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Determining the Association between the Structure of Stream Benthic Macroinvertebrate Communities and Agricultural Best Management Practices

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A thesis submitted in partial fulfillment of the requirements for the degree in Master of Science

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by

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Abstract

Farmers have been encouraged to adopt more sustainable farming practices (BMPs) that mitigate adverse agricultural effects on the natural environment. However, the ability of BMPs to protect or restore riverine systems continues to be questioned due to limited evidence directly linking BMP use with improved ecological conditions. The exclusion of hydrological pathways in previous field studies may explain why a direct link has not yet been established. The goal of this study was to assess the association between benthic macroinvertebrate community structure and the number and location of agricultural BMPs. Macroinvertebrates and water chemistry were sampled in 30 headwater catchments in the Grand River Watershed. Catchments exhibited gradients of BMP use and location as measured by the degree of hydrologic connectedness. Stepwise ordination regressions and variance partitioning were used to determine which environmental variables (i.e., BMP metrics, water chemistry parameters, habitat characteristics, and land use variables) were associated with benthic macroinvertebrate community structure. Water chemistry parameters were negatively associated with BMP metrics suggesting BMPs were mitigating losses of nutrients and sediments. However, BMP abundance and location explained minimal variation in benthic macroinvertebrate structure within the 30 sampled catchments. The absence of a strong association between BMPs and benthic macroinvertebrates may indicate a need for greater numbers and targeted siting of BMPS to improve water quality beyond a threshold point that would allow recolonization of intolerant invertebrate taxa. Focusing of conservation goals on ecological conditions and the promotion of BMPs that enhance in-stream habitat may also be required.

Keywords: best management practices, benthic macroinvertebrates, agriculture, river systems, water chemistry, headwater catchments, Grand River Watershed, hydrological connection, flow distance, flow accumulation.

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Table of Contents

Abstract ii

Acknowledgements iv

List of Tables and Figures vii

List of Abbreviations viii

1.0 Introduction 1

1.1 Agriculture and the Environment 1

1.2 BMPs as a Mitigation Tool for Agricultural Impacts on River Systems 3

1.3 Effectiveness of BMPs at Mitigating Negative Impacts from Agriculture 4

2.0 Research Goal 11

2.1 Objectives 11

2.2 Hypothesis 11

3.0 Methods 12

3.1 Study Area 12

3.2 Site Selection 14

3.3 Identifying the Degree of Hydrological Connection for BMPS 17

3.4 Description of Riparian Vegetation and Tile Drainage 19

3.5 BMP Metrics 20

3.6 Field Sampling 23

3.6.1 Benthic Macroinvertebrates 23

3.6.2 Habitat Assessment 24

3.6.3 Water Chemistry 25

3.7 Data Analysis 26

3.7.1 Association between BMP Metrics and Water Parameters 28

3.7.2 Stepwise Ordination Regressions and Variance Partitioning 28

4.0 Results 31

4.1 Land Use Summary 31

4.2 BMP Abundance and Composition 31

4.3 Degree of Hydrological Connection for BMPs 33

4.4 Habitat Assessment 34

4.5 Water Chemistry 35
4.6 Benthic Invertebrate Composition 38
4.6.1 Potential drivers of the benthic macroinvertebrate community structure 39
4.7 Stepwise Ordination Regressions and Variance Partitioning of Significant Environmental Variables 40
5.0 Discussion 43
5.1 Structural BMP Composition within Headwater Catchments 43
5.2 Influence of BMP metrics on water parameters 47
5.3 Association between benthic macroinvertebrate communities and BMPs 50
6.0 Management Implications and Recommendations 58
7.0 Future Research 62
8.0 Conclusions 64
9.0 References 65
Appendix A 83
Curriculum Vitae 85

List of Tables and Figures

Figure 1.1: Conceptual diagram illustrating the potential impacts that agriculture can have on river systems (A) and how BMPs can potentially mitigate those impacts (B). 5

Figure 3.1: Map of the Grand River Watershed (GRW) within Southwestern Ontario (A) with the locations of the Nith and Conestoga subwatersheds within the GRW (B) and the locations of the 30 sampled catchments with the Nith and Conestoga subwatersheds (C). 13

Figure 3.2: Decision-making criteria and process for site selection. 17

Figure 3.3: Conceptual diagrams for flow accumulation (A), flow distance (B) and how the two measurements for flow distance were measured (C).. 18

Figure 3.4: Conceptual diagram illustrating the output of variance partitioning 30

Figure 4.1: Ordination of benthic community composition based on principal component analysis indicating the arrangement of the benthic community 40

Figure 4.2: Representation of the total variance explained in both the BMP Type model (A) and the BMP Summary model (B) 42

Figure 6.1: Diagram of two-stage ditch with meandering deep main channel (A) compared to straight main channel (B) from a conventional drainage ditch and the adjacent floodplain channel (C) in a two-stage ditch compared to homogenous channel depth (D) from a conventional drainage ditch 61

Table 3.1: Summary of the BMP metrics developed and which BMPs they were applied to 22

Table 4.1: Descriptive statistics for landscape descriptors for 30 sampled headwater catchments in the Nith and Conestoga subwatersheds 31

Table 4.2: Statistical summary of the BMP abundance for the 30 sampled catchments 32

Table 4.3: Summary statistics describing the degree of hydrological connection as measured by flow accumulation and flow distance metrics 34

Table 4.4: Descriptive statistics for habitat parameters assessed using the Environmental Protection Agency's Rapid Bioassessment Protocol 35

Table 4.5: Summary of water chemistry results collected from all 30 sampled catchments using the grab sample technique 36

Table 4.6: Results of the regression analysis between the water parameters and BMP type metrics 37

Table 4.7: Results of the regression analysis between the water parameters and BMP summary metrics 38

List of Abbreviations

BMP – Best Management Practice

GRW – Grand River Watershed

RWQP – Rural Water Quality Program

GRCA – Grand River Conservation Authority

DEM – digital elevation model

GIS – geographic information system

DWM – distance weighted model

RDA – redundancy analysis

DCA – detrended correspondence analysis

VIF – variance inflation factor

CV – coefficient of variation

PCA – principle components analysis

PCoA – principle coordinate analysis

MS – manure storage

LAR – livestock access restriction

EC – erosion control

DHC – degree of hydrological connection

TSS – total suspended sediment

1.0 Introduction

Agricultural production is an economic driver in many regions around the world. Over the past 50 years, agricultural production has continued to grow by 2-4% annually (FAO Statistical Yearbook, 2013; McRae et al., 2000). Agricultural land uses occupy approximately 12% of the world's land surface (FAO Statistical Yearbook, 2013). Growth in livestock production has been relatively stagnant in recent years; however, it accounts for the largest proportion of agricultural land, and requires massive amounts of energy and resources (FAO Statistical Yearbook, 2013). With a rapidly growing population worldwide, the increase in agricultural production needs to continue if the food needs of people and livestock are to be met (AAFC, 2012a; FAO Statistical Yearbook, 2013). To meet these current and growing needs of agricultural expansion, added production from less farmland has largely been achieved through technological advances in machinery and crops (McRae et al., 2000).

1.1 Agriculture and the Environment

Natural resources are continually being impacted by land use change (i.e., deforestation), water abstraction, and soil erosion, which result from common agricultural practices (FAO Statistical Yearbook, 2013). Farming practices, such as fertilizer application, cultivation and manure application, often generate non-point sources of pollution and have the potential to increase sediment and nutrient inputs to river systems (Carpenter et al., 1998; Lenat, 1981; Voora et al., 2012). Non-point sources of pollution often originate from extensive areas of land and can vary depending on the season, weather, and type of agricultural activity (Carpenter et al., 1998). Therefore, non-point sources of pollution are troublesome because they are difficult to measure and mitigate (Walker & Graczyk, 1993). As a result, freshwater systems are put under an immense amount of stress worldwide from agricultural land use (Allan, 2004; Wood & Armitage, 1997).

Declines in water quality and ecological integrity throughout many river systems have been well documented and in many cases linked to agricultural practices (*see review by* Allan, 2004). Irrigation of crops for both humans and livestock consumption has allowed crop production to increase, but it also accounts for 70% of all freshwater abstracted for human use (FAO Statistical Yearbook, 2013). Crop production often involves the direct application of fertilizers to agricultural fields, along with extensive tillage, which can significantly increase the amount and rate that sediments and nutrients enter river systems through runoff (Carpenter et al., 1998; Walser & Bart, 1999) (Figure 1A). Livestock can also increase nutrient and sediment loads in rivers through destabilization of banks, removal of vegetation through grazing, and direct input of fecal matter (*see review by* Belsky et al., 1999; Collins et al., 2007). These excessive inputs of sediment and nutrients are known to alter the river environment and degrade habitat for fish, invertebrates and plant species (Barton & Farmer, 1997; Miltner & Rankin, 1998; Qie et al., 2007; Riseng et al., 2011). Impacts from agricultural practices on physio-chemical and biotic condition have been clearly demonstrated by numerous studies comparing agriculturally dominated catchments to predominantly forested catchments (Kroll et al., 2009; Riseng et al., 2011). Low order streams in agricultural catchments are particularly sensitive to agricultural impacts because they are strongly connected to surrounding landscapes and the associated activities (Miltner & Rankin, 1998). The increased risk of pollution from common farming practices on low order streams can lead to eutrophication and unfavourable conditions for aquatic life through an increase in light and temperature levels, decrease in available substrate, and increase in nutrient concentrations and turbidity (Riseng et al., 2011). Lastly, excess nutrient inputs also create potential health hazards for both humans and livestock that depend on healthy river systems for drinking water (Voora et al., 2012; Willms et al., 2002). Therefore, the effective mitigation of detrimental impacts from agriculture on low order streams is critical to the protection of riverine systems as well as human health and economic productivity.

1.2 BMPs as a Mitigation Tool for Agricultural Impacts on River Systems

The productivity of farming operations is dependent upon natural systems (i.e., rivers and streams) to provide crops and livestock with their basic necessities to grow and reproduce. Past events, such as the Dust Bowl, have shown the importance of understanding the inherent limits of the natural environment and developing farming practices that can be productive, yet sustainable, within the environmental thresholds of natural systems. Farmers have been encouraged to adopt farming practices aimed at protecting the natural environment from farming activities that may degrade the soil, water, air or habitat for wildlife (AAFC, 2006). These farming operations are frequently called Best Management Practices (BMPs) and are regularly funded through government programs that offer financial and technical assistance to farmers for installing and operating them (Napier & Bridges, 2002). BMPs are alternative or modified farming practices that are intended to be cost-effective, not hinder productivity, and mitigate or prevent environmental impacts associated with many common farming practices (AAFRD, 2009). BMPs were originally developed to mitigate soil loss through erosion that was detrimental to agricultural production (Logan, 1993). The environmental benefits, such as water quality protection, were not the primary concern when developing and implementing agricultural BMPs. However, recent acknowledgement of the negative impacts that farming practices can have on surrounding natural resources (e.g., water resources) has begun to be an integral reason why farmers are being encouraged to implements BMPs (Logan, 1993). BMPs can be classified into two types: managerial and structural. Managerial BMPs are typically associated with the source reduction of pollutants and occur at a farm-wide scale (e.g., contour farming, conservation tillage and nutrient management; Rao et al., 2009). The goal of these BMPs is to minimize the application or release of unnecessary pollutants that may harm the environment through tillage, manure and fertilizer application, and other farming practices. Thus, managerial BMPs are focused on environmental awareness and changing the behaviour of farmers to become more responsible with what they apply to the land and how they manage their farms. In contrast, structural BMPs are commonly associated with the interruption of pollutant transport off agricultural lands into waterways. Transport interruption is typically achieved using structures that either naturally filter pollutants (e.g., riparian vegetation) or create a barrier to prevent pollutants from entering waterways (e.g., livestock fencing, manure storage structures) (Figure 1B). Structural BMPs can therefore be placed strategically on the landscape in areas where they will most effectively intercept pollutants before they reach rivers or streams. The specific types and implementation mechanisms for managerial and structural BMPs vary among different regions. However, the BMPs themselves tend to be relatively consistent in their function and purpose. They rely on simple pollution reduction or interruption activities taking place at the individual farm scale, with the purpose of mitigating agricultural impacts on the environment.

1.3 Effectiveness of BMPs at Mitigating Negative Impacts from Agriculture

Past studies have shown that BMPs are successful to some extent at mitigating agricultural impacts by reducing sediment, nutrients, and other pollutants found in river systems that are linked to agricultural practices (Gabel et al., 2012; Marshall et al., 2008; Mayer et al., 2007; Yates et al., 2006; Barton & Farmer, 1997; Herendeen & Glazier, 2009; Park et al. 1994; Walker & Graczyk, 1993). For example, Park et al. (1994) demonstrated that BMP

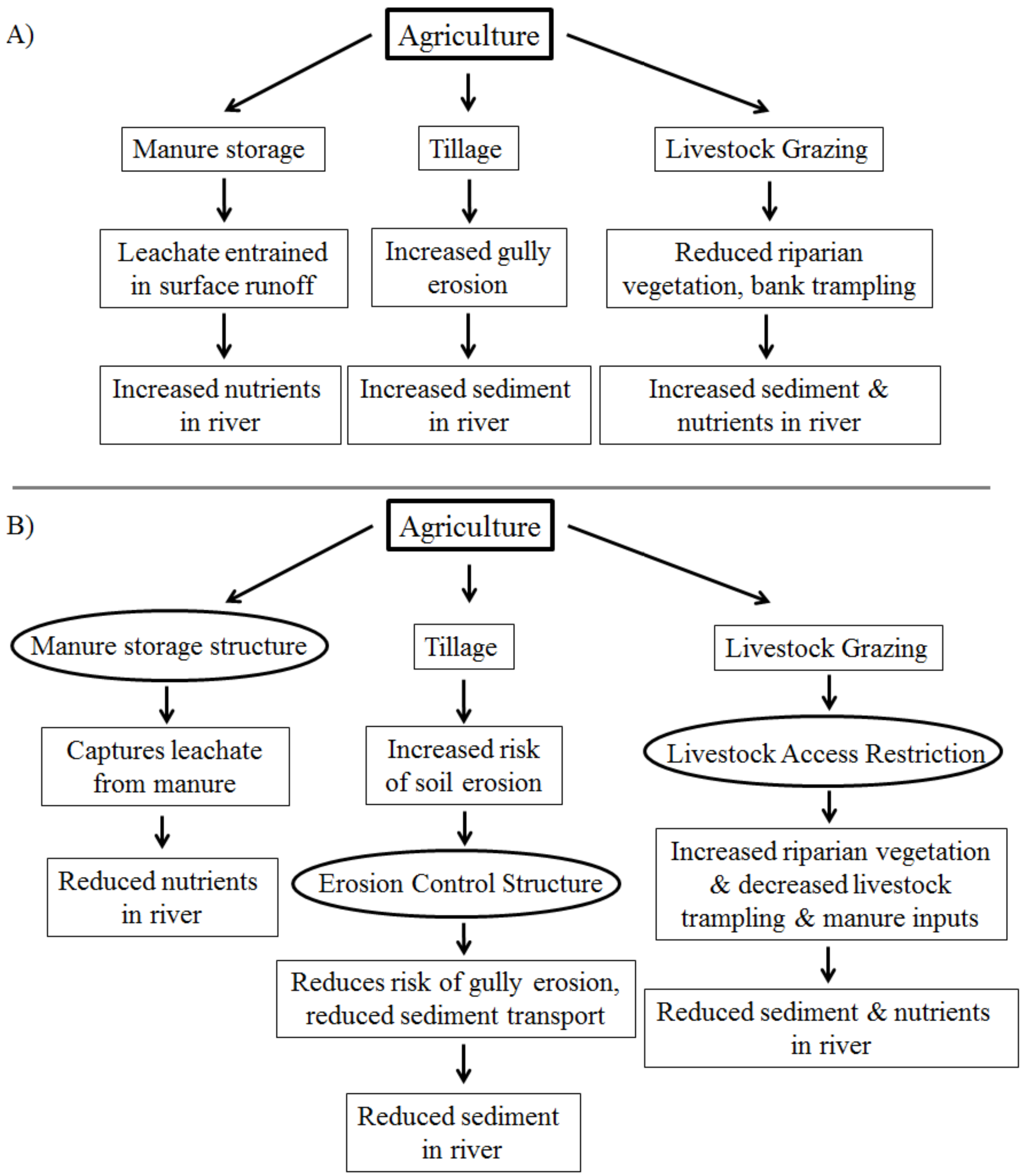

Figure 1.1: Conceptual diagram illustrating the potential impacts that agriculture can have on river systems (A) and how BMPs can potentially mitigate those impacts (B) (circled text indicate BMP).

implementation can significantly reduce sediment and nutrient concentrations from non-point sources of surface runoff, which is the main transport mechanism for pollutants to river systems from non-point sources. Gabel et al. (2012) also found that streams without BMPs along them had increased specific conductance and pH, as well as larger concentrations of TDP, NH_4^+-N, NO_3^--N and reduced diversity of diatoms compared to streams with BMPs. Past studies have also shown that common structural BMPs, such as vegetation buffers and livestock access restrictions to streams, can reduce the concentration of DP, TP, and NO_3^--N contaminants found in river systems (Easton et al., 2008; Panagopoulos et al., 2011) and allow reestablishment of intolerant, native, coldwater fish species (Marshall et al., 2008). However, the ability of BMPs to improve aquatic ecosystems continues to be questioned due to weak and ambiguous associations between BMP use and improved ecological conditions (Allan, 2004). Indeed, the studies that have shown BMPs to be effective (e.g., Barton & Farmer, 1997; Hamlett & Epp, 1994; Herendeen & Glazier, 2009; Makarewicz et al., 2009; Park et al. 1994; Walker & Graczyk, 1993) are countered by numerous studies that have shown weak or no correlation between BMP implementation and improved ecological or water quality conditions in rivers (e.g., Nerbonne & Vondracek, 2001; Schellinger & Clausen, 1992; Yates & Bailey, 2007). For example, in the aforementioned study by Gabel et al. (2012), water quality and diatoms improved in BMP streams, however, metrics of benthic macroinvertebrate community condition did not differ significantly between non-BMP and BMP streams. Nerbonne and Vondracek (2001) also found that both invertebrates and fish species did not significantly change when BMPs were implemented. Similar results have been seen when assessing water quality by Tuppad et al. (2010), Hamlet and Epp (1994), and Bosch et al. (2013), who all found that water quality parameters did not significantly improve when BMPs were implemented in agricultural catchments. Lastly, common structural BMPs such as vegetative buffer strips have been found to be ineffective at significantly reducing solids, N, P, and bacteria concentrations from surface runoff (Schellinger & Clausen, 1992). These studies demonstrate that BMP effectiveness is highly variable, and that simply adding BMPs haphazardly to the landscape may not improve river systems in a predictable manner.

Assessing and quantifying the true effectiveness of BMPs at improving the chemical, biological, and physical habitat of river systems is a difficult task due to complex interactions that occur at the watershed scale, such as variation in substrate, water chemistry, and habitat throughout riverine systems (Tuppad et al., 2010). Changes in these environmental variables, such as land use and soil type, are known to influence biotic communities (Yates & Bailey, 2007), which then makes assessing the BMPs influence difficult. Ensuring that potentially confounding landscape variables are controlled for is thus critical prior to undertaking BMP assessments. It is generally considered that the overall physical, chemical and ecological characteristics of rivers and streams are a function of the catchment (Hynes, 1975; Vannote et al., 1980). However, throughout a watershed system, streams and rivers combine to create an interconnected system that collects and deposits nutrients, pollutants, biota, and sediment at different rates and locations (Hynes, 1975). Numerous studies have shown how different locations on the landscape disproportionately influence river systems because they contribute substantially more surface runoff to rivers (Galzki et al., 2011; Gburek & Sharpley, 1998; Marjerison et al., 2011; Panagopoulos et al., 2011; Piechnik et al., 2012; Pionke et al., 2000). As water travels overland sediment and nutrients can become mobilized and eventually deposited into waterways, which can lead to detrimental impacts on aquatic environments (Allan, 2004). Furthermore, as slope increases and distance to waterways decreases, the risk of pollution from runoff generally increases (Agnew et al., 2006; Kirkby et al., 2002). White et al. (2009) used landscape features (i.e., soil, topography, landcover) to identify areas that contributed significant amounts of pollutants to waterways, which they termed as critical source areas (CSAs). Their study found that 5% of the watershed contributed 50% and 34% of sediment and phosphorus loads, respectively (White et al., 2009). Pionke et al. (2000) predicted a larger influence from CSAs in an agricultural hill-land watershed within the Chesapeake Basin, with only 6% of the watershed contributing 98% of sediment loss from the landscape. Therefore, the effectiveness of a BMP may be dependent on where on the landscape it is located and if that area is hydrologically connected with the receiving river system. In general, past studies (e.g., Cook et al., 1996; Gabel et al., 2012; Yates et al., 2007) have assumed that an increase in the number of BMPs implemented within a watershed will have a consistent incremental benefit on the river ecosystem and have not addressed the role of BMP location. Many studies have used models to demonstrate how the spatial location of BMPs within a catchment is critical to determining their effectiveness (e.g., Bosch et al., 2013; Easton et al., 2008; Tomer et al., 2003). For example, Bosch et al. (2013) used the Soil and Water Assessment Tool (SWAT) to find that BMPs located near the mouth of the river and in areas that intercepted a large amount of water from upland areas were more effective than a random distribution of BMPs across the landscape. Easton et al. (2008) used the Variable Source Loading Function (VSLF) model to determine that BMPs located in areas that generated significant storm runoff from overland flow substantially reduced P loading in streams. Field studies measuring the importance of spatial location to BMP effectiveness are less common. However, Tomer et al. (2003) found that vegetation strips located downslope from a large contributing area provided the greatest potential to reduce sediment from entering the stream when compared to vegetation strips receiving runoff from a small upslope area. Therefore, determining the degree of hydrological connection (DHC) of BMPs to the river system is likely important to understanding the potential of BMPs to mitigate the agricultural impacts on waterways

BMPs are intended to reduce non-point source pollution that results from many farming practices, but it is often difficult to predict, track and quantify non-point source pollution by strictly looking at the source of the pollutant (Walter et al., 2000). In contrast, surface runoff follows basic hydrological principles of flow direction and flow accumulation and can be predicted and quantified with a high degree of accuracy using modern GIS techniques (Jain & Singh, 2005). Therefore, the DHC for a specific point, or BMP, on the landscape can be calculated. Two main components determine the DHC between a specific area on the landscape, which includes the activities that take place on it (i.e., farming, BMPs), and a receiving river system. First, the distance that water must travel to the waterway from a given location on the landscape. Second, the amount of surface runoff that is collected from uplands, which will be influenced by the receiving location, as it travels onwards to the river. Studies have shown that as farming activities occur closer to river systems, there is an increased risk of pollution from runoff, however, stronger relationships are seen when additional landscape features are considered, such as topography (Agnew et al., 2006; Marjerison et al., 2011). For example, Agnew et al. (2006) identified two locations with similar distances to a stream, but each location had a different risk of generating runoff due to topographical differences. This shows the importance of topography in generating runoff and influencing hydrological processes. Therefore, the distance measurement must account for changes in topography that would influence the true length that water would have to travel to reach the river. The second component needed to identify the DHC is determined by assessing the accumulation of overland flow to particular points on the landscape (i.e., flow accumulation). Locations collecting runoff from an upland region influence the water that passes through it depending on the activity taking place at the receiving location, such as filtering a particular nutrient or sediment. Furthermore, the larger the upland area that the receiving location drains the greater the influence that location may have on the eventual receiving river system. For example, a buffer strip that intercepts water from a large upland area as opposed to a small upland area has a greater potential to filter more pollutants from the landscape that may have become entrained in the runoff. When the flow distance and flow accumulation of each BMP are determined, the influence of spatial location on the effectiveness of BMPs can then be assessed. Such an assessment will determine if the strategic placement of BMPs will result in a greater reduction in sediment and nutrient concentrations in river systems and subsequent improvements in ecological conditions. Structural BMPs are of particular interest because they can be strategically placed on the landscape to intercept agricultural pollutants from areas with a high DHC. This would give programs that administer and fund BMPs a strategic edge at improving the effectiveness of BMPs being implemented in a watershed (Tomer et al., 2003). However, even with the research noted previously, there is still a lack of conservation programs taking a targeted approach in their conservation initiatives when they promote and implement BMPs (White et al., 2009).

2.0 Research Goal

The goal of my research project was to describe and assess the associations between the structure of stream benthic macroinvertebrate communities and the number and location of structural agricultural BMPs relative to their degree of hydrological connection (DHC) within headwater catchments with the purpose of informing BMP implementation programs aimed at mitigating agricultural impacts on river systems.

2.1 Objectives

1) Describe the number and location of agricultural BMPs that were implemented by the Rural Water Quality Program (RWQP) within the Nith and Conestoga subwatersheds.
2) Describe the structure of stream benthic macroinvertebrates communities in headwater streams of the Nith and Conestoga subwatersheds of the Grand River Basin.
3) Assess the association between the structure of stream benthic macroinvertebrate communities and the number and location of agricultural BMPs relative to their DHC.
4) Inform the RWQP about potential management strategies when implementing agricultural BMPs based off of findings and make recommendations for agencies across North America implementing similar agricultural BMP programs.

2.2 Hypothesis

It is hypothesized that study sites with numerous BMPs that are in areas with a high DHC will show enhanced ecosystem conditions, which will be represented by a benthic macroinvertebrate community structure with greater richness and abundance of pollutant-intolerant species.

3.0 Methods

3.1 Study Area

My study area was within the Grand River Watershed (GRW), which is located in Southwestern Ontario and covers approximately 7,000 km² (Yates & Bailey, 2010a; Figure 3.1A). The Grand River flows nearly 300 km in a southerly direction from its headwaters near Dundalk, Ontario to its outflow on the north shore of Lake Erie near Dunnville, Ontario (GRCA, 2014). The river collects water from four major tributaries; the Nith, Conestoga, Speed and Eramosa Rivers. The climate in the GRW is temperate, with the central region (i.e., Regional Municipality of Waterloo) having a daily mean temperature of 7ºC, average humidity of 87.8%, and yearly precipitation of 916.8 mm (Environment Canada, 2014). Similar to most watersheds in Southwestern Ontario, the GRW consists of lands that are heavily populated and intensively farmed. 925,000 people currently live in the GRW, but approximately 730,000 are concentrated in the watershed's large urban centres of Kitchener, Waterloo, Guelph, Cambridge and Brantford (GRCA, 2013). The dominant land use, by area, is agriculture, comprising over 75% of the watersheds landscape. 90% of the watershed's original forest cover has been cleared over the past 150 years (Yates & Bailey, 2010b; Holysh, et al. 2000). Agricultural practices in the GRW are a mixture of cash crop, such as corn and soybeans, and livestock operations, such as dairy, beef, hog, and poultry (Yates & Bailey, 2010b).

Sampling sites for my study were located on small headwater streams within the Nith and Conestoga subwatersheds (Figure 3.1B, C). These subwatersheds were selected because they share similar physiographic features, yet have different amounts of BMP implementation projects. Both subwatersheds are dominated by agricultural land use (83% of land area) and are characterized

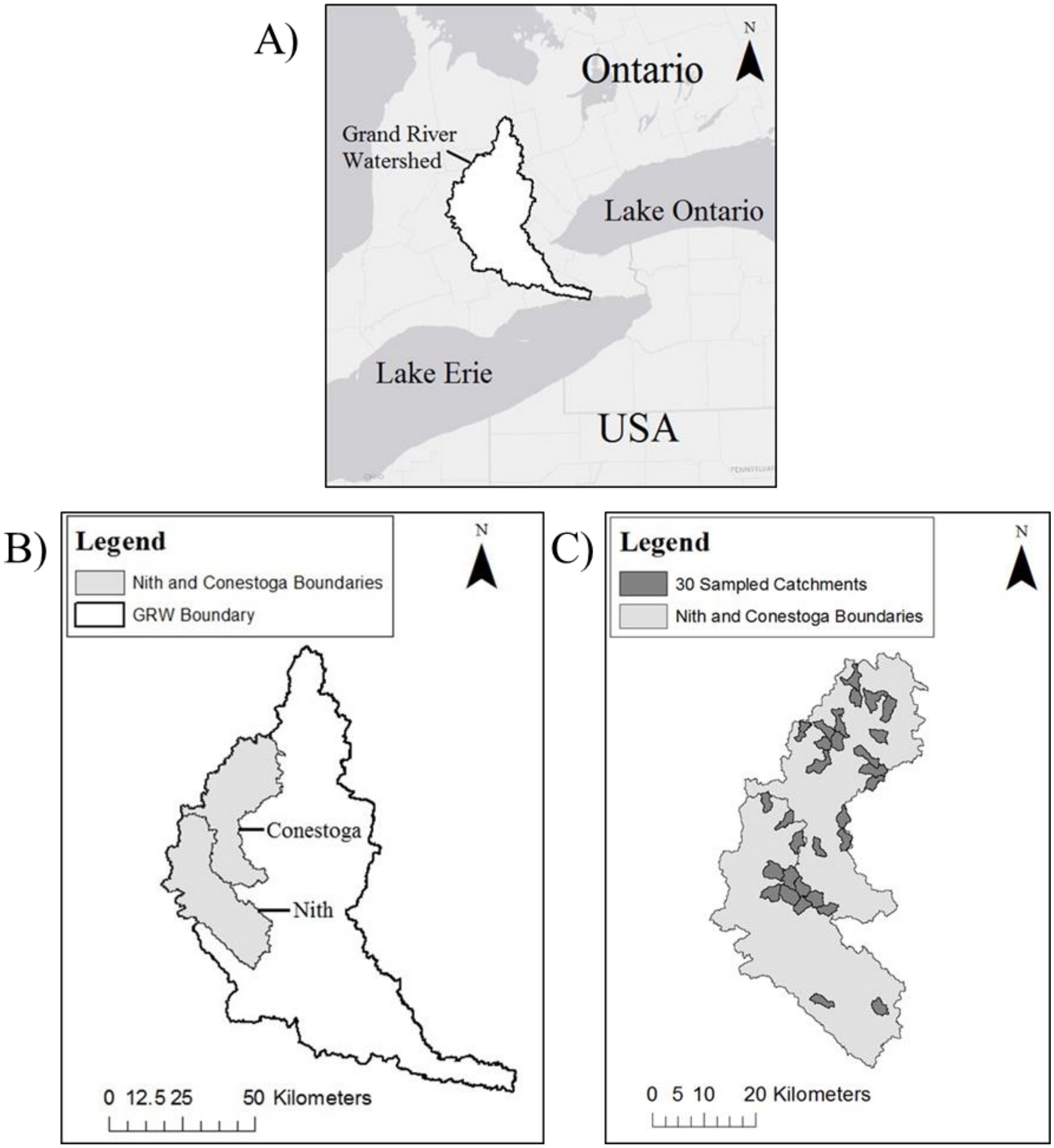

Figure 3.1: Map of the Grand River Watershed (GRW) within Southwestern Ontario (A) with the locations of the Nith and Conestoga subwatersheds within the GRW (B) and the locations of the 30 sampled catchments with the Nith and Conestoga subwatersheds (C).

by till soils (76% of soil) and rolling to fairly flat topography (Holysh, et al. 2000). BMPs in both subwatersheds are primarily implemented though the Rural Water Quality Program (RWQP), which is the main program promoting and funding BMPs in the GRW. The RWQP is administered by the Grand River Conservation Authority (GRCA) and has been developed and administered in collaboration with farming organizations in the area for over ten years (GRCA, 2013). The program's goal is to balance production and environmental needs of farming operations as well as address specific water quality concerns of individual farmers (GRCA, 2013).

3.2 Site Selection

Sites were selected to encompass existing regional variation in BMP use and location while simultaneously maximizing the comparability of physiographic and land cover characteristics among the catchments of sampling sites. Candidate sites were examined and selected using a 9 step process (Figure 3.2). First, all headwater stream segments were identified within the Nith and Conestoga subwatersheds. Catchments were then delineated for all headwater segments using ArcMap 10.0 and ArcHydro 2.0 (ESRI, 2010a, ESRI, 2010b). This process resulted in a total of 3392 catchments being delineated within the Nith and Conestoga subwatersheds. Second, delineated catchments less than 5 km^2 or greater than 12 km^2 were removed from the candidate catchment pool. The catchment area criterion was kept as small as possible for two reasons. First, environmental variables on the landscape (e.g., % agriculture, soil types) are more easily controlled in smaller catchments (Yates & Bailey, 2007). Second, headwater streams have been shown to be the most sensitive to the effects of land use practices (Greenwood et al., 2012; Miltner and Rankin, 1998) and can strongly influence downstream communities and habitat (Dodds and Oaks, 2008). The lower limit catchment area was established based on field observations that flow in catchments below 5 km^2 was commonly intermittent in the Nith and Conestoga subwatersheds and therefore would not have comparable ecological sampling conditions. The 5 to 12 km^2 catchment size criterion limited the candidate pool to 153 catchments. The third step selected catchments on the basis of soil type to ensure that the dominant surface geology texture was comparable among catchments because different soil characteristics alter soil drainage and erosion susceptibility (Bryan, 2000). The criterion was that all selected catchments needed to be comprised of over 65% till, which is the dominant surface geology type in the Nith and Conestoga subwatersheds. % till was determined using the Southern Ontario Surface geology layer generated by the Ontario Ministry of Natural Resources (OMNR, 2010). This geology layer was intersected with the catchment boundaries using ArcGIS 10.0 (ESRI, 2010a) and the percentage of till in each catchment calculated. This criterion limited the number of candidates to 148 catchments. The next three criterion involved different land use types, which were identified using ArcGIS and a 2011 land use layer generated by Agriculture & Agri-Food Canada (AAFC, 2012b). These criteria ensured that sampled sites would have comparable types and proportions of land use occurring at the catchment scale as variation in land use could potentially mask the effects of BMPs. As such the first land use criterion (fourth criterion overall) ensured that agriculture was the dominant land use type in each catchment. For the purpose of this study, agricultural land use was defined as lands used for pasture and cash crops (i.e., corn, soybean, cereals), which are the dominant agricultural crop types in the region. To be selected, agricultural land use had to comprise over 75% of the land in the catchment. The fifth and sixth criterion accounted for the other two major regional land use types, natural (i.e., shrubland, wetland, grassland and forests) and urban, which were limited to less than 25% and 5% of the catchment area, respectively. The three land use criteria resulted in 139 potential sites remaining in the candidate pool. These 139 potential sites were assessed on the basis of accessibility. By limiting the distance of each site from the nearest road to 200 meters, access issues with landowners were minimized and sampling logistics were improved. Accessibility limited the potential sites to 69 catchments. Next, any remaining catchments nested within other catchments were removed to generate independent sampling units. When two catchments were nested, the largest of the nested catchments was retained in the selection process, and the smaller catchment(s) were removed. Eliminating the nested catchments resulted in 58 potential sites. The final step in site selection involved evaluating structural BMP abundance and location with each site's catchment. BMP types and locations were mapped in ArcGIS, based on locations derived from the RWQP dataset obtained from the GRCA. BMP locations were ground-truthed in ArcGIS using high resolution aerial photos taken in 2010 (OMNR, 2012). Additional analysis using high resolution aerial photographs was conducted to identify similar BMPs that had been implemented by farmers without the assistance of the RWQP. I limited the assessed BMPs to structural BMPs (i.e, manure storage structures, livestock access restriction, erosion control structures) because these BMPs are designed to reduce pollutant loading to river systems and can be identified and quantified using aerial photos with greater accuracy than managerial BMPs. Based on the BMP information, catchments were selected to represent the entire range of BMP use present in the region. Additionally, catchments were selected based on where BMPs were located so that a gradient of hydrologic connectivity (i.e., BMP distances from stream channels and sampling points) could be assessed. The site selection did not account for topography in the degree of hydrologic connectivity because BMPs were generally scattered through the catchments and the Nith and Conestoga subwatersheds are known to have similar topography, consisting of gently rolling land. Based on this process, a total of 30 catchments representing gradients of BMP use and location were selected for sampling.

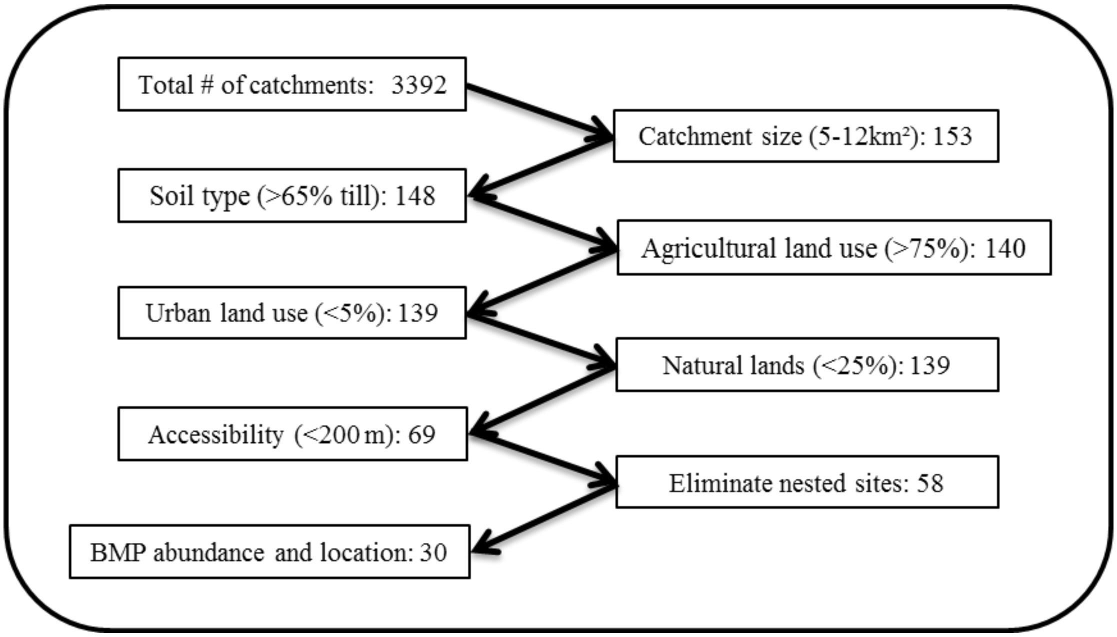

Figure 3.2: Decision-making criteria and process for site selection. Number within each box indicates the number of remaining potential sites after that step.

3.3 Identifying the Degree of Hydrological Connection for BMPS

The degree of hydrological connectivity (DHC) for each BMP was determined by quantifying flow distance and flow accumulation. The DHC of each BMP was determined using ArcGIS 10.0 and Arc Hydro 2.0 (ESRI, 2010a, ESRI 2010b). A digital elevation model (DEM) with a resolution of 26.5 m was used to determine flow accumulation and flow distance for each BMP within the 30 catchments. Flow accumulation was calculated by determining the number of cells that were upslope of each BMP in the DEM and would therefore be expected to contribute overland flow to the BMP (Figure 3A). For my study, is was hypothesized that BMPs located in areas that drain a large upland area have a greater potential to intercept and filter pollutants

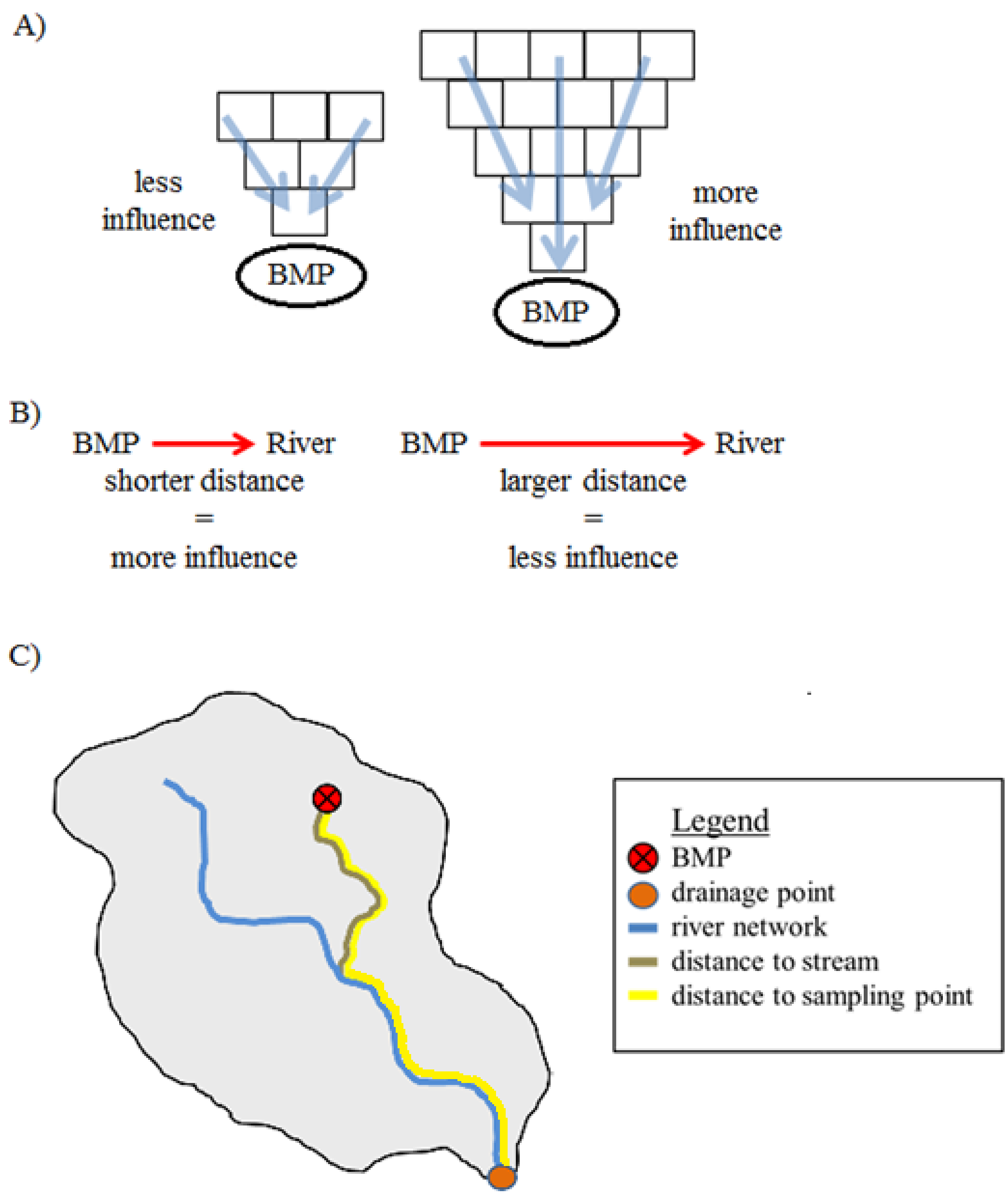

Figure 3.3: Conceptual diagrams showing how different BMP locations may influence their performance based on differences in flow accumulation (A) and flow distance (B). Flow distance involved two measurements: 1) the distance from the BMP to the river edge; and 2) the distance from the BMP to the drainage point (i.e., sampling location) (C).

before they reach the river system, and therefore have a greater potential to positively influence the ecological conditions in the river system. Flow distance involved two measurements: 1) the overland distance from the BMP to the river edge; and 2) the overland and in-stream distances from the BMP to the sampling location (Figure 3C). The flow distance calculations used the DEM to determine the direction of overland flow. The flow distance to the sampling point incorporated the flow distance within the river itself. For my study, it was hypothesized that as distance from a BMP to the river or sampling point increased, the influence that the BMP had on the river system would diminish (Figure 3B). The diminishing effectiveness of a BMP may occur because sources of pollutants between the BMP and the stream or sampling point could mask the effects that the BMP had on the river system.

3.4 Description of Riparian Vegetation and Tile Drainage

Vegetated buffers along rivers and streams are known to improve water quality and habitat for aquatic biota (Kiffney et al., 2003; Muenz et al., 2006). Therefore, riparian vegetation was incorporated into the assessment of BMPs because it may influence macroinvertebrate assemblages found at each study site. Additionally, riparian vegetation can be viewed as a passive BMP that farmers can choose to implement by simply leaving stream edges intact instead of clearing them for crop or pasture use. To determine the location and extent of riparian vegetation along river systems, high resolution aerial photos taken in 2010 (OMNR, 2013) were assessed in ArcGIS (ESRI, 2010a). A 30 m buffer was used to determine the riparian zone, and then the riparian vegetation was mapped onto a new polygon layer so that the extent of riparian vegetation in each catchment could be calculated. The 30 m riparian zone was chosen for two reasons. First, a riparian buffer of 30 m has shown to significantly improve biotic and abiotic factors of a river system in previous studies (Kiffney et al., 2003; Wilkerson et al., 2006).

Second, a wider riparian buffer is an unrealistic goal in these heavily farmed catchments where farm land is highly valuable, whether it be used for crops or pasture land.

Tile drainage was also accounted for when assessing the effectiveness of BMPs. Benefits of BMPs located on the landscape to intercept and filter pollutants can be undermined by tile drainage that directly inputs water into rivers and streams before it can be filtered by vegetation in the riparian zone (Osbourne and Kovaic, 1993). Therefore, if BMPs are found to be ineffective or not functioning properly, tile drainage may provide insight as to why. The extent of tile drainage was calculated for each catchment in ArcGIS by intersecting the catchment areas with a tile drainage layer provided by the Ministry of Agriculture, Food and Rural Affairs (OMAFRA, 2010).

3.5 BMP Metrics

Metrics were used to summarize BMPs both within and among BMP types, depending on the BMP metric and BMP itself (Table 3.1). Due to the large number of BMP metrics calculated in this study, BMPs were grouped into two categories to determine if altering how the BMPs were described influenced their association with the benthic invertebrate community structure. First, a Summary group averaged all of the BMPs into one overall metric score for each catchment (e.g., BMP flow distance to sampling point, BMP % flow accumulation). Second, a Type group analyzed each BMP type to understand the influence that different types of BMPs may have on water chemistry and benthic invertebrates (e.g., MS % flow accumulation, LAR % flow accumulation, EC % flow accumulation). The metrics themselves were used to describe either the abundance or spatial location of BMPs within each catchment. BMP abundance was described by the following metrics: BMPs/farm, BMPs/km^2, % riparian area, % river protected, and % river with LAR. BMPs/farm is a proportional measure of the degree of BMP implementation within a catchment whereas BMPs/km² is a measure of the density of BMPs within a catchment, irrespective of the slight variations in catchment size. For example, a catchment may have a low BMPs/km² value, but a high BMPs/farm value if there are very few farms within the catchment. These two metrics did not include riparian vegetation because of difficulties quantifying the different sizes and types of riparian vegetation buffers. % riparian area accounted for how much of the entire catchments 30 m riparian buffer had riparian vegetation present. % river with LAR was a measurement of the proportion of river with LAR adjacent to it. Similarly, % river protected was a length measurement that determined the proportion of river within each catchment that had either LAR and/or riparian vegetation along the edge of it.

To assess whether the spatial location and subsequent changes in the DHC of BMPs could be linked with improved ecosystem conditions, the following BMP metrics were used: 1) distance to stream; 2) distance to sampling point; 3) mean flow accumulation; 4) median flow accumulation, and; 5) % flow accumulation. Distance to stream was the topographical distance that water must travel overland to reach the river, whereas distance to sampling point was the topographical distance that the water must travel to the stream plus the distance in the river to where sampling occurred. A distance weighted model (DWM) with an exponential decay function of -0.5 was used to give a larger value, which meant more of an influence, to BMPs that were closer to the river. A DWM assumes that as the distance between an activity (i.e., BMP) and a sampling point increases, the influence of that activity on the ecosystem conditions in the river will decrease (Van Sickle & Johnson, 2008). Studies have shown that assessing activities and land cover using a DWM has greater predictive power on water quality and macroinvertebrate assemblages, as opposed to a linear model that assumes all areas of a catchment contribute equally to in-stream conditions (King et al, 2005; Yates et al., 2014). Only manure storage structures had the ‘distance to sampling point’ metric because LARs were consistently adjacent to the stream, which would have generated confounding results with the ‘distance to stream’ metric, and erosion control structures were too sparse to warrant calculation of the metric. Therefore, the distance to stream metric was not be applied to the BMP summary group because it was the same as the MS distance to stream metric. The mean and median flow accumulation metrics were calculated at the catchment scale for all BMPs and all BMP types. In contrast, % flow accumulation summed the area contributing surface runoff to BMPs and divided the sum by the total area of the catchment, providing a measure of the proportion of the catchment that flowed through BMPs. Riparian vegetation was not included in the flow accumulation or flow distance metrics because of the difficulty in accurately quantifying the influence of numerous different widths, vegetation types, and locations of vegetation that occurred within the 30 m buffer zone, which are known to influence the effectiveness of riparian vegetation to filter pollutants. (Kiffney et al., 2003; Muenz et al., 2006; Osborne & Kovacic, 1993).

Table 3.1: Summary of the BMP metrics developed and which BMPs they were applied to. Some metrics were BMP specific (i.e., % riparian area, % river protected, Length of LAR % river with LAR).

BMP Metrics	All BMPs	Manure Storage	Livestock Access Restriction	Erosion Control	Riparian Vegetation
BMPs/farm	Yes	Yes	Yes	Yes	No
BMPs/km²	Yes	Yes	Yes	Yes	No
% riparian area	No	No	No	No	Yes
% river protected	No	No	No	No	Yes
Distance to stream	No	Yes	No	No	No
Distance to sampling point	Yes	Yes	Yes	Yes	No
Mean flow accumulation	Yes	Yes	Yes	Yes	No
Median flow accumulation	Yes	Yes	Yes	Yes	No
% flow accumulation	Yes	Yes	Yes	Yes	No
Length of LAR	No	No	Yes	No	No
% river with LAR	No	No	Yes	No	No

3.6 Field Sampling

3.6.1 Benthic Macroinvertebrates

Benthic macroinvertebrates offer many benefits when assessing BMP effectiveness in river systems. They are a diverse assemblage with numerous species traits that can be assessed to determine changes in ecosystem state (Usseglio-Polatera et al., 2000). As seen in previous studies, when environmental conditions worsen, such as with an increase in sediment or nutrients, predictable changes are often detected in taxa richness, community composition and the presence of pollutant-intolerant taxa (Barton & Metcalfe-Smith, 1992; Barbour et al., 1996; Reynoldson et al., 1997). In comparison to physico-chemical measures for assessing water quality, which measure a snapshot of the environment at the time of sampling, biological measures can indicate previous impairment that has occurred over weeks or months (Reynoldson et al., 2012). Furthermore, benthic macroinvertebrate communities are present in a wide variety of aquatic habitats, both pristine and degraded (Barton & Metcalfe-Smith, 1992), and can be sampled and identified relatively quickly and at minimal cost (Reynoldson et al., 2012; Whiles et al., 2000).

In this study, aquatic macroinvertebrates were sampled using the Canadian Aquatic Biomonitoring Network (CABIN) protocol for sampling aquatic macroinvertebrates in wadeable streams (Reynoldson et al., 2012). In brief, the CABIN protocol is a national standardized sampling protocol that recommends using the travelling kick method to sample benthic macroinvertebrates in wadeable streams and rivers. This technique uses a triangular net (400 microns mesh size) that is dragged upstream along the bottom of the stream or river as the substrate is disturbed to dislodge any benthic invertebrates into the net. Each sample was collected over a three minute period to standardize sampling effort. Every microhabitat within the sampling reach was sampled for a duration proportional to its occurrence within that reach (i.e., more prevalent habitats sampled longer). A reach was defined as six times the bankfull width and encompassed common microhabitat types, such as pools, riffles, and runs. All sampling was conducted in early autumn (September/October) because the majority of benthic macroinvertebrates are typically in their aquatic life stage during this time which improves capture rates and provides the best opportunity to sample the entire benthic invertebrate community (Reynoldson et al., 2012). Furthermore, benthic invertebrates are more easily identified in early autumn as they have grown and matured throughout the summer (Reynoldson et al., 2012). In accordance with the CABIN protocol, collected samples were preserved in 90% ethanol and later subsampled and counted using a Marchant box (Marchant, 1989) until a minimum of 300 individuals of 5% of the sample was recorded and identified to the lowest possible taxonomic level, which was often the genus level. To ensure that taxonomic resolution was consistent amongst all sampled catchments, taxonomic adjustments were conducted using a 25/75 rule similar to that used by Vlek et al. (2004). Under this rule if more than 25% of the individuals of a taxon for all sites were limited to family level identification, then individuals of that family identified to the genus level were aggregated to the family level. If more than 75% of the individuals for a taxon for all sites were identified to the genus level, than the remaining family taxa were proportionally distributed to the identified genera (i.e., more abundant genus taxa would receive more of the distributed family taxa).

3.6.2 Habitat Assessment

Habitat quality was assessed using the United States Environmental Protection Agency rapid habitat assessment protocol (Barbour et al., 1999). In brief, the protocol is a qualitative assessment of ten physical habitat parameters that are important for aquatic biota (i.e., epifaunal substrate, pool substrate characterization, pool variability, sediment deposition, channel flow status, channel alteration, channel sinuosity, bank stability, vegetation protection, riparian vegetation zone width). These physical habitat parameters are independent of both water quality and biota, and include characteristics of the stream channel and neighboring riparian zone. Habitat parameters are scored out of 20, with lower scores indicating an increase in impairment or degradation (Appendix A). Consistent scoring amongst sites is critical to ensure minimal variation in the ranking of each parameter (Barbour et al., 1999). To maintain this needed consistency, one trained person completed the scoring of each parameter at all the sampled sites in my study.

3.6.3 Water Chemistry

Grab water samples were collected from all 30 sites over a two-day period in early November, 2013. The two-day sampling period minimized temporal variability due to climatic conditions. Similar to the benthic macroinvertebrate sampling, grab samples were collected using the CABIN protocol to measure major forms of nitrogen (i.e. total, nitrate, nitrite and ammonia) and phosphorus (i.e., total, dissolved, and soluble reactive). In brief, 250 ml bottles were rinsed at the sampled site, submersed in the center of the stream, and filled, leaving a small air pocket at the top of each bottle. The bottles were then labelled, stored in a cooler, and delivered to a laboratory for analysis within 24 hours of sampling (Reynoldson et al., 2012). An in-situ field probe was used to measure specific conductivity. A Hoskin Scientific Professional Plus probe (Model: Pro 10102030) was placed in an area with flowing water and given time to stabilize. Total suspended solids (TSS) and turbidity samples were also collected using grab samples following the CABIN protocol and were assessed in the Freshwater Ecosystem and Assessment

Research (FEAR) lab at Western University. For TSS and turbidity analyses, samples were collected in 1 L Nalgene bottles. In the lab, TSS was analyzed by filtering 900 ml of water through Whatman grade 934AH filter paper. Filter papers were then placed on aluminum weighing dishes and dried overnight in an oven at 105ºC. The next day, the filter paper was weighed on balance to a precision of 0.0001 g. Turbidity was measured using a Turner Designs Trilogy Laboratory Fluorometer. 10 ml aliquots of sample water were assessed following calibration of the fluorometer. Samples were analyzed 9 times and the average of the measurements calculated to account for inherent variability in the sample turbidity associated with particulates.

3.7 Data Analysis

Corrected abundance data was used to analyze the community structure of the collected benthic invertebrates. Benthic invertebrate data was corrected after subsampling occurred, which was done by dividing the number of individual benthic invertebrates in each taxa by the proportion of each sample that was subsampled (e.g., 5 individuals/0.05 subsampled = 100 individuals after abundance is corrected for). The corrected abundance was then natural log transformed to normalize the data. Rare taxa, defined as taxa present at less than 5% of the sample sites, were removed prior to analysis on the benthic invertebrates. This was done because abundant species tend to give more reliable results that are representative of the entire community, whereas the inclusion of rare species often contributes little additional information about the community or provides redundant, less reliable results (*see review by* Cao et al., 2001).

To normalize all of the variables and collected samples (i.e., BMP metrics, water parameters, habitat characteristics, land use variables), data transformations were performed. All proportional measures of BMP metrics, landscape variables, water parameters, and habitat characteristics were arcsin transformed, while the corrected benthic invertebrate taxa and all other variables were natural log transformed.

To understand and visualize which BMP metrics explained the most variation amongst the 30 sampled catchments, principal coordinate analysis (PCoA) was used. PCoA uses a dissimilarity/distance matrix that preserves the Euclidean distance to plot the BMP metrics and then uses axes to explain the variability amongst the different metrics (Gotelli & Ellision, 2004). To assist in the explanation of the variation in a large dataset, PCoA reduces complex datasets to a few key variables, or composite groups of variables, which are expressed as axes (Gotelli & Ellision, 2004). The direction of the first axis explains the most variation, whereas the second axis explains the next greatest variation, but in an orthogonal direction to the first axis. Additional axes explain subsequent variation in the data until the desired proportion of variation is explained. The PCoA was conducted using the vegan package in R version 3.1.0 (Oksanen et al., 2013; R Core Team, 2014).

An unconstrained detrended correspondence analysis (DCA) (Hill & Gauch, 1980) was conducted on the benthic invertebrate community data to determine if the data was unimodally or linearly distributed (Borcard et al., 2011a), which would influence how the benthic invertebrate data would be further analyzed. Based on the resultant length of the axes (axis 1 = 2.0049, axis 2 = 1.9362, axis 3 = 1.22602, axis 4 = 1.58759) a linear method was deemed appropriate for analysis of the data. As such, principal component analysis (PCA) (Borcard et al., 2011b) was performed. PCA is similar to PCoA in that it plots the data points on a matrix and finds the strongest axes that explain the most variation in the original variables (Gotelli & Ellision, 2004). However, PCA assesses the similarity amongst the different variables in the dataset. To run the PCA, the vegan package in R version 3.1.0 was used (Oksanen et al., 2013; R Core Team, 2014).

3.7.1 Association between BMP Metrics and Water Parameters

To determine which BMP metrics were associated with water parameters, least squared regressions were performed for each individual water parameter. Separate regressions were performed on both the BMP Type and Summary groups due to collinearity amongst the 2 groups (VIF > 5). A backwards stepwise regression technique was used with a confidence interval of 0.95 and probability of 0.15. This technique showed the power and direction of influence that each BMP metric within each group had on the water parameters. The regressions were conducted using SYSTAT 13 Version 13.00.05 (Systat Software, 2008).

3.7.2 Stepwise Ordination Regressions and Variance Partitioning

Before the variance partitioning analysis was conducted, all variables and collected samples were assessed for collinearity to ensure that the statistical outputs were accurate and stable. A variance inflation factor (VIF) >5 was used to determine if variables were collinear ($VIF_x = 1/1\text{-}R_x^2$). The procedure was performed in R using the “vif.cca” function (R Core Team, 2014). From the Summary BMP group, none of the original 5 BMP metrics (i.e., BMPs/farm. BMPs/km², BMPs mean flow accumulation, BMPs % flow accumulation, BMPs distance to sampling point) were determined to be collinear. From the BMP Type group, 9 of the original 18 metrics had to be removed from further analysis due to collinearity. The metrics that remained in the BMP Type group were MS/farm, MS mean flow accumulation, MS median flow accumulation, MS distance to sampling point, LAR/farm, LAR % flow accumulation, EC/farm, EC % flow accumulation, and % riparian area. When water parameters were assessed for collinearity issues, 3 of the 9 parameters had to be removed due to collinearity issues. The water parameters that remained were NH4+, NO2+NO3, SRP, turbidity, TSS, and specific conductivity. None of the landscape variables (i.e., % agriculture, % urban, % natural area, % till soil, % tile drainage, # of farms) exhibited significant collinearity. However, collinearity analysis revealed 2 pairs of habitat variables with significant collinearity. The habitat characteristics that remained were epifaunal substrate, pool substrate characterization, sediment deposition, channel sinuosity, bank stability, vegetation protection and riparian vegetation.

All environmental variables that may have influenced the benthic invertebrate community were assigned to one of four groups (i.e., BMPs metrics, landscape variables, water parameters, and habitat characteristics) and individual forward stepwise ordination regressions were conducted to determine the environmental variables in each group that were significantly ($P > 0.1$) associated with the benthic invertebrate community. The two BMP metric groups (i.e., Summary and Type) were assessed in separate regressions. Stepwise ordinations were conducted using the vegan package in R version 3.1.0 (Oksanen et al., 2013; R Core Team, 2014). A bidirectional stepwise ordination regression ran through permutations (# of steps =999; # of permutations = 999) to assess all of the variables in each group independently and determine if any were significant. If more than one variable was insignificant, it removed the least significant variable and repeated the initial assessment until there was only significant variables remaining and the model did not change during one step. Once completed, a list of significant variables from each group was compiled to understand which of the independent variables were associated with the changes seen in the benthic invertebrate community.

Variance partitioning was then conducted to determine the relative amount of variance in the observed benthic community that each group of variables explained (i.e., BMP metrics, habitat characteristics, water parameters, and landscape variables). Variance partitioning calculates both the variance explained by each individual variable group and the interactions that occur between the variable groups (Figure 3.4). Changes observed in the benthic community amongst the different sites must account for all potential driving variables and shared spatial influences to ensure that an accurate representation of the BMPs influence is being reported (Borcard et al., 1992). Variance partitioning treats all explanatory variables as non-mutually exclusive contributors to the changes seen in the dependent variables (i.e., invertebrates). Variance partitioning was conducted using redundancy analysis (RDA) (Borcard et al., 2011a) to quantify the individual contribution that each variable group had in shaping the benthic invertebrate community along with the contributions from the interactions amongst the environmental variables. Two separate variance partitioning analyses were run where the BMPs were represented by the significant variables from either the Summary BMP or Type BMP groups. The significant variables from each of the remaining environmental groups (i.e., habitat characteristics, water parameters, and land use variables) were the same for both analyses. Variance partitioning was conducted in R using the vegan package in R version 3.1.0 (Oksanen et al., 2013; R Core Team, 2014).

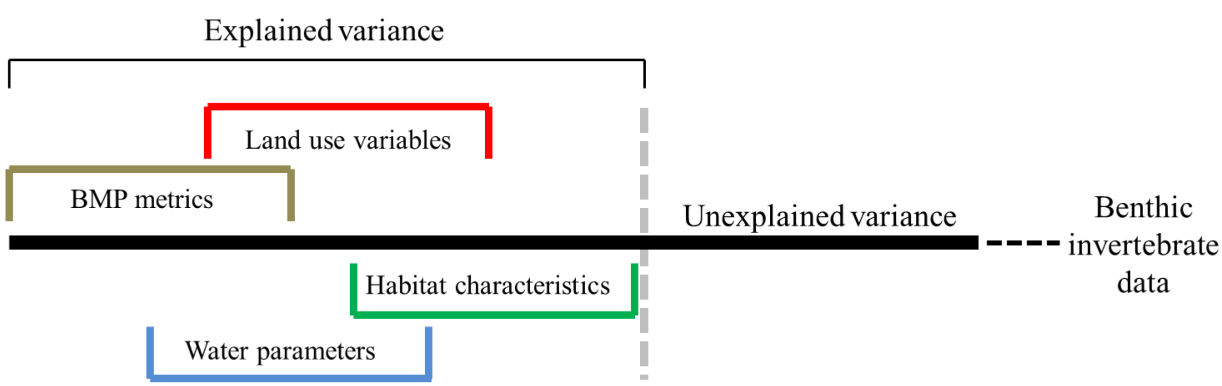

Figure 3.4: Conceptual diagram illustrating the output of variance partitioning. The variance explained by each individual variable group and the interactions that occur between the variable groups are calculated using variance partitioning. Adapted from Borcard et al. (1992).

4.0 Results

4.1 Land Use Summary

Agriculture was the dominant land use type in each catchment with a mean of 88% (Table 4.1). Urban and natural land uses comprised on average only 1% of the catchment area for urban and 9% for natural lands. However, both urban and natural lands were variable, with coefficient of variations (CV) of 1.05 and 0.60, respectively. A median of 88% for the percentage of till soil confirmed that till was the dominant soil type for all catchments. Tile drainage and # of farms were not controlled for prior to site selection, and as such these two descriptors exhibited the greatest variation. Tile drainage had a range of 8% to 83% and a CV of 0.43, while the # of farms in each catchment ranged from 3 to 22 (CV = 0.49).

Table 4.1: Descriptive statistics for landscape descriptors for 30 sampled headwater catchments in the Nith and Conestoga subwatersheds of the Grand River Basin, Ontario, Canada.

Land Use Variable	Mean	Min.	Max.	Med.	St. Dev.	CV
% Agriculture	0.88	0.76	0.97	0.89	0.06	0.06
% Urban	0.01	0.00	0.05	0.01	0.01	1.05
% Natural	0.09	0.02	0.24	0.08	0.05	0.60
% Till Soil	0.86	0.67	0.99	0.88	0.08	0.10
% Tile Drained	0.46	0.08	0.83	0.45	0.20	0.43
# of Farms	10.13	3.00	22.00	9.00	4.96	0.49

4.2 BMP Abundance and Composition

There were a total of 129 structural BMPs (i.e., MS structures, LARs, EC structures) in the 30 sampled catchments (Table 4.2). The maximum number of BMPs within a single catchment was 16. However, the mean for all 30 catchments was 4.3, with a median of 2.5. MS structures were the most common BMP used in the Nith and Conestoga subwatersheds. 25% of all farms in the study area had a manure storage structure. Additionally, manure storage (MS) structures accounted for more than half of the BMPs assessed in this study (n = 79). The number of livestock access restrictions (LAR) per catchment was low ($\bar{x}$ = 1.33) and variable (s = 1.92). The length of LARs was also variable, ranging in length from 0 m to over 2850 m and as such exhibited a high standard deviation (738.56 m). On average, only 4% of the river systems within each catchment were being protected by LARs. Erosion control (EC) structures were the least common BMP, having a mean of 0.17 per catchment. In contrast, on average nearly a quarter (23%) of the length of stream network in the sampled catchments was buffered by riparian vegetation.

Table 4.2: Statistical summary of the BMP abundance for the 30 sampled catchments. '# of BMPs' summarizes the three structural BMP types being assessed in this study (manure storage structures, livestock access restrictions, and erosion control structures), but does not include riparian vegetation in this count.

	Count	Mean	Min.	Max.	Med.	St. Dev.	CV
# of BMPs	129	4.30	1.00	16.00	2.50	4.09	0.95
# of Manure Storage Structures	79	2.63	0.00	10.00	2.00	2.40	0.91
Manure Storage Structures/# of Farm		0.25	0.00	0.62	0.25	0.15	0.59
# of Livestock Access Restriction	40	1.33	0.00	8.00	0.50	1.92	1.44
Length of Livestock Access Restriction		478.25	0.00	2850.25	31.59	738.56	1.54
% River with Livestock Access Restriction		0.04	0.00	0.20	0.01	0.05	1.44
# of Erosion Control Structures	10	0.17	0.00	2.00	0.00	0.42	2.51
% Buffer with Riparian Vegetation		0.21	0.00	0.77	0.19	0.16	0.79
% River Protected (with Riparian Vegetation)		0.23	0.00	0.52	0.21	0.15	0.64
% River Protected (with Riparian Vegetation + LAR)		0.27	0.00	0.61	0.24	0.15	0.56
BMPs/farm		0.39	0.12	0.94	0.33	0.21	0.52
BMPs/km²		0.57	0.12	2.24	0.37	0.48	0.85

Principal coordinates analysis (PCoa) of the BMP summary metrics resulted in the identification of one important axis describing nearly all the variation (92.5%) in BMP use within the sampled catchments. The first axis was associated almost exclusively with BMP mean flow accumulation (loading = -2.35), whereas loadings for the remaining metrics were small on the first axis (BMPs/farm = -0.08, BMPs/km^2 = -0.10, BMP % flow accumulation = -0.03, BMP distance to sampling point = -0.03). PCoA on the metrics describing the separate BMP types resulted in the first 3 axes explaining nearly 98% of the variation in the distribution of BMP types among the sampled catchments. The first axis explained 84% of the total variation and was primarily associated with length of LAR (loading = 3.98) and LAR mean flow accumulation (loading = 0.99). The second axis explained 9% of the total variation and was associated with EC mean flow accumulation (loading = 1.31). The third axis explained 5% and was associated with MS mean flow accumulation (loading = -0.23).

4.3 Degree of Hydrological Connection for BMPs

The degree of hydrological connection (DHC) for BMPs and individual BMP types varied across the sampled catchments (Table 4.3). When BMPs were summarized together, the range in flow accumulation varied between 52 m^2 and 921,176 m^2, whereas flow distance varied between 1467 m^2 and 6027 m^2. MS structures and LAR's had similar means for flow accumulation (MS = 3691 m^2, LAR = 3385m^2) and flow distance to sampling point (MS = 3406 m, LAR = 2864 m), but also had large ranges (MS flow accumulation = 176 m^2 – 1983 m^2, MS distance to sampling point = 1356 m – 5210 m, LAR flow accumulation 52 m^2 – 11393 m^2, LAR distance to sampling point = 1579 m – 7237 m) in these variables as well. EC structures were the fewest BMP in abundance (n = 10), but had the largest maximum flow accumulation of all BMP types (921176 m^2).

Table 4.3: Summary statistics describing the degree of hydrological connection as measured by flow accumulation and flow distance metrics for all BMPS together and for individual BMP types for 30 sampled Grand River Basin headwater catchments in the Nith and Conestoga subwatersheds. Livestock access restrictions and erosion control structures do not have a 'flow distance to stream' measurement due to their common placement adjacent to rivers and low abundance, respectively. '# of BMPs' summarizes the three structural BMP types being assessed in this study (manure storage structures, livestock access restrictions, and erosion control structures), but does not include riparian vegetation.

	Count	Mean	Min.	Max.	Med.	St. Dev.	CV
All BMPs	129						
Flow accumulation (m^2)		26851	52	921176	2259	134035	4.99
Flow distance (m)		3238	1467	6027	3309	1120	0.35
Manure Storage Structures	79						
Flow accumulation (m^2)		3691	176	19839	2252	4535	1.23
Flow distance to Sampling Point (m)		3406	1356	5210	3461	1050	0.31
Flow distance to stream (m)		449	69	1713	327	388	0.86
Livestock Access Restrictions	40						
Flow accumulation (m^2)		3385	52	11393	2891	2822	0.83
Flow distance (m)		2864	1579	7237	2231	1478	0.52
Erosion Control Structures	10						
Flow accumulation (m^2)		149015	176	921176	1644	324591	2.18
Flow distance (m)		2667	1328	4978	2226	1354	0.51

4.4 Habitat Assessment

The majority of the physical habitat parameters assessed using the rapid habitat assessment protocol scored in the mid to low range for their means (4-13). All habitat characteristics also exhibited large ranges (≥12). Both mean and median results were similar for most parameters (i.e., <3 difference). Only epifaunal substrate, sediment deposition, bank stability, and vegetation protection had means and medians that were above 10. The highest mean score was for vegetation protection ($\bar{x}$ = 12.63), followed closely by epifaunal substrate ($\bar{x}$ = 12.27). The lowest mean scores were for channel sinuosity ($\bar{x}$ = 5.27) and pool variability ($\bar{x}$ = 4.30), which also had large CV's of 0.93 and 0.99, respectively.

Table 4.4: Descriptive statistics for habitat parameters assessed using the Environmental Protection Agency's Rapid Bioassessment Protocol at 30 headwater catchments in the Nith and Conestoga subwatersheds of the Grand River Basin. Highest possible score (least degraded) for each habitat parameter was 20.

Habitat Characteristic	Mean	Min.	Max.	Med.	St. Dev.	CV
Epifaunal Substrate	12.27	1.00	19.00	12.00	5.36	0.44
Pool Substrate	6.80	0.00	18.00	8.50	6.15	0.90
Pool Variability	4.30	0.00	12.00	3.00	4.24	0.99
Sediment Deposition	11.17	2.00	19.00	13.50	5.31	0.48
Channel Alteration	8.13	0.00	17.00	7.50	5.54	0.68
Channel Sinuosity	5.27	0.00	14.00	4.00	4.88	0.93
Bank Stability	12.20	2.00	20.00	12.00	4.49	0.37
Vegetation Protection	12.63	5.00	18.00	13.00	3.78	0.30
Riparian Vegetation	6.40	0.00	15.00	4.00	4.68	0.73

4.5 Water Chemistry

All phosphorus and nitrogen parameters were highly variable among the sampled catchments. In particular, NH4+ (0.00 – 0.13 mg/L) and SRP (0.01 – 0.29 mg/L) both had CVs of 0.91. The majority (86.3%) of TN ($\bar{x}$ = 5.54 mg/L) came from NO2+NO3 ($\bar{x}$ = 4.78 mg/L). All P forms had high CVs (SRP = 0.91, TP = 0.83, TDP = 0.94), with similar means (SRP = 0.07, TP = 0.10, TDP = 0.08). Specific conductivity was fairly consistent across most sites as indicated by a similar mean (660.67) and median (662.15), along with a low CV (0.09). TSS ($\bar{x}$ = 7.68 mg/L) and turbidity ($\bar{x}$ = 12.44 ntu) were slightly variable with CVs of 0.50 and 0.46, respectively.

Table 4.5: Summary of water chemistry results collected from all 30 sampled catchments using the grab sample technique.

Water Parameters	mean	min.	max.	med.	st. dev.	CV
NH_4^+ (mg/L)	0.04	0.00	0.13	0.03	0.04	0.91
NO_2+NO_3 (mg/L)	4.78	1.29	8.50	5.08	1.70	0.36
TN (mg/L)	5.54	1.91	10.28	5.80	1.85	0.33
SRP (mg/L)	0.07	0.01	0.29	0.05	0.07	0.91
TP (mg/L)	0.10	0.02	0.34	0.06	0.08	0.83
TDP (mg/L)	0.08	0.01	0.32	0.05	0.07	0.94
TSS (mg/L)	7.68	2.99	19.87	7.12	3.86	0.50
Turbidity (ntu)	12.44	1.56	26.39	14.09	5.71	0.46
Spec. Cond. (µS/cm)	660.67	510.10	801.00	662.15	60.07	0.09

Results of the regression between water parameters and the BMP Type group showed that the N forms had the most variation explained by BMP metrics (i.e., MS median flow accumulation, LAR % flow accumulation, % riparian area) with R^2 values over 0.38 (NH_4^+ = 0.45, NO_2+NO_3 = 0.38, TN = 0.38) (Table 6). All N forms were negatively associated with the BMP metrics, as shown by the std coefficient, aside from the relationship between NH_4^+ and LAR % flow accumulation (std. co. = 0.37). % riparian area was the only metric to be associated with the three P forms (i.e., SRP, TP, TDP). The P forms were weakly explained by the BMP metric, with R^2 values less than 0.23. However, all P forms were negatively associated with % riparian area (SRP = -0.46, TP = -0.47, TDP = -0.45). Specific conductivity had the same variation explained (R^2 = 0.38) as the N forms, and was negatively associated with LAR % flow accumulation (std. co. = -0.30) and % riparian area (std. co. = -0.52). TSS was the least explained water parameter, with an R^2 value of 0.14. Only % riparian area was associated with TSS (std. co. = -0.37). Lastly, turbidity was moderately explained (R^2 = 0.29) by MS median flow accumulation (st. co. = -0.38) and % riparian area (std. co. = -0.44). Overall, only 3 BMP type metrics were significantly associated with the water parameters (MS median flow accumulation, LAR % flow accumulation, % riparian area). % riparian area was present in all 9 regressions, and reduced water parameters in all cases. Additionally, the 3 forms of N were all explained by the same 3 metrics. LAR % flow accumulation was significant in explaining the variation in 4 of the 9 water parameters (NH4+, NO2+NO3, TN, spec. cond.), although the positive std. coefficient for NH4+ indicated an increase in N with the presence of the metric. MS median flow accumulation was also significant in explaining the variation in 4 of the 9 water parameters (NH4+, NO2+NO3, TN, turbidity), showing a reduction in the water parameter in all cases when it was present.

Table 4.6: Results of the regression analysis between the water parameters and BMP type metrics

Parameter	Significant Predictor(s)	Std. Coefficient	P-Value	R^2
NH4+	MS median flow accumulation	-0.29	0.001	0.45
	LAR % flow accumulation	0.37		
	% riparian area	-0.56		
NO2+NO3	MS median flow accumulation	-0.39	0.005	0.38
	LAR % flow accumulation	-0.39		
	% riparian area	-0.35		
TN	MS median flow accumulation	-0.44	0.006	0.38
	LAR % flow accumulation	-0.32		
	% riparian area	-0.38		
SRP	% riparian area	-0.46	0.011	0.21
TP	% riparian area	-0.47	0.009	0.22
TDP	% riparian area	-0.45	0.013	0.20
Spec. Cond.	LAR % flow accumulation	-0.30	0.002	0.38
	% riparian area	-0.52		
TSS	% riparian area	-0.37	0.043	0.14
Turbidity	MS median flow accumulation	-0.38	0.012	0.29
	% riparian area	-0.44		

Results of the regression between water parameters and the BMP Summary group showed that only 4 parameters (NH4+, SRP, TP, TDP) were significantly associated with the BMP metrics. All 4 water parameters were weakly explained by the BMP metrics, with R^2 values of less than 0.20. NH_4^+ had little variation explained ($R^2 = 0.16$) by BMP % flow accumulation, and the relationship was positive (std. co. = 0.40). As indicated by the std. coefficient, all P forms were negatively associated with BMPs/farm (SRP = -0.60, TP = -0.58, TDP = -0.58), while BMPs/km² was positively associated with all P forms (SRP = 0.62, TP = 0.60, TDP = 0.59). Specific conductivity, TSS, and turbidity were not associated with any BMP metrics.

Table 4.7: Results of the regression analysis between the water parameters and BMP summary metrics

Parameter	Significant Predictor(s)	Std. Coefficient	P-Value	R^2
NH4+	BMP % flow accumulation	0.40	0.027	0.16
NO2+NO3	N.S.	-	-	-
TN	N.S.	-	-	-
SRP	BMPs/farm	-0.60	0.071	0.19
	BMPs/km²	0.62		
TP	BMPs/farm	-0.58	0.084	0.18
	BMPs/km²	0.60		
TDP	BMPs/farm	-0.58	0.013	0.16
	BMPs/km²	0.59		
Spec. Cond.	N.S.	-	-	-
TSS	N.S.	-	-	-
Turbidity	N.S.	-	-	-

4.6 Benthic Invertebrate Composition

108 taxa were identified across the 30 sampled catchments. Of these 108 taxa, 46 were identified as rare. The average number of taxa at each site was 25, with a standard deviation of 5.5 and a range from 16-40. The average corrected abundance at each site was 6256, with a standard deviation of 6483 and a range from 632-35,300. The dominant taxa at each site accounted for an average of 44% of the taxa abundance, with a standard deviation of 19% and a range from 15%-91%. The 3 most common taxa were two genera of the Chironomidae family,

Thienemannimyia (present at 100% of sites) and Microtendipes (present at 90% of sites), and the Isopod genus Lirceus (present at 90% of sites). The 3 most abundant taxa were Dubiraphia sp. (14988 individuals – present at 80% of sites), Thienemannimyia (6749 individuals), and Hyalella sp. (4225 individuals – present at 50% of sites).

4.6.1 Potential drivers of the benthic macroinvertebrate community structure

Principal component analysis revealed only small differences in the composition of the benthic community among the sampled catchments (Figure 4.1). Accordingly, the first four axis of the PCA explained 47.1% of the variation in the benthic community, with only the first two axes (31%) showing a discernable pattern in the benthic invertebrate characteristics. The first axis (16.3%) was associated with differences in the abundance of Gyraulus sp. (loading = -0.13), Pisidium sp. (loading = -0.12), Micropsectra sp. (loading = -0.13), Orthocladius (loading = -0.11), and Optioservus.sp. (loading = 0.10). The taxa with the more negative loadings are pollution-tolerant, widespread species that are commonly found in slow-moving, silty habitats (Brown, 1991; Lenat, 1993; McHahon, 1991; Merritt et al., 2008; Strayer, 1990; Yuan, 2004), whereas the one with the most positive loading (i.e., Optioservus.sp) prefers fast flowing water, or riffles (Peckarsky et al., 1990). Therefore, the first axis was defined as distinguishing between taxa that prefer slow flowing water (e.g., pools) and those that prefer faster flowing water (e.g., riffles). The main benthic invertebrates associated with the second axis were Quistradrilus multisetosus (loading = -0.17), Micropsectra sp. (loading = 0.11), Cheumatopsyche sp. (loading = 0.07), Simulium sp. (loading = 0.08), Hyalella sp. (loading = -0.08) and Lirceus sp. (loading = -0.11). The second axis distinguished between different trophic relationships, with Micropsectra sp. and Cheumatopsyche sp. being collectors (gatherers or filterers), and Quistradrilus multisetosus, Hyalella sp., Quistradrilus multisetosus, and Lirceus sp. being scavengers (Merritt et al., 2008; Strayer, 1990; Peckarsky et al., 1990).

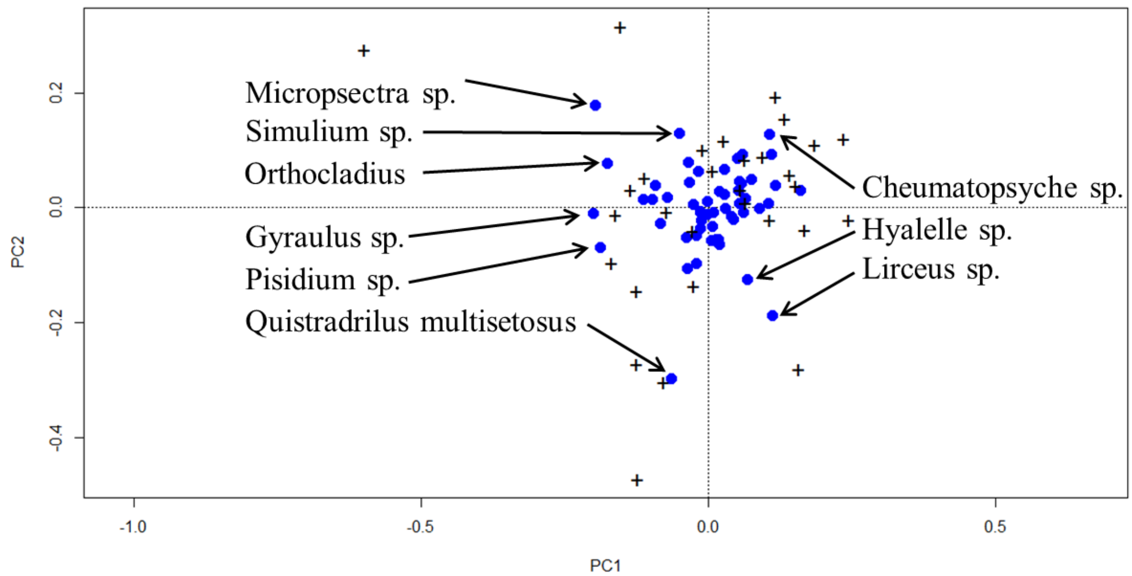

Figure 4.1: Ordination of benthic community composition based on principal component analysis indicating the arrangement of the benthic community (blue dots) throughout the sampled catchments (+). The first four axes of the PCA explained 47.1% of the variation in the benthic community.

4.7 Stepwise Ordination Regressions and Variance Partitioning of Significant Environmental Variables

The stepwise ordination regression on the BMP summary metrics found that 2 of the 5 metrics were significantly associated with the benthic community composition. The 2 significant metrics from the BMP Summary group were BMPs/km² ($p = 0.03$) and BMPs/farm ($p = 0.02$). The stepwise ordination regression on the BMP Type metrics found that 4 of the 18 metrics were significantly associated with the benthic community composition. The 4 significant metrics from the BMP Type group were MS/farm ($p = 0.07$), EC/farm ($p = 0.06$), LAR/farm, ($p = 0.04$), and LAR % flow accumulation ($p = 0.01$).

Of the remaining three groups of environmental variables, a total of 6 variables were found to be significantly associated with benthic macroinvertebrates. Of the land use variables, the only significant variable was # of farms ($p = 0.01$). Of all the water parameters, 3 significant variables were identified (SRP; $p = 0.01$, turbidity; $p = 0.03$, and specific conductance; $p = 0.03$). Lastly, there were 2 significant habitat characteristics (sediment deposition; $p = 0.03$ and channel sinuosity; $p = 0.01$).

Variance partitioning analysis revealed that less than 20% of the variation in benthic community composition could be explained by the described environmental variables regardless of whether BMPs were summarized or kept as individual types. However, the total amount of variation explained did increase by almost 5% when BMPs were assessed by individual type along with the environmental variables (17.9% variance explained; Figure 4.2A) as opposed to being aggregated into summary metrics (13.4% variance explained; Figure 4.2B). In the variance partitioning for the BMP type group, the BMPs individually explained 5.7 % of the variation in the benthic invertebrate community. The individually explained variance from each of the environmental variables was 3.9% for the water parameters, 1.6% for the habitat characteristics, and 0.9% for the land use variables. The interactions amongst the variable groups increased all of the explained variances, but only marginally (0.4%) for BMPs (BMP Types = 6.1%, water parameters = 9.9%, habitat characteristics = 6.3%, land use variables = 3.3%).

In the variance partitioning for the BMP Summary group, the BMPs individually explained 1.1% of the variation in the benthic invertebrate community. The individually explained variance from each of the environmental variables was 5.4% for the water parameters, 0.9% for the habitat characteristics, and 0.1% for the land use variables. Similar to the BMP Type model, when the interactions amongst the variables groups was accounted for, the explained variation for all the groups increased, although BMPs still only accounted for a small portion of the explained variance (BMP Summary = 4.1%, water parameters = 9.9%, habitat characteristics = 6.3%, land use variables = 3.3%).

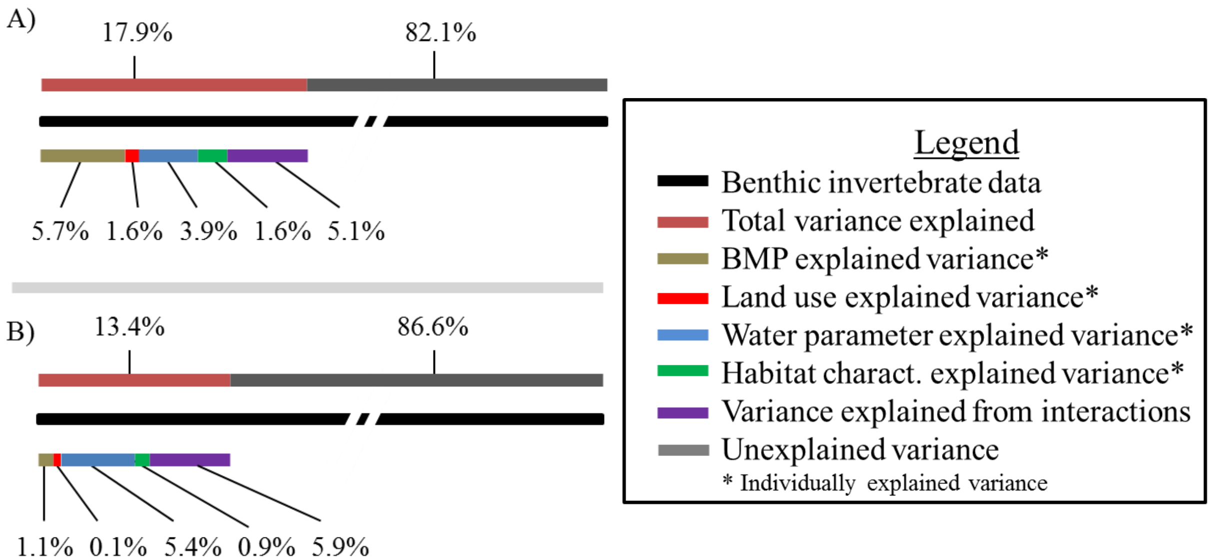

Figure 4.2: Representation of the total variance explained in both the BMP Type model (A) and the BMP Summary model (B), along with the individual variance explained by each variable group. Note: none of the interactions amongst the individual groups are included in the figure; therefore, the total variance is more than the sum of the individually explained variances.

5.0 Discussion

Benthic macroinvertebrates and water parameters were used as indicators of ecosystem conditions to assess the effectiveness of BMPs in mitigating the effects of agricultural pollutants in 30 headwater streams in the Grand River Basin. BMP metrics, particularly those measuring the extent of riparian vegetation, were correlated with reductions in in-stream nutrient and sediment concentrations, suggesting that BMPs are mitigating losses of sediment and nutrients to regional streams. However, results of my study indicated that variation in the structure of benthic macroinvertebrate communities was only poorly explained by BMP abundance and location, suggesting that current BMPs may be insufficient to overcome the effects of agricultural stressors on benthic invertebrate communities. The absence of a strong association between BMPs and benthic macroinvertebrates may be due to a threshold effect in which water quality and in-stream habitat conditions are still beyond a state that would allow sensitive taxa to recolonize the streams. Additional reasons for a lack of response from benthic macroinvertebrates are that BMPs in general were too sparse, too few BMPs were located in hydrologically connected areas, conservation goals are solely focused on improving water quality, and/or there were a lack of nearby source populations of sensitive benthic taxa available to repopulate streams.

5.1 Structural BMP Composition within Headwater Catchments

The abundance of structural BMP types in the headwater catchments of the Nith and Conestoga subwatersheds is currently low, with an average of 0.39 BMPs/farm in the 30 sampled catchments. The low number of structural BMPs may be due to the voluntary nature of the RWQP program, which others have cited as a possible reason for low BMP implementation rates

(Wang et al., 2002; Yates et al., 2007). The regional rate of BMP adoption by farmers is even lower (3.5 BMPs/catchment throughout all Nith and Conestoga catchments) than observed in this study as catchments included in this study had a slightly higher number of BMPs when compared to the regional average (4.2 BMPs/catchment in my study). However, the BMP implementation rates from my study were comparable to or higher than past studies that have assessed BMPs. For example, Wang et al. (2002) had a total of 6 structural BMPs (i.e., manure storage and barnyard control systems) in their 2 studied catchments and Yates and Bailey (2007) averaged 1.59 BMPs/catchment throughout 32 catchments in the Upper Thames River Watershed in southern Ontario. The higher implementation rates from the RWQP may be due to the emphasis on benefits to the farmers, such as sufficient funding for BMPs (i.e., ≥ 50% funding for MS structures, EC structures, LARs), or the collaboration with farming agencies (i.e., Ontario Farm Environmental Coalition, Ontario Federation of Agriculture, and Ontario Soil and Crop Improvement Association), which help to further promote the RWQP (GRCA, 2014). However, as indicated by the low BMPs/farm value (0.39), there are likely still many farms without BMPs, which may be due to a lack of awareness by farmers on the benefits or perceived needs of conservation projects (Yates and Bailey, 2006), or of the RWQP program itself. Additionally, farmers may not be participating in BMP programs because: 1) the short-term economic loss for installing a BMP is perceived as too costly even with financial assistance; 2) there may be a lack of trust or belief that the BMPs will not inhibit farming productivity, and/or; 3) personal beliefs or attitudes towards new farming practices discourage farmers from participating (*see review by* Nazarko et al., 2005). Thus, although overall BMP implementation across the Nith and Conestoga subwatersheds appears to be higher than in previous studies, there may be numerous economic and personal factors that need to be overcome to get a larger proportion of farmers to adopt structural BMPs in the headwater catchments of the GRW.

BMP implementation across the Nith and Conestoga subwatersheds was dominated by manure storage (MS) structures, whereas livestock access restrictions (LARs) and erosion control (EC) structures were less prominent. MS structures accounted for 61% of all BMPs, excluding riparian vegetation, with LARs and EC structures accounting for 31% and 8% of all BMPs, respectively. MS structures were likely the most common BMP because they are often needed on livestock farms to store manure, regardless of the farmer's environmental concerns. Furthermore, MS structures typically have the largest amount of available grant funding for farmers to apply for from the RWQP (GRCA, 2014). Previous studies have stated that farmers are risk-adverse when it comes to changing their farming practices and tend to only adopt changes if they do not significantly interfere with their current farming operations (*see review by* Nazarko et al., 2005; Sharley et al., 2012), which may be why MS structures are so prevalent in the Nith and Conestoga subwatersheds. The simple addition of a MS structure, which may make the storage of manure easier, has a relatively simple design and function that does not significantly alter a farmer's current operation. Conversely, the installation of LARs may require additional watering mechanisms for livestock that have been restricted from streams, and EC structures may make planting or harvesting crops more difficult due to an added obstacle in their field. This may explain why LARs were not as common in my study (LAR = 40), and because they are only needed when a farm has pasturing livestock adjacent to a stream. Additionally, there were very few EC structures (n = 10), which are typically only installed when a farmer can identify an area that is prone to gullying or rilling, and views it as a serious enough problem to install a BMP. Riparian vegetation was present along 23% of all Nith and Conestoga subwatershed streams, which was higher than anticipated in these heavily farmed regions. Muenz et al. (2006) observed less than 15% vegetation cover at all 5 of their study sites, which was in an agricultural catchment in southwestern Georgia. Kamp et al. (2013) determined that 27.5% of their studied streams had riparian vegetation, and would categorize the 23% riparian vegetation from my study as a moderate amount of riparian protection in an agricultural watershed. Riparian vegetation was likely present at many of my sites because farmers did not remove existing riparian vegetation, as opposed to actively planting trees and grasses to create a vegetation buffer. Nevertheless, riparian vegetation is a low cost, low maintenance BMP that has been shown to be associated with improvements in water quality in previous studies (Dunne et al., 2011; Muenz et al., 2006; Osborne and Kovavic, 1993). Therefore, the dominant BMP types in the Nith and Conestoga subwatersheds were likely implemented based on their convenience (i.e., riparian vegetation) and a general need to improve farming operations (i.e. MS structures), as opposed to an explicit desire to protect aquatic systems.

The distribution of structural BMPs throughout the Nith and Conestoga subwatersheds appeared to be haphazardly implemented, indicating a lack of planning when it came to installing BMPs. The random placement of BMPs was shown by the PCoA results from the BMP summary metrics, which indicated a large amount of variation in terms of how the BMP metrics varied from one another was in the flow accumulation that occurred at each BMP. If BMPs were being targeted to areas that would maximize the amount of pollutants that they could intercept or filter from surface runoff, then they majority of BMPs would have a high flow accumulation because surface runoff to rivers is the main transport mechanism for pollutants (Galzki et al., 2011; Gburek & Sharpley, 1998; Marjerison et al., 2011; Panagopoulos et al., 2011; Piechnik et al., 2012; Pionke et al., 2000). However, the voluntary nature of the RWQP may be why BMPs are being placed at random across the landscape, as opposed to a lack of planning from the RWQP in terms of BMP placement. Typically adoption rates are low when it comes to BMP programs (Yates and Bailey 2007; Wang et al., 2002). Therefore, it is likely difficult to turn down farmers who are willing to implement conservation projects, even if their farms are in hydrologically disconnected areas, which may be why the distribution of BMPs in the Nith and Conestoga subwatersheds appears to be haphazardly done.

5.2 Influence of BMP metrics on water parameters

In my study, in-stream nutrient and sediment measurements generally decreased in association with BMP metrics. Water parameters are commonly measured to assess the effectiveness of BMPs (Park et al., 1994; Gabel et al., 2012; Easton et al., 2008; Panagopoulos et al., 2011; Tuppad et al., 2010; Hamlet & Epp, 1994; Bosch et al., 2013). Reductions in water quality parameters, similar to my results, have been seen in other studies that have assessed the effectiveness of agricultural BMPs (Park et al., 1994; Tuppad et al., 2010; Gabel et al., 2012). The association between BMP type metrics and lowered concentrations of TN ($R^2 = 0.38$) and TP ($R^2 = 0.22$) in my study were also seen by Park et al. (1994), who observed reductions in TN (42%) and TP (35%) at a watershed scale after BMPs were implemented. Hamlet and Epp (1994) also demonstrated through modelling techniques that while BMP effectiveness differs based on BMP type and location, BMPs that reduce surface runoff (i.e., EC structures) can effectively reduce P and N losses from the landscape to river systems. TSS was shown to decrease over time with the implementation of BMPs in a north central Texas watershed (Tuppad et al., 2010), which follows the results in my study of reduced TSS in association with BMP metrics. Additionally, Yates et al. (2007) also saw that minimal BMP implementation was needed in the Upper Thames River Watershed, an intensely farmed area west of the GRW, to begin seeing less sediment in river systems. Thus, in my study BMP metrics were generally associated with improved water quality (i.e., a reduction in nutrient and sediment concentrations), although these results may be misleading due to the large influence from riparian vegetation. Riparian vegetation was shown to be associated with all 9 water parameters, and indicated a reduction in nutrient and sediment concentrations when riparian vegetation was present. Conversely, all other BMP type metrics were only associated with 4 water parameters, and indicated an increase in one circumstance (i.e., NH_4^+ increased with LAR % flow accumulation). Past studies have also found similar positive results in water quality when riparian vegetation was present along stream edges (Muenz et al., 2006; Osborne & Kovavic., 1993; Dunn et al., 2011). Results from the regression analysis indicated that the % riparian area metric was associated with reductions in TSS, which was supported by results from Muenz et al. (2006) who showed that buffered streams had lower, and more stable concentrations of TSS when compared to unbuffered streams. Osborne and Kovavic (1993) demonstrated how forested and grassed riparian buffers can significantly reduce nitrate and TP concentrations in streams by filtering N and acting as a sink for P. My study also found similar results, finding that riparian vegetation was associated with reduced TN and TP concentrations. Although my study did not control for the vegetation width within the 30 m buffer, even narrow widths of riparian vegetation can improve water quality parameters (Dunn et al., 2011). Dunn et al. (2011) observed reductions in pesticides, SRP, N, and sediment at both 10 m and 30 m buffer widths on operational farms in Prince Edward Island over a 6-year study. A reason why riparian vegetation may be often linked to improved water quality is because from a strategic placement perspective of BMPs, the location of riparian vegetation offers the greatest potential to intercept and filter pollutants from the landscape before they enter a river system because of its close proximity to river systems. Thus, my results support previous studies that BMPs, especially riparian vegetation, are an effective tool to mitigate sediment and nutrient losses from agricultural lands into river systems.

Results of this study indicated that BMP location may be an important consideration when implementing BMPs with the goal of reducing sediment and nutrients losses from agricultural lands to streams in the Nith and Conestoga subwatersheds. Along with riparian vegetation, spatial metrics (i.e., flow accumulation) were the only metrics I found to be significantly associated with water quality parameters. Past studies have also found that the location of BMPs in areas that are hydrologically active significantly reduced nutrient and sediment concentrations in river systems when compared to BMPs in hydrological disconnected areas (Bosch et al.,2013; Easton et al., 2008; Tim et al., 1995; Tomer et al., 2003). For example, Bosch et al. (2013) conducted a modelling study of 6 large watersheds that drain into Lake Erie, including the GRW, and predicted that that nutrient levels would decrease only if BMPs (i.e., cover crops, filter strips, no-till BMPs) were located in high nutrient source locations, and sediments levels would only decrease if BMPs were located near the river outlet. Both of these locations (i.e., high source locations, near the river outlet) would be assumed to have a large flow accumulation, which supports the results of my study that only flow accumulation metrics were significantly associated with water parameters. Although spatial metrics were not calculated for riparian vegetation in my study, the location of riparian vegetation along stream edges increases the upland area that the vegetation can filter pollutants from (i.e., a large flow accumulation area). Therefore, riparian vegetation is generally hydrologically connected due to its large flow accumulation area and close proximity to the stream, which is likely why riparian vegetation was associated with decreases in all 9 water parameters. The positive association that riparian vegetation had with water quality is supported by Tomer et al. (2003), who used terrain analysis to determine that the strategic placement of riparian vegetation along specific stream edges that intercepted surface runoff from large upland areas would significantly reduce surface runoff pollution from entering waterways. In their study, they concluded that large amounts of the riparian zone needed little or no riparian vegetation due to a lack of upslope runoff, whereas other areas needed a significant amount of riparian vegetation because they would intercept surface runoff from a large portion of the watershed (Tomer et al., 2003). Their study shows how different locations on the landscape are associated with increased risk of pollution from surface runoff, which may be why a location specific BMP metric (i.e., flow accumulation) was associated with water quality parameters. Therefore, my results support past studies findings that the location of BMPs likely influences their ability to mitigate nutrients and sediments from entering river systems due to the disproportionate amount of surface runoff that is generated from certain areas on the landscape.

5.3 Association between benthic macroinvertebrate communities and BMPs

The results of my study indicated that variation in the structure of benthic macroinvertebrate communities was only poorly explained by BMP abundance and location. This finding was contrary to my prediction that catchments with BMPs located in areas with a high DHC would be associated with improved ecological conditions, which would be represented by benthic macroinvertebrate community structures with greater richness and abundance of pollutant-intolerant species. Past studies have also found that BMPs were only weakly associated with benthic macroinvertebrates in agriculturally dominated watersheds (Gabel et al., 2012; Nerbonne & Vondracek, 2001; Wilcock et al., 2010; Yates et al., 2007). However, I hypothesized that past studies had limited power to detect patterns in the benthic macroinvertebrate structure associated with the implementation of BMPs because landscape level variables (i.e., land use, soil type) were not controlled for (Nerbonne & Vondracek, 2001; Sovell et al., 2000; Yates et al., 2007). Landscape level variables are known to be key drivers of benthic macroinvertebrate communities (Cuffney et al., 2000; Yates & Bailey, 2010a), but these landscape descriptors are also know to covary with each other as well as with smaller scaled variables, such as habitat and water quality (Yates and Bailey, 2006). Uncontrolled variation in landscape scale descriptors can thus mask relationships between small scaled variables and benthic macroinvertebrate community structure (Richards et al., 1996; Yates and Bailey, 2010c). I hypothesized that controlling the large scale factors throughout the landscape (i.e., land use, soil type) would result in a clear correlation between BMP use and benthic macroinvertebrates because the abundance and location of BMPs would be the only changing variable throughout the sampled catchments. However, as shown by the PCA results, the benthic macroinvertebrates invertebrate communities did not significantly vary across the 30 sampled sites, which could be characterized as a fairly homogenous, pollution-tolerant group of taxa. Therefore, it appears likely that as landscape level variables became more homogenized, the ecosystem conditions of rivers also became homogenized. These changes to the river system are conceptually consistent with previous studies that have shown that the large scale land use of a catchment can be used as a predictor of stream assemblages (Nash et al., 2009; Richards et al., 1996). In particular, agriculture has been shown to negatively influence water and habitat quality by increasing in-stream sediment, decreasing stream depth heterogeneity, decreasing substrate complexity, and altering the hydrologic regime (see review by Allan, 2004; Soininen & Könönen, 2004; Walser & Bart, 1999). Therefore, the overall influence of extensive agriculture in the Nith and Conestoga subwatersheds has likely created homogenous watershed characteristics in terms of water quality and habitat (i.e., increased concentrations of sediment and nutrients, pools and runs become more common, riffles become rare), which limited the amount of benthic macroinvertebrate species that could survive in those conditions, and is why benthic macroinvertebrates did not vary throughout the 30 sampled catchments.

I hypothesized that the presence of BMPs on the landscape would be associated with variability in the benthic macroinvertebrate community structure because BMPs have been shown to mitigate excess nutrients and sediment from entering river systems (Gabel et al., 2012; Marshall et al., 2008; Mayer et al., 2007; Yates et al., 2006; Barton & Farmer, 1997; Herendeen & Glazier, 2009; Park et al. 1994; Walker & Graczyk, 1993), which negatively influences many sensitive benthic macroinvertebrate species (Barton & Metcalfe-Smith, 1992; Barbour et al., 1996; Reynoldson et al., 1997). The presence of BMPs at the local scale was predicted to mitigate the overarching negative influences from agriculture, and would create heterogeneity in the water and habitat quality throughout the 30 sampled sites that had different BMP implementation rates. The anticipated differences in water and habitat quality would then create a more diverse assemblage of benthic macroinvertebrate communities within catchments that had more BMPs and/or BMPs with a high DHC. However, differences in BMP use and location throughout the Nith and Conestoga subwatersheds were only weakly associated with variations in the benthic macroinvertebrates community structure, despite improved water quality being associated with BMP metrics. A lack of association between the benthic macroinvertebrates and the BMPs may be because of one or all of the following four reasons: 1) BMP implementation rates were too low to overcome a water quality or habitat threshold that is limiting benthic macroinvertebrate communities; 2) certain BMP types that were shown to be associated with benthic macroinvertebrates were too sparse; 3) management goals were focused on improving water quality, not the ecological conditions, and; 4) there was a lack of nearby source populations of sensitive taxa to repopulate the streams with improved water quality.

Improvements in water quality parameters were associated with BMP metrics, but these improvements did not translate into a more diverse, pollutant-intolerant benthic macroinvertebrate community structure. Past studies have claimed that a threshold effect may occur between aquatic biota and ecosystem conditions (Cuffney et al., 2000; Gabel et al., 2012; Yates et al., 2007), which hinders aquatic biota populations that cannot tolerate a certain degree of impairment in water quality or habitat availability. Yates et al. (2007) began to see a non-linear relationship between BMPs and ecosystem quality in a southern Ontario watershed. They concluded that to detect an improvement in overall ecosystem quality, a certain degree of BMP implementation was required within a catchment (Yates et al., 2007). Although BMP implementation rates were relatively higher in my study comparted to Yates et al. (2007), BMP abundance may still be too low to initiate a change in benthic macroinvertebrate community structure. Cuffney et al. (2000) found that relatively low concentrations of pollutants that are commonly associated with agriculture (e.g., turbidity, total nitrogen, dissolved ammonia, total phosphorus) resulted in a rapid decline in benthic macroinvertebrate community condition in their study, indicating a threshold effect with minimal agriculture present, which may explain why BMPs could not overcome the large scale influence of intensive agriculture in my study. Determining when a threshold effect occurs along a gradient of stressors can be difficult due to the various interactions that occur at a watershed scale that may influence how a species responds to changing ecosystem conditions (Kaller and Hartman, 1999; Wang et al., 2002), but it is likely location specific. It can be assumed though that intensive and extensive BMP implementation is likely needed to create significant improvements in water quality and aquatic biota assemblages in heavily farmed regions (Tuppad et al., 2010). Past studies have shown that BMP implementation rates must reach a certain level before ecological conditions significantly improve (Bosch et al., 2013; Moore & Palmer, 2005; Wang et al, 2002). For example, Moore and Palmer (2005) conducted a study in headwater catchments where 65% of farms had BMPs (i.e., no-till cultivation, riparian buffers) and found that their streams had higher levels of macroinvertebrate diversity when compared to other studies in agricultural regions. Specifically in the GRW, Bosch et al. (2013) modelled BMP implementation across the entire GRW and determined that moderate BMP implementation (i.e., cover crops, filter strips, and no-till) that covered only 25% of agricultural land would result in modest reductions (10%) of sediment and nutrient concentrations. Additionally, Wang et al. (2002) stated that in order for improvements to be seen in aquatic biota, 30-50% of farms must be engaged in BMP implementation. While 39% of farms in my study had BMPs, the majority of BMPs were MS structures, which have been shown to have little influence on water quality (Easton et al., 2008). There may be numerous thresholds in both water quality and habitat that need to be overcome to allow the ecological conditions to improve, which may be why the benthic macroinvertebrates are not responding to the current water quality conditions in the Nith and Conestoga subwatersheds. Therefore, the addition of more BMPs is likely required to overcome a possible threshold effect in these heavily farmed regions because when BMP implementation rates are generally low (e.g., below 30-50% of farms with BMPs), improvements in the ecological conditions of streams are unlikely to occur (Wang et al., 2002).

The types of BMPs being implemented in a watershed appear to be as important as the abundance of BMPs when it comes to mitigating the impacts from surface runoff pollution. Certain BMP types (i.e., LARs, EC structures) that were less abundant in my study were still associated with nearly the same amount of improved water parameters as the most prominent BMP (i.e., MS structures). From the regression analysis on the water parameters, riparian vegetation was the only BMP metric to explain some of the variation in all 9 water parameters, while the most abundant BMP (MS structures, n = 79) could only explain some of the variation in 4 water parameters. A less common BMP, LARs (n = 40), almost matched MS structures in their ability to explain the variation in water parameters by being associated with reductions in 3 water parameters. For the benthic macroinvertebrates, the results from the stepwise ordination regression revealed that BMPs that were few in numbers (i.e., EC structures, LAR) were significantly associated with benthic macroinvertebrates. While 1 MS structure (n = 79) metric was associated with the benthic macroinvertebrates, so too was 1 EC structure (n = 10) metric and 2 LAR (n= 40) metrics. This finding suggests that fewer BMPs of a certain type may be capable of a similar influence on both the water quality and ecological conditions of a river system. The influence from certain BMP types (i.e., EC structures, LARs) that are sparse in numbers may be due to their ability of intercept or filter more pollutants from the landscape. Along with riparian vegetation, LARs can reduce the direct input of fecal matter and associated nutrients into rivers from livestock (Collins et al., 2007; Easton et al., 2008) and EC structures are often placed in areas where they are needed due to significant erosion issues (e.g., areas prone to riling or gullying). Additionally, LARs and riparian vegetation are only installed adjacent to a stream, which puts them in an optimal position to help filter pollutants before they reach the stream. Conversely, BMPs that do not actively filter or intercept large amounts of surface runoff (i.e., MS structures) have been shown to have little influence on river systems because they are typically only associated with surface runoff at the barnyard scale (Easton et al., 2008). Therefore, BMPs that filter or intercept large amounts of surface runoff (i.e., riparian buffers, LARs, EC structures) may be more beneficial to river systems in terms of reducing nutrient and sediment concentrations, which would be assumed to improve water quality and ecological conditions. However, these BMP types (i.e., riparian buffers, LARs, EC structures) were likely too sparse to significantly influence the benthic macroinvertebrate community structure in the Nith and Conestoga subwatersheds.

Current management goals of the RWQP are aimed at improving the water quality in the GRW (GRCA, 2014), as opposed to improving the ecological conditions of the GRW, which may be why benthic macroinvertebrates did not respond to the current BMPs being implemented. As stated earlier, agriculture has been shown to negatively affect habitat quality for aquatic biota by increasing in-stream sediment, decreasing stream depth heterogeneity, decreasing substrate complexity, and altering the hydrologic regime (see review by Allan, 2004; Soininen & Könönen, 2004; Walser & Bart, 1999). The current BMPs being promoted by the RWQP do not actively promote the rehabilitation or creation of in-stream habitat, which may be limiting the diversity of benthic macroinvertebrate species that can survive in the current homogenous habitat that was observed in the Nith and Conestoga subwatersheds (i.e., abundance of runs, very few pools and riffles, minimal woody debris). BMP types that promote in-stream habitat creation are likely needed to create habitat heterogeneity (e.g., two-stage ditches), although the lack of benefits for the farmers from such BMPs may make adoption difficult. Nevertheless, a lack of such BMPs that promote in-stream habitat may be a reason why benthic macroinvertebrates were not associated with BMP implementation in the Nith and Conestoga subwatersheds.

The final reason why benthic invertebrates may not be responding to the modest improvements in water quality is that source populations of sensitive taxa to repopulate the rivers are absent. The GRW, which contains the Nith and Conestoga subwatersheds, is a heavily impacted and fragmented landscape consisting of over 75% agricultural land and has had 90% of the original forests cleared (Yates & Bailey, 2010b; Holysh, et al. 2000). Previous studies have mentioned the lack of nearby source populations as a potential reason why aquatic biota may have not responded to in-stream or water quality improvements as anticipated (Parkyn et al. 2003; Wilcock et al., 2010). Wilcock et al. (2010) saw significant improvement in water quality that was associated with riparian fencing, but these improvements in water quality did not result in changes to the benthic macroinvertebrate community structure, which is similar to the lack of response from benthic macroinvertebrates in my study to changes in water quality. Parkyn et al. (2003) also observed improvements in water clarity and channel stability with the installation of riparian vegetation buffers. However, significant changes in macroinvertebrate communities in response to these improvements did not occur, which they claimed may be due to a lack of source populations and pathways for recolonization of restored sites to occur (Parkyn et al. 2003). From an in-stream habitat perspective, Sundermann et al. (2011) found that while restoration work was successful in terms of increasing in-stream microhabitat heterogeneity, a lack of source populations in the region likely led to benthic macroinvertebrate assemblages that were still very similar to unrestored sites. While my analysis of the 30 sampled catchments did not include a landscape level analysis of nearby forested areas or potential pathways for new taxa to travel and disperse, the intensely farmed and populated region of the GRW likely limited the possibility of there being nearby source populations of sensitive taxa. Therefore, recolonization of sensitive taxa in heavily impacted regions may not occur due to a lack of source populations and pathways, even if water quality and in-stream habitat are improved.

6.0 Management Implications and Recommendations

From the findings in my study, it is difficult to demonstrate that BMPs should continue to be promoted to improve the ecological conditions in river systems because they were only weakly associated with benthic macroinvertebrate community structure. However, the improvements seen in the water quality parameters, along with the potential for a threshold effect, should permit for the continued encouragement of farmers to implement BMPs. However, to see significant changes in the benthic macroinvertebrate community structure, the following action plans are likely needed: 1) overall BMP implementation rates should be increased and targeted to areas that experience significant surface runoff, 2) certain BMP types that are known to intercept surface runoff from large upland areas need to be better promoted, and 3) additional BMP types aimed at in-stream habitat creation may need to be promoted. With the current modest BMP implementation rate in the Nith and Conestoga subwatersheds (0.39 BMPs/farm), additional BMPs are likely needed to see a shift in benthic macroinvertebrate communities. In intensely farmed areas, such as the Nith and Conestoga subwatersheds, one must expect that intensive BMP implementation is needed to mitigate the impacts from agriculture. Past studies have also advocated that to achieve significant changes in river systems that are in heavily farmed regions, such as the GRW, intensive BMP implementation is needed (Bosch et al., 2013; Tuppad et al., 2010), which may help overcome a potential threshold effect in water quality and/or habitat availability that may be limiting benthic macroinvertebrate communities. Furthermore, BMPs should be targeted to areas on the landscape that intercept overland flow from large upland areas to maximize the potential of BMPs to intercept overland surface runoff that may contain pollutants before it reaches the river system. Effective management strategies need to be custom-tailored to the region in which they are being implemented to maximize their effectiveness due to changes in landscape characteristics that can influence BMP performance (Tomer et al., 2003).

Based on my findings, BMP types that intercept surface flow from large upland areas (i.e., riparian buffers, EC structures, LARs) should be installed over other types of BMPs. These BMPs have shown that they are associated with the reductions in water quality parameters in the Nith and Conestoga subwatersheds, even when they are not as abundant as other BMP structures. In particular, maintenance and planting of riparian vegetation should continue to be supported by the RWQP. In this study, riparian vegetation was shown to be associated with reductions of all the water parameters tested, even though riparian vegetation covered only 23% of the buffer zone on average. Furthermore, riparian vegetation has many added benefits to nearby wildlife, downstream populations, and receiving water bodies (see review by Lovell and Sullivan, 2006; Hudon & Carignan, 2008). If riparian vegetation is already present on the landscape, it is a cost-effective BMP to maintain. BMPs that do not actively filter pollutants (i.e., MS structures) should not be a priority when it comes to implementing and funding BMPs for conservation purposes because they appear to have little association with water quality and benthic macroinvertebrates. However, exceptions may be necessary if manure is stored in areas that have a high DHC, which increases the risk of manure laden runoff entering rivers.

Conservation goals must be aligned with the function and purpose of BMPs. The main goal of the RWQP is to improve water quality within the GRW (GRCA, 2014), which is why the majority of BMPs are aimed at mitigating pollutants from entering the stream (e.g., EC structures, LARs) or protecting stream edges (i.e., tree planting, riparian area restoration). However, there are not many BMPs that actively improve the in-stream habitat structure for aquatic biota, which may be limiting the diversity of benthic macroinvertebrates in these agricultural streams. One potential BMP that could improve in-stream habitat is the two-stage ditch (Figure 6.1). A two-stage ditch consists of a meandering deep main channel with an adjacent floodplain channel that will hold water during periods of high flow (Powell et al., 2007). The main purpose of a two-stage ditch is to create a stable ditch system that transports water and sediment more efficiently, but there are potential added benefits such as reduced maintenance for farmers and improved water quality and habitat (Powell et al., 2007). The literature on two-stage ditches is currently very limited, but it is a promising option that is worth considering in areas where recreating natural meandering streams may not be an option. Furthermore, the effectiveness of BMPs, especially those that require the establishment of in-stream habitat, may require additional effort and time before improvements in ecological conditions are seen because natural systems can take decades to fully recover from agricultural impacts (Harding et al., 1998). Past studies have found it can takes years (3+) to detect a positive ecological change when BMPs are implemented (Carline & Spotts, 1998; Stuber, 1985: Wang et al., 2002). Therefore, conservation goals must incorporate the needs of aquatic biota if rivers in the Nith and Conestoga subwatersheds are to resemble natural systems with diverse assemblages of benthic macroinvertebrates.

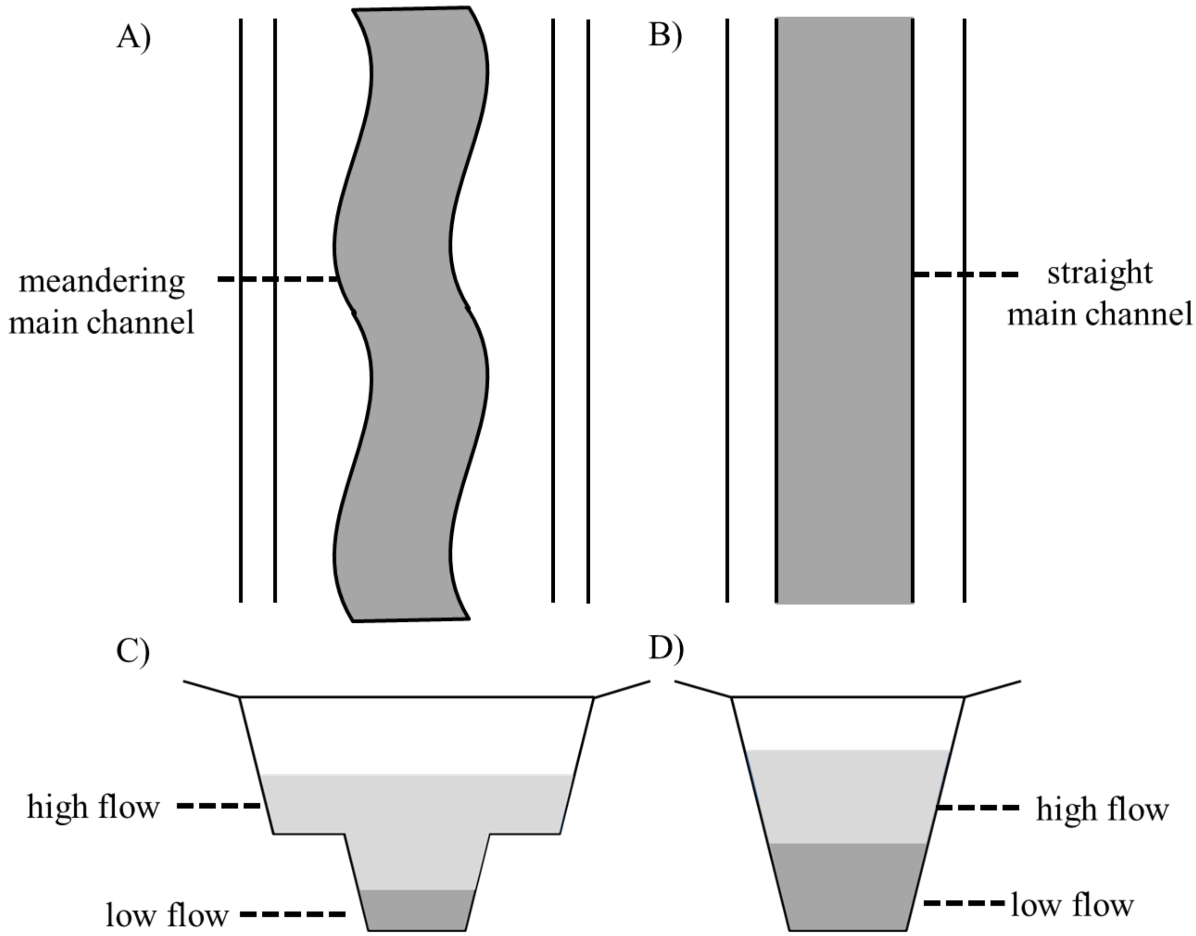

Figure 6.1: Diagram of two-stage ditch with meandering deep main channel (A) compared to straight main channel (B) from a conventional drainage ditch and the adjacent floodplain channel (C) in a two-stage ditch compared to homogenous channel depth (D) from a conventional drainage ditch.

7.0 Future Research

My study showed that both the current number and placement of BMPs is not sufficient to improve benthic macroinvertebrate communities in the Nith and Conestoga subwatersheds. However, the process in which BMPs are being assessed for their ability to improve ecosystem conditions may be giving misleading results due to the confounding interaction between the numbers of farms and BMPs. The confounding interaction is that the number of BMPs implemented in a catchment is likely proportional to the number of farms in the catchment, which consequently is also likely to increase the number of potential stressors within the catchment that the BMPs must mitigate. Therefore, catchments with numerous BMPs are likely the catchments that have numerous potential sources of pollutants. Although BMPs may be successful at intercepting and filtering pollutants, the large amount of stressors within a catchment under intensive agriculture may still result in a stream environment with low species diversity and/or excess nutrients and sediment. Assessing the effectiveness of BMPs based solely on their abundance, without looking at the farming intensity within that catchment, may thus not be an accurate representation of whether BMPs are performing effectively. Therefore, future BMP assessment needs to account for the farming intensity in the region to determine if BMPs are having a net benefit on the ecological condition and water quality of river systems. This approach would require removing the effects of farming intensity to create an unbiased measure of the BMPs performance. Once we can eliminate the confounding influence between BMP implementation and farming intensity, we will better understand the ability of BMPs to mitigate agricultural impacts on ecological conditions and water quality.

A second research need is to establish baseline ecosystem conditions prior to BMP implementation. What was not known for my study was the past ecosystem conditions of the

Nith and Conestoga subwatersheds. Therefore, improvements seen in the water quality are difficult to directly link to the implementation of BMPs because extraneous variables (e.g., land use change) may be influencing the water quality along with BMPs. These extraneous variables need to also be assessed over time to determine how the abundance and location of potential stressors on the landscape change. An understanding of pre-BMP ecosystem conditions would allow future research to better understand how the water quality and aquatic biota change over time in response to both extraneous variables and BMP implementation. Pre-BMP monitoring of the ecosystems conditions and landscape variables would be required, and has obvious issues such as funding for prolonged monitoring, although modest monitoring effort of a select few streams may provide insight into how BMPs directly influence ecosystem conditions.

Future research also needs to determine how, or if, aquatic biota assemblages can shift if presented with improved water quality parameters and habitat. Currently, it is assumed that as chemical conditions improve so too will the aquatic biota (i.e., benthic macroinvertebrates). However, the ability of aquatic biota to repopulate to a new habitat may be limited by a lack of surrounding source populations, or distance limitations of certain species to migrate (Parkyn et al. 2003; Sundermann et al., 2011; Wilcock et al., 2010). Therefore, as ecological conditions improve, we may just find a greater abundance in pollution-tolerant species rather than the addition of new pollution-intolerant species. Determining the potential for recolonization would require a landscape assessment of potential reference, or least degraded sites, where source populations would be expected to inhabit. Additionally, understanding the distance that benthic macroinvertebrates can travel would create realistic expectations about which benthic macroinvertebrates may repopulate a recently restored site.

8.0 Conclusions

Structural BMPs being implemented by the RWQP were positively associated with improved water quality conditions, but were weakly associated with benthic macroinvertebrate community structure. BMPs should continue to be promoted in the Nith and Conestoga subwatersheds because of the modest improvements seen in the water quality, and for the potential to overcome a water quality and habitat threshold that may be hindering the benthic macroinvertebrate communities. Additional efforts may be needed to raise the BMP implementation rates in the GRW, most notably BMPs that filter pollutants from large upland areas. BMPs also need to be targeted to areas that are hydrologically connected to the river system so that they can filter pollutants before they reach the river. Future research on BMPs needs to eliminate the confounding influence between BMP implementation and farming, assess pre-post BMP implementation ecosystem conditions to better link BMP use with benthic macroinvertebrate communities, and determine the ability of surrounding source populations of benthic invertebrates to repopulate streams.

9.0 References

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Appendix A

Habitat assessment field data sheet for low gradient streams that was used to assess the habitat characteristics in the 30 sampled catchments in the Nith and Conestoga subwatersheds.

HABITAT ASSESSMENT FIELD DATA SHEET—LOW GRADIENT STREAMS (FRONT)

STREAM NAME	LOCATION	
STATION #________ RIVERMILE________	STREAM CLASS	
LAT ____________ LONG ____________	RIVER BASIN	
STORET #	AGENCY	
INVESTIGATORS		
FORM COMPLETED BY	DATE ________ TIME ________ AM PM	REASON FOR SURVEY

Parameters to be evaluated in sampling reach

Habitat Parameter	Condition Category			
	Optimal	Suboptimal	Marginal	Poor
1. Epifaunal Substrate/ Available Cover	Greater than 50% of substrate favorable for epifaunal colonization and fish cover; mix of snags, submerged logs, undercut banks, cobble or other stable habitat and at stage to allow full colonization potential (i.e., logs/snags that are not new fall and not transient).	30-50% mix of stable habitat; well-suited for full colonization potential; adequate habitat for maintenance of populations; presence of additional substrate in the form of newfall, but not yet prepared for colonization (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 of 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

HABITAT ASSESSMENT FIELD DATA SHEET—LOW GRADIENT STREAMS (BACK)

Parameters to be evaluated broader than sampling reach

Habitat Parameter	Condition Category			
	Optimal	**Suboptimal**	**Marginal**	**Poor**
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 past 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
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 1 to 2 times longer than if it was in a straight line.	The bends in the stream increase the stream length 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 ___ (LB)	Left Bank 10 9	8 7 6	5 4 3	2 1 0
SCORE ___ (RB)	Right Bank 10 9	8 7 6	5 4 3	2 1 0
9. Vegetative Protection (score each bank) Note: determine left or right side by facing downstream.	More than 90% of the streambank surfaces and immediate riparian zone covered by native vegetation, including trees, understory shrubs, or nonwoody 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 centimeters or less in average stubble height.
SCORE ___ (LB)	Left Bank 10 9	8 7 6	5 4 3	2 1 0
SCORE ___ (RB)	Right Bank 10 9	8 7 6	5 4 3	2 1 0
10. Riparian Vegetative Zone Width (score each bank riparian zone)	Width of riparian zone >18 meters; human activities (i.e., parking lots, roadbeds, clear-cuts, lawns, or crops) have not impacted zone.	Width of riparian zone 12-18 meters; human activities have impacted zone only minimally.	Width of riparian zone 6-12 meters; human activities have impacted zone a great deal.	Width of riparian zone <6 meters: little or no riparian vegetation due to human activities.
SCORE ___ (LB)	Left Bank 10 9	8 7 6	5 4 3	2 1 0
SCORE ___ (RB)	Right Bank 10 9	8 7 6	5 4 3	2 1 0

Total Score _________

Curriculum Vitae
Roger Holmes

Education

- MSc – Geography, University of Western Ontario
- BES – Bachelor of Environmental Studies, Honours, Environment and Resource Studies University of Waterloo – Sept 2007-Apr 2012
- Diploma in Ecological Restoration and Rehabilitation, University of Waterloo
- Diploma in Environmental Assessment, University of Waterloo

Skills and Areas of Experience

- Benthic macroinvertebrate identification and collection skills
- Canadian Biomonitoring Network (CABIN) certified
- Class 2 Backpack Electrofishing certified
- Water chemistry collection experience (i.e., grab samples and with in-situ probe)
- Proficient using GIS for field work and research purposes (ArcMAP, ArcHydro)
- Experience handling, identifying, and monitoring turtle and fish species
- Full G licence and Pleasure Craft Operator Card
- Extensively worked with data books and online databases
- Knowledgeable on conservation and restoration techniques
- First Aid and WHMIS certified

Employment History

University of Western Ontario – Sept 2012-Aug 2014

Title: Research/Teaching Assistant

- Collected benthic macroinvertebrate and water chemistry samples for numerous projects following the CABIN protocol.
- Planned and participated in field research projects throughout the Grand River Watershed
- Contacted landowners and conservation authorities to engage in research initiatives.
- Presented MSc research at the Society of Freshwater Sciences conference in Portland, OR.
- Developed and delivered lecture material for university classes

Ontario Ministry of Transportation – Jan 2009-Apr 2009; Sept 2009-Dec 2009

Title: Assistant Environmental Planner

- Assisted on a variety of MTO highway projects during the Class Environmental Assessment process
- Reviewed environmental documents and reports from MTO and consultant staff
- Conducted numerous site visits with MTO staff to observe and monitor projects for environmental issues
- Drafted various environmental clearances and other correspondence for MTO staff

- Prepared agendas and took minutes at various environmental meetings
- Attended Public Information Centres and assisted MTO and consultant staff

Toronto Zoo – May 2010-August 2010; May 2011-August 2011

Title: Biologist Assistant/Conservationist Assistant

- Conducted field research on the Blanding's turtle in the Oakland Swamp complex
- Trapped, tagged, and tracked turtles using radio-telemetry and GPS to assess population size, health, and behavioural patterns
- Communicated and worked with landowners on habitat creation and protection projects
- Organized and participated in nesting surveys and nest protection initiatives with local landowners and volunteers
- Designed, planned, and constructed an artificial nesting site on a local landowners property
- Assisted in the design and creation of a booklet for landowners to conserve wetlands and turtle habitat

Agriculture and Agri-Food Canada – Jan 2011-Apr 2011

Title: Assistant Agro-Ecosystem Analyst

- Analyzed the use and effectiveness of Best Management Practises (BMP's) being implemented throughout Canada for manure storage on farms
- Interviewed key experts and gathered information on potential impacts from climate changes on Canada's crop industry
- Created two reports on manure storage BMP's and climate change impacts on crops
- Prepared and presented both papers to entire policy research division

Relevant University Courses and Projects

- MSc Thesis Topic
    - Described and assessed the association between the structure of stream benthic macroinvertebrate communities and the number and location of agricultural BMPs relative to the position of hydrologically connected areas
    - Informed BMP implementation programs aimed at mitigating agricultural impacts on river systems.
    - Extensive ArcGIS work delineating watersheds and assessing BMP implementation by analyzing aerial photos.
    - Field sampling of benthic macroinvertebrates and water chemistry throughout the Nith and Conestoga subwatersheds.
- BES Thesis Topic
    - Developed habitat suitability model for largemouth bass in lake environments in southern Ontario
    - Used water quality measurements, vegetation classification, and water depth profiles to determine habitat quality and create a ranking system

References

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