat greatly altered or removed entirely.					
SCORE	**20**	**19**	**18**	**17**	**16**	**15**	**14**	**13**	**12**	**11**	**10**	**9**	**8**	**7**	**6**	**5**	**4**	**3**	**2**	**1**	**0**

Habitat parameter	Condition category																				
	Excellent					**Good**					**Fair**					**Poor**					
7. Channel sinuosity	The bends in the stream increase the stream length 3 to 4 times longer than if it was in a straight line. (Note – channel braiding is considered normal in coastal plains and other low-lying areas. This parameter is not easily rated in these areas).					The bends in the stream increase the stream length 2 to 3 times longer than if it was in a straight line.					The bends in the stream increase the stream 1 to 2 times longer than if it was in a straight line.					Channel straight; waterway has been channelized for a long distance.					
SCORE	**20**	**19**	**18**	**17**	**16**	**15**	**14**	**13**	**12**	**11**	**10**	**9**	**8**	**7**	**6**	**5**	**4**	**3**	**2**	**1**	**0**

8. Bank stability (score each bank)	Banks stable; evidence of erosion or bank failure absent or minimal; little potential for future problems. <5% of bank affected.			Moderately stable; infrequent, small areas of erosion mostly healed over. 5-30% of bank in reach has areas of erosion.			Moderately unstable; 30-60% of bank in reach has areas of erosion; high erosion potential during floods.			Unstable; many eroded areas; 'raw' areas frequent along straight sections and bends; obvious bank sloughing; 60-100% of bank has erosional scars.		
SCORE	Left bank	**10**	**9**	**8**	**7**	**6**	**5**	**4**	**3**	**2**	**1**	**0**
SCORE	Right bank	**10**	**9**	**8**	**7**	**6**	**5**	**4**	**3**	**2**	**1**	**0**

9. Vegetative protection **(score each bank)**	More than 90% of the streambank surfaces and immediate riparian zone covered by native vegetation, including trees, understorey shrubs, or non woody macrophytes; vegetative disruption through grazing or mowing minimal or not evident; almost all plants allowed to grow naturally.		70-90% of the streambank surfaces covered by native vegetation, but one class of plants is not well-represented; disruption evident but not affecting full plant growth potential to any great extent; more than one half of the potential plant stubble height remaining.			50-70% of the streambank surfaces covered by vegetation; disruption obvious; patches of bare soil or closely cropped vegetation common; less than one-half of the potential plant stubble height remaining.			Less than 50% of the streambank surfaces covered by vegetation; disruption of streambank vegetation is very high; vegetation has been removed to 5 centimetres or less in average stubble height.		
SCORE	Left bank	**10**	**9**	**8**	**7**	**6**	**5**	**4**	**3**	**2**	**1**	**0**
SCORE	Right bank	**10**	**9**	**8**	**7**	**6**	**5**	**4**	**3**	**2**	**1**	**0**
10. Riparian zone score **(score each bank)**	Width of riparian zone >18 metres; human activities (i.e. roads, lawns, crops etc.) have not impacted the riparian zone.		Width of riparian zone 12-18 metres; human activities have impacted the riparian zone only minimally.			Width of riparian zone 6-12 metres; human activities have impacted the riparian zone a great deal.			Width of riparian zone <6 metres; little or no riparian vegetation is present because of human activities.		
SCORE	Left bank	**10**	**9**	**8**	**7**	**6**	**5**	**4**	**3**	**2**	**1**	**0**
SCORE	Right bank	**10**	**9**	**8**	**7**	**6**	**5**	**4**	**3**	**2**	**1**	**0**

Appendix 2 – Complete list of fish species collected and the number and proportion of the 160 sites at which each species was found for SVS-SWO sites from Yates (2008). Note that three of the original sites used by Yates (2008) were not used in the current study.

Species Name	Common Name	# of Sites Present	% of Sites Present
Lampetra appendix	American Brook Lamprey	6	4
Dorosoma cepedianum	Gizzard Shad	5	3
Moxostoma Spp	Redhorse spp.	3	2
Hypentelium nigricans	Northern Hogsucker	4	2
Catostomus commersonii commersonii	White Sucker	106	66
Cyprinus carpio	Common Carp	13	8
Semotilus atromaculatus atromaculatus	Northern Creek Chub	122	76
Nocomis biguttatus	Hornyhead Chub	9	6
Margariscus nachtriebi	Northern Pearl Dace	7	4
Phoxinus neogaeus	Finescale Dace	1	<1
Phoxinus eos	Northern Redbelly Dace	38	24
Rhinichthys obtusus	Western Blacknose Dace	98	61
Rhinichthys cataractae cataractae	Great Lakes Longnose Dace	9	6
Campostoma anomalum pullum	Central Stoneroller	44	28
Cyprinella spiloptera	Spotfin Shiner	15	9
Luxilus cornutus frontalis	Northern Common Shiner	42	26
Luxilus chrysocephalus	Striped Shiner	16	10
Notropis rubellus	Rosyface Shiner	11	7
Notropis volucellus	Mimic Shiner	20	13
Notropis heterolepis	Blacknose Shiner	6	4
Pimephales notatus	Bluntnose Minnow	73	16
Pimephales promelas promelas	Fathead Minnow	35	22
Hybognathus hankinsoni	Brassy Minnow	3	2
Ameiurus melas	Black Bullhead	2	1
Ameiurus nebulosus	Brown Bullhead	6	4
Ameiurus natalis	Yellow Bullhead	6	4
Noturus flavus	Stonecat	1	<1
Salvelinus fontinalis	Brook Trout	14	9
Salmo trutta	Brown Trout	10	6
Orcorhynchus mykiss	Rainbow Trout	11	7
Esox lucius	Northern Pike	4	2
Umbra limi	Central Mudminnow	33	21
Labidesthes sicculus	Northern Brook Silverside	1	<1
Perca flavescens	Yellow Perch	3	2
Percina maculata	Blackside Darter	14	9
Percina caprodes semifasciata	Northern Logperch	1	<1
Etheostoma nigrum nigrum	Central Johnny Darter	91	57
Etheostoma blennioides blennioides	Northern Greenside Darter	8	5
Etheostoma exile	Iowa Darter	12	8
Etheostoma microperca	Least Darter	27	17
Etheostoma caeruleum	Rainbow Darter	15	9

Species Name	Common Name	# of Sites Present	% of Sites Present
Etheostoma flabellare flabellare	Barred Fantail Darter	12	8
Micropterus dolomieu	Smallmouth Bass	5	3
Micropterus salmoides	Largemouth Bass	21	13
Lepomis Cyanellus	Green Sunfish	20	13
Lepomis peltastes	Northern Longear Sunfish	1	<1
Lepomis gibbosus	Pumpkinseed	29	18
Lepomis macrochirus macrochirus	Bluegill	8	5
Ambloplites rupestris	Northern Rockbass	21	13
Cottus Spp.	Sculpin Spp.	25	16
Neogobius melanostomus	Round Goby	2	1
Culaea inconstans	Brook Stickleback	89	56

Appendix 3 – Complete list of benthic macroinvertebrate taxa collected and the number and proportion of the 160 sites at which each species was found for SVS-SWO sites from Yates (2008). Note that three of the original sites used by Yates (2008) were not used in the current study.

Taxon Group	# of Sites Present	% of Sites Present
Amphipoda Gammaridae	36	23
Amphipoda Hyalellidae	62	39
Bivalvia Pisidiidae	135	84
Coleoptera Curculionidae	7	4
Coleoptera Dytiscidae	31	19
Coleoptera Elmidae	146	91
Coleoptera Gyrinidae	2	1
Coleoptera Haliplidae	38	24
Coleoptera Hydrophilidae	18	11
Coleoptera Psephenidae	14	9
Coleoptera Scirtidae	1	<1
Collembola Isotomatidae	8	5
Decapoda	16	10
Diptera Athericidae	1	<1
Diptera Ceratopogonidae	126	79
Diptera Chaorboridae	1	<1
Diptera Chironomidae	158	99
Diptera Chironiminae	137	86
Diptera Diamesinae	12	8
Diptera Orthocladinae	124	78
Diptera Prodiamesinae	10	6
Diptera Tanypodinae	136	85
Diptera Tanytarsini	105	66
Diptera Dixidae	5	3
Diptera Empididae	62	39
Diptera Ephrydidae	6	4
Diptera Muscidae	1	<1
Diptera Psychododae	18	11
Diptera Ptychopteridae	2	1
Diptera Simuliidae	44	28
Diptera Stratiomyiidae	6	4
Diptera Tabanidae	44	28
Diptera Tipulidae	75	47
Emphemeroptera Ephemeridae	1	<1
Ephemeroptera Baetidae	74	46
Ephemeroptera Caenidae	53	33
Ephemeroptera Ephemerellidae	18	11
Ephemeroptera Heptageniidae	49	31
Ephemeroptera Leptohyphidae	29	18
Ephemeroptera Leptophlebiidae	33	21
Gastropoda Ancylidae	17	11
Gastropoda Hydrobyiidae	7	4
Gastropoda Lymnaeidae	33	21
Gastropoda Physidae	82	51
Gastropoda Planorbidae	42	26
Gastropoda Valvatidae	11	7
Hemiptera Belostomatidae	1	<1
Hemiptera Corixidae	63	39
Hemiptera Pleidae	1	<1

Taxon Group	# of Sites Present	% of Sites Present
Hemiptera Veliidae	5	3
Hirudinea	48	30
Isopoda Assellidae	75	47
Lepidoptera	7	4
Megaloptera Corydalidae	8	5
Megaloptera Sialidae	30	19
Odanata Aeshnidae	19	12
Odanata Calopterygidae	40	25
Odanata Coenagrionidae	33	21
Odanata Cordulegasridae	5	3
Odanata Gomphidae	2	1
Odanata Libellulidae	6	4
Oligochaeta	129	81
Plecoptera Capniidae	22	14
Plecoptera Leutridae	1	<1
Plecoptera Nemouridae	6	4
Plecoptera Perlidae	3	2
Plecoptera Perlodidae	6	4
Plecoptera Taeniopterygidae	16	10
Prostigmata Arrenuridae	5	3
Prostigmata Hydromidae	1	<1
Prostigmata Hygrobatidae	75	47
Prostigmata Hygrophantidae	5	3
Prostigmata Lebertiidae	91	57
Prostigmata Limnesiidae	1	<1
Prostigmata Mideopsidae	6	4
Prostigmata Oribatidae	6	4
Prostigmata Pionidae	7	4
Prostigmata Sperchonidae	42	26
Prostigmata Torrenticolidae	8	5
Prostigmata Unionicolidae	15	9
Trichoptera Brachycentridae	3	2
Trichoptera Dipseudopsidae	1	<1
Trichoptera Glossosmatidae	4	3
Trichoptera Helicopsychidae	12	8
Trichoptera Hydropsychidae	76	48
Trichoptera Hydroptilidae	6	4
Trichoptera Lepidostomatidae	2	1
Trichoptera Leptoceridae	47	29
Trichoptera Limnephilidae	60	38
Trichoptera Molannidae	4	3
Trichoptera Philopotamidae	16	10
Trichoptera Phyrganiidae	28	18
Trichoptera Polycentropidae	11	7
Trichoptera Psycomyiidae	10	6
Trichoptera Rhyacophilidae	6	4
Turbellaria	39	24

Appendix 4 – Animal care protocol approval

The UNIVERSITY *of* WESTERN ONTARIO – COUNCIL ON ANIMAL CARE

PROTOCOL RENEWAL APPLICATION #2007-075 Bailey

A. PROJECT/INVESTIGATOR INFORMATION

I. PROTOCOL INFORMATION

Investigator Name: Robert Bailey	**Current Protocol #: 2007-075**
Project Title: *This is a **NEW title***- **Yes** ☐ The effects of headwater enclosures on stream function and fish habitat	

II. INVESTIGATOR CONTACT INFORMATION

Department: Biology	**Email Address:** drbob@uwo.ca	
Office Phone: ext 86775	**Lab Address:** Staging 124	**Lab Phone:** ext 86775

B. INVESTIGATOR DECLARATION

I. All animals used in this research project will be cared for in accordance with the recommendations of the Canadian Council on Animal Care and the requirements of the provincial legislation entitled, "The Animals for Research Act," of the Province of Ontario.

II. I confirm that the changes associated with this Protocol Renewal, in addition to previous form submissions associated with this protocol, accurately represents the proposed animal use.

III. I will ensure that any individual who will perform any animal-related procedure(s) within this protocol will complete all related mandatory training AND will be made familiar with the contents of this document.

1. I support the above declaration - YES ☒ Today's Date *mm/dd/yy*: 08/04/09
2. By checking 'YES' in this section, I authorize the contents of this form and its electronic delivery via acvs@uwo.ca YES ☒ NO ☐
3. I authorize Katie Stammler to submit this form and to receive a copy of authorization via email on my behalf. Today's Date *(mm/dd/yy)* 08/04/09

AUS APPROVAL - *AUS Office Use Only-*

Veterinary Authorization by Click Here

☐ **Authorized On Date** *(mm/dd/yy)*

Appendix 5 - *Curriculum vitae*

KATIE L. STAMMLER

EDUCATION

Ph.D. (*Biology and Environment & Sustainability*), University of Western Ontario, London, ON
Thesis: Extent and characteristics of stream enclosures in southwestern Ontario and their effects on downstream aquatic biota
June 2011

M.Sc. (*Zoology*), University of Guelph, Guelph, ON
Thesis: Agricultural drains as fish habitat in southwestern Ontario
December 2006

B.Sc. (*Honours Biological Sciences, Environmental Biology with Co*-op), University of Windsor, Windsor, ON
Thesis: Assessment of interaction outcomes by round gobies, *Neogobius melanostomus*
May 2003

AWARDS

- 2010 Canadian Aquatic Resources Section Peter A. Larkin Student Travel Award
- 2010 UWO Graduate Thesis Research Award
- 2009 UWO Environmental Science Graduate Instream Scholarship
- 2008 American Fisheries Society Ontario Chapter Student travel award
- 2008 American Society of Ichthyologists and Herpetologists' General Endowment Fund travel award
- 2008-09 UWO Biology travel award
- 2007-10 UWO Environment and Sustainability travel award
- 2006-09 Western Graduate Research Scholarship
- 2006-08 NSERC Canada Graduate Scholarship
- 2003-05 Ontario Graduate Scholarship
- 2002 Franklin/Holder-Franklin Award for excellence in Undergraduate Research
- 2002 Co-operative Education Certificate of Excellence
- 2001-2002 Alcan Aluminum Corporation and the Great Lakes Research Consortium Undergraduate Research Fellowship
- 2001 NSERC Undergraduate Student Research Award
- 2000-2001 Andrew Douglas Bowlby Scholarship in Biology
- 1998-2002 University of Windsor Entrance Scholarship

PUBLICATIONS

Stammler, K.L., McLaughlin, R.L. and Mandrak, N.E.. 2008. Streams modified for drainage provide fish habitat in agricultural areas. Canadian Journal of Fisheries and Aquatic Sciences, 65: 509-522.

Stammler, K.L. and Corkum, L.D. 2005. Assessment of interaction outcomes by round gobies, *Neogobius melanostomus.* Environmental Biology of Fishes, 73:117-123.

ORAL PRESENTATIONS

Guest Lectures

October 2009, 2010. Presentation Skills. Faculty of Science undergraduate course, Environmental Issues. University of Western Ontario, London, ON. Course instructor: Dr. Christie Stewart.

March 2010. Techniques in Aquatic Ecology. Faculty of Science undergraduate course, Research Techniques in Environmental Science. Course instructor: Dr. Christie Stewart.

September 2007-2010. Measuring stream discharge using an Acoustic Doppler Velocity meter – field demonstration and lecture. Faculty of Science undergraduate course, Watershed Hydrology**.** Course instructor: Dr. Robert Schincariol.

Invited Presentations

Stammler, K.L., Bailey R.C., Yates, A.G. & Mandrak, N.E. Extent and characteristics of enclosed watercourses and their effects on downstream aquatic ecosystems. 42nd Drainage Engineers Conference. Guelph, ON. October 22, 2010.

Stammler, K.L., Ward-Campbell, B., McLaughlin, R.L., Bailey, R.C. & Mandrak N.E. Biological fisheries implications of agricultural drainage practices. Stressors, Habitat, and Biology in Agricultural Streams: a 1-Day Workshop. Utopia, ON. February 17, 2010

Stammler, K.L. What is happening in drains – Effects of headwater enclosures. LICO and DSAO Convention. London, ON. January 20-22, 2009.

Stammler, K.L., Ward-Campbell, B., McLaughlin, R.L., Bailey, R.C. & Mandrak, N.E. What's happening in drain research. Ontario Great Lakes Area Learning and Development Session. Barrie, ON. December 4, 2008.

Stammler, K.L., Ward-Campbell, B., McLaughlin, R.L., Bailey, R.C. & Mandrak, N.E. Addressing uncertainties related to fish habitat in drains – overview of current research. 40th Drainage Engineers Conference. Guelph, ON. October 24, 2008.

Stammler, K.L., Bailey R.C. & Mandrak, N.E.. Fish assemblage and water chemistry responses across a gradient of headwater stream enclosures. Biodiversity and agricultural sustainability in North America symposium at the 2008 Joint Meeting of Ichthyologists and Herpetologists. Montreal, QC. July 23-28 2008.

Stammler, K.L., McLaughlin, R.L., Mandrak, N.E., Bailey, R.C. & Ward-Campbell, B. Do drains provide fish habitat? Land Improvement Contractors of Ontario Conference. London, ON. Jan 25-25, 2007.

Stammler, K.L., McLaughlin, R.L, Mandrak, N.E., Bailey, R.C. Ward-Campbell, B & Wren,L. What's happening in drains? A review of current research. A.D. Latornell Conservation Symposium, Orillia, ON. Nov 15-17 2006.

Regular Contribution Oral Presentations

Stammler, K.L., Bailey, R.C. & Mandrak, N.E. Responses of downstream aquatic ecosystems across a gradient of headwater enclosure in agricultural streams. Headwater stream symposium at the American Fisheries Society 140th Annual Meeting. Pittsburgh, PA. September 12-17, 2010.

Stammler, K.L., Yates, A.G. & Bailey R.C. Where have all the headwaters gone? Calculating the extent and characteristics of headwater enclosures. American Fisheries Society, Ontario Chapter General Meeting. Orillia, ON. March 4-6, 2010.

Stammler, K.L., Bailey, R.C. & Mandrak, N.E. Downstream effects of headwater enclosures on fishes and their habitat. Headwater stream symposium at the American Fisheries Society 139th Annual Meeting. Nashville, TN. August 30-September 3, 2009.

Stammler, K.L., Bailey, R.C. & Mandrak, N.E Effects of enclosing headwaters on downstream agro-ecosystems: Hierarchical variability in biota and their environment. Impacts of agriculture on stream ecosystems symposium at the North American Benthological Society, 57th Annual Meeting. Grand Rapids, MI. May 17-22, 2009.

Stammler,K.L., Bailey, R.C. & Mandrak, N.E Examining the cumulative effects of headwater enclosures on downstream fish assemblages and water chemistry in stream agro-ecosystems. American Fisheries Society, 138th Annual Meeting. Ottawa, ON. August 17-21, 2008.

Stammler, K.L. & Bailey, R.C. Scratching the surface: Cumulative effects of headwater enclosures on fish and BMI assemblages and water chemistry in stream agro-ecosystems. North American Benthological Society, 56th Annual Meeting. Salt Lake City, UT. May 25-30, 2008.

Bailey, R., Yates, A. & **Stammler, K.L.** 2008. What's happened down on the farm? Applying the Reference Condition Approach to bioassessment of Great Lakes stream agro-ecosystems. St. Lawrence River Ecosystem Conference. Cornwall, ON. May 6-8, 2008.

Stammler, K.L. & Bailey, R.C. Uncovering the cumulative effects of headwater enclosures in stream agro-ecosystems. North American Benthological Society, 55th Annual Meeting. Columbia, SC. June 3-7, 2007.

Stammler, K.L., McLaughlin, R.L. & Mandrak, N.E. Streams modified for drainage provide fish habitat in agricultural areas. American Fisheries Society, 136th Annual Meeting. Lake Placid, NY. September 10-14 2006.

Stammler, K.L., McLaughlin, R.L. & Mandrak, N.E. Agricultural drains as fish habitat in southwestern Ontario. Canadian Conference for Fisheries Research. Calgary, AB. January 5-8, 2006.

Stammler, K.L., McLaughlin, R.L. & Mandrak, N.E. Do agricultural drains provide fish habitat? Ontario Ecology and Ethology Colloquium, Ottawa, ON. May 7-9, 2005.

Stammler, K.L. & Corkum, L.D.. 2002. A study of shelter occupancy and agonistic behaviour in male round gobies. Great Lakes Research Consortium, Syracuse, NY. March 16, 2002.

Corkum, L.D. & **Stammler, K.L.**. 2001. Nest choice by male round goby, *Neogobius melanostomus*. Canadian Conference for Fisheries Research, Toronto, ON. January 4-6, 2001.

Posters

Stammler, K.L., Yates, A.G. and Bailey, R.C. Searching for buried treasure: Calculating the extent and characteristics of headwater enclosure. North American Benthological Society, 58th Annual Meeting. Santa Fe, NM. June 6-11, 2010.

Stammler, K.L., McLaughlin, R.L. and Mandrak, N.E. Do agricultural drains provide fish habitat? Canadian Conference for Fisheries Research, Windsor, ON. January 6-9, 2005.

PEER REVIEW ACTIVITIES

Journal Reviewer

Canadian Journal of Fisheries and Aquatic Sciences, Hydrobiologia

COMMITTEE INVOLVMENT

- Environment and Sustainability Collaborative Program Planning Committee Graduate Student Representative (elected position) (2009-2010)
- UWO Environmental Research Western Earth Day Colloquium Organizing Committee (2007, 2008, 2009 (co-chair), 2010 (co-chair))
- Society of Biology Graduate Students Committee Chairperson (2008-2009)
- Society of Biology Graduate Students Biology Outreach Representative (2007-2008)

- Pugnose Shiner Recovery Team (2007-present)
- Ausable River Recovery Team (2006-present)
- American Fisheries Society Ontario Chapter Student Representative on the Executive Committee (2007)
- Ontario Ecology and Ethology Colloquium Organizing Committee (2007)

PROFESSIONAL AFFILIATIONS, WORKSHOPS AND CERTIFICATES

- American Fisheries Society – student member (2004-present)
- North American Benthological Society – student member (2007-present)
- Canadian Water Network – student member (2007-present)
- ROM Ontario Fish ID workshop
- Canadian Water Network South Saskatchewan River basin watershed workshop
- Introduction to Digital Geography and GIS (distance education course)
- Class 2 Electrofishing certificate
- Red Cross Standard First Aid includes CPR Level C
- Valid Class G Ontario Driver's License
- WHIMS certification

TECHNICAL SKILLS

- Stream fish identification and collection using backpack electrofisher and seine nets
- Benthic macroinvertebrate family level identification and collection using kicknets
- Stream habitat measurement and assessment
- Instantaneous and continuous measurement of stream discharge using an Acoustic Doppler Velocity meter and water level data loggers.
- Independent research design and implementation
- Supervision of field and laboratory staff
- Collection, synthesis, management and analysis of large and varied datasets
- Strong statistical knowledge including multivariate techniques
- Geographical Information Systems (GIS) including ArcHydro
- Computer software including MS Office and statistical packages

TEACHING EXPERIENCE

University of Western Ontario

Graduate Teaching Assistant – January 2006 – December 2010

Courses: Environmental Issues, Natural Science of Environmental Problems, Environmental Biology, Research and Hypothesis Testing, Scientific Methods in Biology (writing and ecology sections), Terrestrial and Aquatic Ecology Field Course – Algonquin Park, Analysis and Interpretation of Biological Data

University of Guelph

Graduate Teaching Assistant – January 2004-April 2005

Courses: Humans in the Natural world, Environmental Biology of Fishes, Biology of Polluted Waters, Cellular Biology (Bio 2210)

WORK EXPERIENCE

University of Western Ontario – International Polar Year, Whitehorse, Yukon
Field Assistant – August 2007

- Collected fish, benthic and water samples from various sites near Kluane National Park accessed by helicopter

Department of Fisheries and Oceans
Research contract, September – September-December 2003

- Collected and identified fishes from the Sydenham River
- Conducted site surveys for future sampling in Essex county
- Prepared a literature review

Wings Avian Rehabilitation Centre, Amherstburg, ON
Wildlife Attendant, April-June 2003

- Raised orphaned passerines, raptors, water fowl and small mammals
- Rehabilitated and transported sick and injured wild birds and small mammals
- Responsible for daily care and feeding of wild birds and small mammals

Lakeside Produce, Leamington, ON
Assistant Grower (Co-op), Summer 2002

- Scout for the Integrated Pest Management Team
- Responsible for maintenance of bee hives
- Monitored plant growth and production

Lower Thames Valley Conservation Authority, Chatham, ON
Drainage Technician (Co-op), Winter 2002

- Analyzed collected data to determine drain classification
- Created maps to display drainage classification
- Completed LTVCA & DFO drain classification project

VOLUNTEER EXPERIENCE

- American Fisheries Society Ontario Chapter conference volunteer (2008)
- Kortright Presbyterian Church (2005) – Children's program volunteer
- All Saints Anglican Church and Paulin Memorial Presbyterian Church (2004, 2005) – Speaker and participant in a forum focusing on environmental issues
- Wings Avian Rehabilitation Center (2000, 2003) – wildlife attendant
- David Maxwell Public School (2003, 2004) – Junior Kindergarten volunteer
- International Freedom Festival Children's Day (2001, 2003)